European Energy Policy in Transition:
Critical Aspects of Emissions Trading

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To my mother and father

Contents*

* Until 2008, Markus Wråke's surname was 'Åhman'.

Acknowledgments

Huge thanks go to to my parents for their love that has supported me always, and to my sister, whose friendship and personality I appreciate more and more every day that passes.

Finally, my deepest, most heart-felt love goes to Anna and Frida. I am so lucky to have you.

Stockholm, June 17, 2009

Markus Wråke

Introduction

Climate change represents a unique challenge to policy making. It is a global problem that will affect generations to come. Its causes lie at the core of the lifestyles of western societies, lifestyles which many more people aspire to have. While consensus around climate change is growing across the world, introducing ambitious policies is still politically difficult. Ethical considerations are inevitable when discussing measures to reduce emissions, how to adapt to the effects of climate change, and who should bear the costs of these efforts.

The science of climate change is complex and there are aspects of the problem that are still poorly understood. However, even conservative scenarios of potential effects that climate change could have on the global economy indicate that the problem constitutes a market failure of grand scale.

Economic analysis of climate change, consequently, has to take a global perspective, consider long time horizons, deal with high uncertainty, and include the possibility of large, non-marginal changes in technologies and resource distribution.

Climate change can be described as a classic public goods problem; the capacity of the atmosphere, the oceans, and the terrestrial systems to assimilate the greenhouse gases, which human and natural activities add to the atmosphere and which warm the earth, has no owner and access is unrestricted. Market-based policy instruments are one way of dealing with a market failure such as this, and the theory of incentive-based environmental regulation is one of the most important contributions from the field of economics to public policy.

The aim of this thesis is to analyse the design and implementation of a particular market-based policy instrument, namely, emissions trading systems.[1] Emissions trading addresses the public goods problem by rationing the access to the resource (in this case, the atmosphere) and privatising the resulting access right (in this case, the right to emit CO_2). Another route imposes a tax on emissions. In principle, an optimal cap in a system, such as the European Union's Emissions Trading Scheme (EU ETS), should be set where the damage of an additional ton of emissions equals the cost of not emitting that last ton. If figured in this way, the cap would deliver a market price equalling an optimal tax level.[2]

Thus, in principle, an emissions trading system and a tax could deliver the same outcome, at the same cost to society. However, there is an economic debate over the relative merits of taxes and emissions trading. Among other issues, it revolves around the nature and level of the uncertainties in the information about both damages to the environment resulting from emissions and about the costs associated with reducing emission. A tax fixes the price of emissions, but leaves the volume of emissions undetermined, whereas emissions trading sets the total emissions volume, but leaves the costs uncertain. If one believes that the damages of emitting one additional ton of CO_2 into the atmosphere is relatively small and constant, and that the costs of reducing emissions are uncertain and could rise rapidly, a tax would be preferable to emissions trading. Instead, if there is concern that thresholds may exist above which damages could increase very quickly, a policy which imposes an absolute cap on emissions would be safer.[3]

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[1] Unless otherwise specified, 'emissions trading' refers to a system where entities can trade emissions permits under a fixed cap. In the United States, the more descriptive term 'cap-and-trade' is often used for this version of emissions trading. There are other forms of emissions trading that which don't necessarily have an absolute cap on emissions, for example baseline and credit systems, such as the Clean Development Mechanism of the Kyoto Protocol.

[2] Cf. Pigou (1920).

[3] The key point, made by Weitzman (1974), is that the expected efficiency of the policies will depend on the relative slopes of the curves for marginal costs and marginal benefits of emissions reductions, as well as the associated uncertainties in these curves.

The term 'efficiency' appears frequently throughout this thesis. By an 'efficient' policy, I mean one that can deliver the identified objectives of the policy at lowest total cost to society. This can be interpreted in a static sense, for example, assuming fixed reduction targets and available technologies, or in a dynamic context, where second order effects (such as technology development, altered trade flows, etc.) are also considered. I also use the term 'distribution' in many places. The distribution of the costs of climate policies across households, industries, and countries may be equally important in the process of designing policy instruments as are the efficiency properties of the different policy options.

European environmental policy has traditionally relied heavily on command and control type policies. These can take the form of plant-level permits which stipulate how much pollution each plant is allowed to emit or specific requirements on the technologies firm can use. For consumer products, performance standards and labelling are often used, for example, EU fuel standards for passenger cars and the requirement for energy performance labelling on household appliances, such as refrigerators and washing machines. Such measures are often introduced for good reason, but they fall short of creating fully efficient incentives throughout the economy. For example, a compulsory fuel standard for cars will force car manufacturers to improve the fuel efficiency of the cars they produce, but it will not give them incentives to go *beyond* the standard. Nor will a fuel standard give people who buy the car any incentives to change their behaviour and drive less.[4] By contrast, a tax on petrol gives the car manufacturer an incentive to produce more fuel-efficient cars in order to make them cheaper to use and more attractive to customers, and it also changes behaviour as the cost of petrol at the pump increases. Europe also has, on average, considerably higher fuel taxes than other countries on other continents, and this particular instrument appears to have actually led to the largest reductions in carbon emissions.

The argument for pricing carbon emissions in general is analogous: it creates incentives for reducing emissions, stimulates innovation in low-carbon technologies, and drives substitution of lower carbon fuels, products, and services throughout the economy. Market-based instruments may also reduce problems related to information asymmetries. For instance, setting an appropriate fuel standard requires in-depth knowledge about available engine technologies— knowledge that industry may possess, but which is difficult for the regulator to obtain.

The efficiency advantages of market-based instruments have made them increasingly popular with policy makers. Europe primarily used taxes and charges, and emissions trading was regarded with scepticism until the late 1990s.

The first paper in this thesis recounts how the concept of emissions trading was gradually accepted in Europe, and eventually resulted in the launch of the EU ETS. We go on to analyse some of the most contentious issues that have emerged and conclude by prospecting the future, highlighting important revisions of the trading systems and some of the questions that remain unresolved.

The picture that emerges is one of a process that was coloured by political pragmatism and industry lobbying, where the objective to get the buy in from important private stakeholders was a priority for policy makers. The result was a trading system with many flaws and which probably has not yet spurred any significant emissions reductions over and above business-as-usual.

However, we also see the development of an institutional infrastructure that can be valuable for the future of European energy and climate policy. What is more, the initial years of the EU ETS have provided a large-scale testing ground for emissions trading, offering opportunities for needed institutional learning and practical market experience. The lessons learned are diverse

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[4] In fact, the result could be rather the opposite since the improved fuel efficiency will make the car cheaper to use. This is usually referred to as the *rebound effect*.

and not all experiences are positive, but the accomplishment of creating a common carbon price across a large part of the EU economy should not be underestimated. Policy makers in Europe and elsewhere would be wise to make use of the information gained from the EU ETS, be they supporters of emissions trading or sceptics to such policies.

A central feature in any emissions trading system is how the permits to pollute—the emission allowances—are initially allocated to participants. Allocation is also a recurring topic in this thesis. A fundamental choice is whether firms should receive allowances free of charge or whether they should have to pay for them, for example, via an auction. Because the emissions allowances in the EU ETS represent a substantial monetary value—approximately € 35 billion annually at current prices—how they are distributed is of great economic interest to many stakeholders.

The EU ETS is set up in trading periods, or phases. Phase I, also known as the trial period or pilot phase, ran from 2005–2007. Phase II, which is ongoing at the time of this writing, coincides with the first commitment period of the Kyoto Protocol and is longer, 2008–2012. The third phase will be extended even longer, from 2013–2020.

The primary allocation method of choice in phase I and phase II distributed the allowances for free. If done as a one-off gift, based on historical activities, this should not in itself affect incentives for firm production choices or investment decisions. Nor should the allocation methodology affect how firms price their products; a profit-maximising firm should include the value of the allowances in its pricing strategy, regardless how they obtained their allowances. Nevertheless, there has been an intense debate over the effect that the EU ETS allocation has had on product prices, most visible in the electricity sector.

In paper 2, we hope to shed light on this issue. Using experimental methods, we look at whether the pricing strategies of firms in competitive markets will differ depending on whether they receive the allowances for free or must pay for them. Participants initially display a variety of pricing strategies. However, given a simple economic setting where earnings depend on behaviour, we find that subjects learn to consider the value of allowances and their overall behaviour moves toward that predicted by economic theory.

The free allocation used in the EU ETS deviates in many respects from the textbook version. In phases I and II, member states are responsible for National Allocation Plans, which govern the initial distribution of emission allowances. Significant discretion regarding the specifics of the allocation was given to the member states, which resulted in a plethora of different methodologies. The allocation procedures have been complex and opaque, and have had important implications for efficiency, as well as the perceived fairness of the trading system by the public. Several papers in this thesis look closer at the effects the allocation has had on incentives for firm production decisions and investments.

In paper 3, we examine the rules governing allocations to installations[5] that close and to new entrants. We find that the treatment of such installations by member states is inconsistent with the general guidelines provided by the EU, which seek to discourage allocation methodologies that produce incentives affecting firms' compliance behaviour, for example, by rewarding one type of investment over another. We propose stronger EU guidance on firm closures and new entrants, a more precise compensation criterion by which to justify free allocations, and a ten-year rule as a feature of future EU policy to guide a transition from current practice to one with greater weight on efficiency.

Paper 4 looks at one of the most fundamental questions in relation to the EU ETS: to what extent does the price of carbon drive emissions reductions? We focus on the electricity sector

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[5] An 'installation' is the official EU term for a factory or a plant-emitting CO, i.e., the entity that must comply in the EU ETS.

because it was widely regarded as having more low-cost abatement opportunities than other sectors covered by the EU ETS. In the aggregate, supply of emissions allowances in an emissions trading system will be constant, given the cap on total emissions. If demand for allowances in certain sectors of the economy increases, this will push the price of the allowances up. Because marginal abatement costs vary across firms and sectors, their emissions elasticities, in regard to change in allowance price, will be different. If the Swedish electricity sector does have lower marginal abatement costs than other sectors, it is more likely to adjust its demand for emissions allowances in response to price variations in the market than sectors with higher marginal costs for emissions reductions. Hence, the EU ETS would have a visible impact on the CO_2 intensity of electricity generation, even though total emissions in the economy are constant. We use an econometric time series analysis to study the relationship between the price of carbon emissions and the carbon intensity of Swedish power generation in the period 2004–2008. We find no indication that the price of carbon had an impact on the carbon intensity of electricity generation. Hence, we conclude that either the ex ante assumption—that there exists more low-cost abatement opportunities in the power sector compared to other sectors—was wrong or there exist other and stronger drivers of the use of fossil fuels in Swedish power generation, which diminish the effect of the EU ETS on carbon intensity in the sector.

In paper 5, we return to the issue of allocation, focusing on the treatment of firms that enter the EU ETS, specifically in the power sector. We analyse the impact of allocation to new entrants and identify options for improved regulation. The discussion compares the allocations in phases I and II of the EU ETS to two hypothetical energy installations located in different EU member states. The study focuses on the Nordic countries and their integrated energy market. The quantitative analysis was complemented by interviews with policy-makers and industry representatives. The results suggest that current allocation rules can significantly distort competition. The annual value of the allocation is comparable to the fixed investment costs for a new installation and is not insignificant, compared to expected revenues from sales of electricity from the installation. We find that the preferred option for the Nordic countries is not to allocate free allowances to new entrants in the energy sector. It should be combined with adjusted rules on allocation to existing installations and closures in order to avoid putting new installations at a disadvantage. A second, less-preferred choice suggests harmonized benchmarks across the Nordic countries for the allocation.

Paper 6 also uses the power sector as the reference for the analysis. We quantify the volume of free allowances that member states proposed to allocate to existing and new installations in phase II of the EU ETS. Most countries continue to allocate based on historic emissions, contrary to hopes for improved allocation methods, frequently using 2005 emission data. We draw the conclusion that this may strengthen the belief by the private sector that emissions in the coming years will influence their subsequent allowance allocation. Allocations to new installations translate into large (and frequently fuel-differentiated) subsidies, which risk significant distortions to in investment choices. Thus, in addition to supplying a long market in aggregate, proposed allocation plans reveal continuing diverse problems, including perverse incentives. We conclude that in order to ensure the efficiency of the EU ETS in the future, the private sector will need to see credible evidence that free allowance allocation will be drastically reduced post-2012, or that these problems will be addressed in some other way.

Paper 7 investigates four alternative methodologies for free allocation based on historical activities that were under discussion before the allocation methodologies for phase I had been established. The allocation methodologies were evaluated against the criteria for a National Allocation Plan[6] and their conformity with the criteria introduced by the Swedish Parliamentary Delegation on Flexible Mechanisms (the FlexMex 2 Commission), which did a substantial part of the preparatory work in Sweden ahead of the launch of the EU ETS. We find that no allocation

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[6] Listed in annex III of the EU ETS Directive (European Union 2003).

methodology unambiguously meets all criteria. Emission-based allocation is most straightforward, transparent, and the easiest to implement. Production-based allocation meets more of the criteria, but is more costly to implement and requires more data. Due to the lack of abatement cost curves, it is not possible to accurately model potential capital flows between the trading sectors, but we believe it is unlikely that any given allocation scheme will be perceived as fair by all concerned parties, no matter how sophisticated it is. A final conclusion is that data availability probably limits the options available to the authorities designing the allocation schemes. For example, data on best available technology was not available in time in the allocation in phase I of the EU ETS.

The last paper in the thesis, paper 8, has a slightly different character than the others. It evaluates the climate impact from the use of peat for energy production in Sweden. Although it only contributes marginally to the European energy system, the use of peat continues to draw significant political attention in some member states, including Sweden, Finland, and Ireland. As the planning of EU ETS progressed and details were revealed, there was growing concern in the peat industry and in some political camps that the way emissions from the use of peat for energy purposes were calculated was incorrect and would make peat unattractive from an economic standpoint. In the paper, we apply a dynamic energy model to study the effect on climate change from the use of peat, measured as the contribution to atmospheric radiative forcing when using $1\ m^2$ of mire for peat extraction over a 20-year period. Two different methods of after treatment of the mire were studied: restoration of wetlands and afforestation. The climate impacts from peatlands–wetland scenario and a peatlands–forestation–bioenergy scenario are compared to the climate impacts from coal, natural gas, and forest residues for energy generation. Sensitivity analyses are performed to evaluate which parameters are important to take into consideration to minimize the climate impact from peat utilisation. In a 'multiple generation scenario', we investigate the climate impact if 1 megajoule of energy is produced from peat every year for 300 years and compare it to other energy sources. The results are sensitive to what after-treatment is used and what time horizon is applied. In a majority of the scenarios, however, the climate impact of peat is lower than if coal is used to generate the energy, but higher than the corresponding values for natural gas and forest residues.

A final remark is that the popularity of market-based policy instruments, manifested by the introduction of the EU ETS, has by no means supplanted other types of policies, such as subsidies and command and control instruments. The sometimes implicit motivation is that the politically acceptable price of emissions (or the tax level) will be 'too low' to induce the changes that are needed in the economy. (The EU subsidies for carbon capture and storage facilities are one example where the price of carbon emissions are not expected to be high enough to simulate sufficient research and deployment of a new technology.) In addition, overlapping policy objectives are too common, such as the EU targets for both the proportion of renewable energy sources and overall emissions of greenhouse gases. Often, the rationales underlying policy objectives differ, even though some of the effects overlap. For example, greater use of renewable energy sources not only reduces carbon emissions but also decreases the EU's dependence on imported fossil fuels and helps improve security of supply, which is a growing concern of the EU. The interaction between different types of policies and policy objectives deserve more attention by the academic community and policy makers alike, and it is an area for important future research which is touched on only briefly in this thesis.

References

European Union. 2003. Directive 2003/87/EC of the European Parliament and the European Council of 13 October 2003 establishing a scheme for greenhouse gas emission allowance trading within the Community and amending the Council Directive 96/61/EC.

Pigou, A.C. 19202. *The Economics of Welfare*. London, UK: Macmillan and Co. (The 1932 4th edition is available online at http://www.econlib.org/library/NPDBooks/Pigou/pgEW.html. Accessed June 2009.)

Weitzman, M. L. 1974. Prices vs. quantities. *Review of Economic Studies* 41: 477–91.

Paper I

Emissions Trading:
The Ugly Duckling in European Climate Policy?

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Abstract

The initial years of the European Union's Emissions Trading System (EU ETS) have provided a large-scale testing ground for trading of a new environmental commodity, carbon dioxide. This paper provides an overview of the origins and characteristics of the EU ETS. It then goes on to analyse the most contentious issues that have been discussed in the economics literature and in the public debate surrounding the trading system. The lessons learned are diverse and not all experiences are positive. Nevertheless, invaluable information has been gained from the EU ETS and policy makers in Europe and elsewhere would be wise to make use of it, be they supporters of emissions trading or sceptics to such policies. The paper concludes with a look toward the future, highlighting some upcoming revisions of the EU ETS and at what issues remain unresolved.

Introduction

The initial years of the European Union's Emissions Trading System (EU ETS) have been a large-scale testing ground for trading a new environmental commodity, carbon dioxide (CO_2). In its current form, the EU ETS includes some 12 000 installations, representing approximately 45% of EU emissions of CO_2. It is by far the largest emissions trading system in the world. This paper provides an overview of the origins and characteristics of the EU ETS and analyses the most contentious issues surrounding it in the economics literature and in public debate. It concludes with a look towards the future, highlighting some major forthcoming revisions of the EU ETS and what issues remain unresolved.

European environmental policy has traditionally been dominated by command and control-type policy instruments. When market-based instruments have been used, they have primarily

been taxes. Most countries in Europe have high fuel taxes (which are at least partly motivated by environmental considerations), and some countries have taxes or charges on waste, sulphur, nitrogen, and other emissions. Alternative market-based instruments, such as refunded emission payments, deposit refunds, and subsidies, among others, are used in various areas.[1]

As concern about climate change rose on the political agenda in the early 1990s, the European Commission made efforts to set up a common European carbon tax, but this work met intense resistance from industry and some member states, as well as from many finance ministries which were anxious to keep exclusive national sovereignty in this area. As a result, the political momentum gradually shifted away from a common tax and no strong agreement was reached. Emissions trading was widely regarded with great scepticism in Europe at the time, and the experience with this type of policy instrument was limited. The political turnabout that ultimately resulted in the creation of the EU ETS has been reviewed extensively in the political economy literature.[2]

A central factor in the shift in the EU position was the adoption of the Kyoto Protocol in 1997, which included emissions trading as one of the "flexible mechanisms" along with the Clean Development Mechanism (CDM) and Joint Implementation (JI). Although the EU strongly opposed the US-led push to include flexible mechanisms in the Protocol, the final outcome of the negotiations in Kyoto propelled emissions trading into the mainstream political debate in Europe. In the five years that followed, the discussion of how and when to implement an emissions trading system for private entities evolved from narrow academic circles to a much broader set of stakeholders

The remainder of the paper is structured as follows. Section 1 describes the motivation and decision-making process for setting up the EU ETS, as well as the fundamental characteristics of the system. Section 2 discusses some of the most contentious issues that have emerged in the EU ETS to date. Section 3 looks towards the future and what lies ahead for the EU ETS, and concludes.

1. From Unwanted Idea to Directive

The Kyoto Protocol[3] required signatories to show "demonstrable progress" in reducing emissions by 2005. The EU quickly determined that an internal emissions trading system could potentially show such progress and the first official EU document indicating the possibility of a European pilot trading system appeared in 1998.[4]

Basing an emissions trading system on article 17 of the Kyoto Protocol, which lays out the principles for emissions trading between countries, was quickly identified as an option. This structure would delegate the trading of assigned amount units[5] to private entities and the principles, rules, and protocols of the trading regime would be decided by the Conference of the

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[1] For an overview, see for instance the OECD Environmentally Related Taxes database, www.oecd.org/env/policies/database (accessed June 2009).

[2] See, for instance, Skjaerseth and Wettestad (2008) and Christiansen and Wettestad (2003) for accounts from a political science perspective.

[3] UNFCCC (1998). The Kyoto Protocol can be downloaded from http://unfccc.int/resource/docs/convkp/kpeng.pdf.

[4] European Commission (1998).

[5] Parties with commitments under the Kyoto Protocol (annex B) have accepted targets for limiting or reducing emissions. These targets are expressed as levels of allowed emissions, or "assigned amounts," over the 2008–2012 commitment period. The allowed emissions are divided into "assigned amount units" (AAUs), each equal to 1 ton of CO_2 equivalent.

Parties (COP).[6] Such a set up seemed to offer more advantages, particularly regarding harmonisation and compatibility, but given the likely difficulties in achieving consensus across all parties on such a detailed level, it was discarded as unrealistic for Europe.

Another early design proposed setting up individual member-state emissions trading systems with the option of linking them into a common European system.[7] The rules and provisions of each system would be decided by each member state, with article 17 of the Kyoto protocol serving as a loose framework. Member states would have significant flexibility to accommodate national circumstances and interests, but this would also create potential problems with harmonisation and compatibility. Although support for this option persisted into the 2000s, most observers agreed that a common EU approach would be preferable to linking a large number of individual national systems.[8] Two member states, Denmark and the U.K., went ahead and set up their own national emissions trading systems for greenhouse gases, partly to gain experience before a common European system came into play. Some firms also tested internal emissions trading systems several years before the start of EU ETS. BP's system received extensive public attention. Its design and function deviated in many respects from a textbook cap-and-trade system, and no money actually changed hands, but the system effectively raised awareness of the opportunities to save money with emissions reduction and how emissions trading could work in practice.[9]

In 2000 the EU published its "Green Paper on Emissions Trading."[10] It analysed the critical factors for an EU trading system and outlined some preferred design options. In less than two years, the EU Commission published its proposal for the EU ETS Directive,[11] which differed in two principal ways from the Green Paper's recommendations on allocation procedures. First, it chose a decentralised approach, giving significant discretion to the member states regarding the number of allowances they could allocate. Second, it proposed that the initial allowances be allocated free of charge as the basic allocation principle for the first trading period 2005–2007.

In the negotiations between the European Parliament (EP) and the European Council that followed, it quickly became clear that the EP would like to see a larger proportion of allowances allocated by auction and broader coverage of the system, whereas the Council largely defended the Commission proposal. The mounting political pressure to get a directive accepted during 2003 resulted in an agreement in July 2003, and the final directive was published in the *EU Official Journal* on October 25, 2003. The outcome was close to the original proposal, and its key features were a largely decentralised approach to allocation and at least 95% of allowances allocated free of charge. The system covered CO_2 emissions from four main 'activities':[12]

- Energy, including combustion installations with a rated thermal input above 20MW, mineral oil refineries, and coke ovens

- Production and processing of ferrous metals, including metal ore and production of pig iron and steel

- Mineral industry, including production of cement, glass, and ceramic products

- Other activities, including pulp and paper production

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[6] The COP is the collection of nations which have ratified the UN Framework Convention on Climate Change (UNFCCC). The primary role of the COP is to oversee the implementation of the Convention. The first COP took place in Berlin, March 28–April 7, 1995.

[7] This is basically the approach taken for trading green and white certificates (renewable electricity and energy savings, respectively).

[8] Zapfel and Vainio (2002) give an insider's perspective on the early development of the EU ETS.

[9] See Victor and House (2006) for an interview based analysis of BP's system.

[10] European Commission (2000).

[11] European Commission (2001).

[12] For exact definitions, see annex I of the EU ETS Directive (European Union 2003).

When it adopted the EU ETS Directive, the European Union went from the drawing board to practical implementation of an idea that, less than a decade earlier, had seemed impossible in Europe.

2. Contentious Issues in Phase I and II of the EU ETS (2005–2012)

This section briefly analyses some important features of the EU ETS. Although this account is by no means comprehensive, it offers an overview of the most contested issues and the arguments put forward in discussions about the design of the EU ETS.[13]

Setting the Cap

The environmental effect of a cap and trade system is governed by the total allocated volume of allowances.[14] The price of emissions and the resulting economic incentives for firms to reduce emissions are determined by the scarcity of allowances.

In the EU ETS (phases I and II), each member state is responsible for allocating allowances to the emissions-producing installations in its territory. The number of allowances given to each installation is spelled out in a National Allocation Plan, (NAP). The total cap in the trading system, thus, is the aggregate of all member state allocation plans. Member states have considerable discretion in deciding allocation methodology, but their NAPs must conform to a number of criteria set by the EU.[15]

In the first trading period, the European Commission aimed at ensuring that allocations were not to generous using two principal criteria. First, the total number of allowances proposed by the member state should be lower than business-as-usual projections, and second, the member state had to show that the intended allocations would achieve its target reduction set by the EU burden-sharing agreement or the Kyoto Protocol. (Both of these criteria had qualitative dimensions and were susceptible to different interpretations.)

The process of setting up the NAPs turned out to be complex and sometimes controversial, characterised by lobbying and strategic interaction between industry, member states, and the EU Commission.[16] An unfortunate consequence of the decentralised allocation procedure was that member state governments faced incentives that could lead to decisions that were not efficient for the trading program as a whole—the 'prisoner's dilemma'.[17] When a government decides on the rules for allocation, it is likely to consider the tax base and the job opportunities that installations provide. For instance, it may be rational, from a member state's point of view, to reward continued production in the own country or attempt to enhance the competitiveness of its own industry through the allocation, even though such measures may raise the overall social cost of the trading system.

Concerns over a 'race to the bottom' between member state allocations were augmented by the fact that not all NAPs were submitted at the same time. For example, the U.K. NAP was

[13] Omitted questions, in particular, include monitoring, reporting and verification, compliance and enforcement, and potential linkage of the EU ETS to other emerging trading systems.

[14] In practice, as Tietenberg (2002) notes, the level of the cap is determined not only by what may be socially optimal, but is also a function of the design of the trading system.

[15] See annex III of the EU ETS Directive.

[16] A detailed account of this process lies beyond the scope of this article. See, for example, Ellerman et al. (2007) for illustrative examples from ten member states.

[17] The prisoner's dilemma constitutes a problem in game theory. In the classic form, cooperating is strictly dominated by defecting, so that the only possible equilibrium for the game is for all players to defect, even though each player's individual reward would be greater if they played cooperatively. The term 'prisoner's dilemma' stems from the example used in its original form, with two hypothetical prisoners who were the participants in the game.

published early and judged to be relatively stringent. Once other member states published their NAPs—which turned out to be more lax—the U.K. filed a request to adjust its NAP and increase its allocation volumes. Although the request was disallowed by the Commission, this example indicates that the allocation process likely contained elements of strategic behaviour by member states. A centralised allocation at a European level, or at least a common decision on the total volumes to be allocated, would mitigate this problem. However, such an approach had little support among member states, several of which reluctantly endorsed the creation of the trading system.[18]

The European Commission decided to reduce the proposed totals in 14 of the 25 phase-I NAPs that were submitted by the member states, representing some 5% of the total cap.[19] Still, assessments by Zetterberg et al. (2004) and Gilbert, Bode, and Phylipsen (2004) indicated that the allocation was generous. Installations were given more allowances than their historical emissions warranted and they were also given more allowances than needed to carry an equal burden in relation to the EU Kyoto target (compared to sectors outside the trading system). Consequently, the trading system was criticised for not being stringent enough even before it was launched.

Nevertheless, the first year of trading saw prices of emission allowances (EUA),[20] which were higher than many observers had expected, peaking at over 30 €/ton early in 2006 (figure 1). This sparked calls from in particular the energy intensive industry to scrap the system, with

Figure 1   Price of EU Allowances in the EU ETS
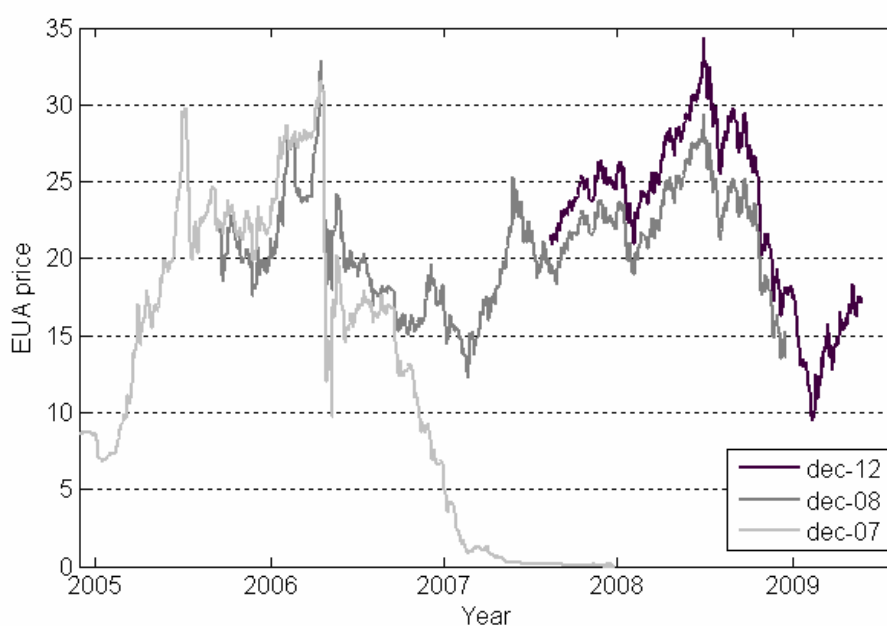
"dec -07" is the phase I futures contract for delivery in December 2007, and so on.
Source: Point Carbon

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[18] Skjaerseth and Wettestad (2008) categorised the member states by their positions on emissions trading into leaders (the Scandinavian countries, the Netherlands, the UK, Germany, and Austria), laggards (Greece, Spain, Portugal, and Ireland), and those in between (Belgium, Italy, Luxembourg, and France).

[19] Ellerman and Buchner (2007)

[20] EUA, or European Union allowances. These are the tradable asset in the EU ETS, each permit representing 1 ton of CO_2 emitted.

claims that it was hurting the economy. Most of these calls fell silent as the first 2005 verified numbers of emissions for 2005 were published in April 2006, showing that the market had too many allowances. This information caused EAU prices to fall dramatically. Although the immediate drop slowed and prices stabilised for a while, by mid-2007, they reached near-zero levels. This development supported the view that phase I had an over-allocation. The empirical literature assessing the effect of the EU ETS on abatement is still scarce, but it seems unlikely that phase I of the EU ETS led to significant reduction in CO_2 emissions compared to business-as-usual.[21]

Repeating this situation—very low allowance prices—in phase II (2008–2012) would have seriously jeopardised the credibility of the trading scheme. Furthermore, as the second phase coincided with the first commitment period in the Kyoto Protocol, a continued liberal allocation would implicitly impose large emission reductions on sectors not included in the trading scheme. Alternatively, the member states might have to make greater use of the CDM and JI in order to reach their reduction targets.[22] As a final resort, a member states could buy Kyoto emission credits (AAUs) from countries outside the EU ETS (for instance, Russia or Ukraine), but that would be politically controversial.

In order to avoid this situation, the EU Commission repeatedly stated its intention to tighten the cap during the second trading period, as member states prepared their NAPs for phase II. In a guidance document,[23] it laid out new principles for the NAPs, making verified emissions for 2005 the basic yardstick for the assessment.[24] But, although this reduced the occurrence of lofty sector growth projections that were widespread in the first set of NAPs,[25] early assessments of NAPs submitted for phase II suggested that allocations continued to be lavish.[26] This lent support to the EU Commission's actions limiting the allocation by requiring significant cutbacks in several of the proposed allocation plans.[27]

Although it is still too early to assess how much scarcity there is in the trading system, current EUA prices are back to the levels of 2005–2006. Market participants should have learnt enough to make the system work and information on emissions and allocations is more readily available and better understood, indicating that the cap is tighter in phase II than in phase I.

Free Allocation or Auction?

Emissions trading rations access to the resource—in this case, the atmosphere—and privatises the resulting access right—in this case, the right to emit CO_2. A central question is how the property rights (here, emission allowances) are initially distributed among participants, and a fundamental choice is whether firms should receive allowances for free or if they should have to pay for them, for example, in an auction. There is considerable discussion in the economics literature about the efficiency and equity properties of each option.

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[21] It is, however, difficult to determine to what extent abatement measures were implemented. See Ellerman and Buchner (2006) and Widerberg and Wråke (forthcoming 2009) for a deeper discussion.

[22] This option is limited by the Kyoto Protocol, which stated that JI and CDM should be "supplementary" to domestic action.

[23] Communication from the Commission on guidance to assist member states in the implementation of the criteria listed in annex III of Directive 2003/87/EC.

[24] The EU Commission even developed an explicit formula for the assessment: allocation = verified 2005 ETS emissions * GDP growth rates for 2005–2010, based on PRIMES model * carbon intensity improvements rate for 2005–2010 + adjustment for new entrants and other changes, for example in ETS coverage.

[25] See, for instance, the LETS Update (2006) for assessments of the projections.

[26] Rogge et al. (2007) and Neuhoff et al. (2006).

[27] In total, the EU Commission shaved off some 10% of the proposed allocation volumes.

The efficiency[28] of the trading system, in principle, does not hinge on whether the allocation is free of charge or not. The possibility of trading the allowances will ensure that they flow to the participants who value them most, no matter how they were initially distributed.[29] From this perspective, allocation is a matter of distribution of costs, not efficiency. Although the allocation may constitute a significant transfer of assets from governments to firms,[30] the allowance price, the environmental effectiveness of the system, choice of abatement method by firms, and downstream price effects should all be the same whether firms pay for allowances initially or not.[31]

A vast majority of earlier allowance trading systems implemented to manage fisheries, air pollution, and water resources have used free allocation based on historical activities—usually referred to as 'grandfathering'.[32] Classic grandfathering is a one-off initial allocation of allowances to existing installations, valid for a long time into the future. If these installations close, they still retain their allocation, while new entrants do not receive free allowances.

However, the grandfathering applied in the EU ETS (as in most, if not all, previous trading systems) deviates in many respects from the textbook version. The allocation procedures have been complex and opaque, and have damaged the perceived fairness of the trading system by the public. Further, a large body of research shows that the allocation methodologies used in the EU ETS so far have given perverse incentives to firms regarding how they reduce emissions and have distorted competition between firms, technologies, and member states. Grandfathering encourages regulated parties to engage in (potentially costly) rent-seeking behaviour in order to gain a more generous future allocation. Pointing to their high marginal costs for abatement has been a common strategy used by some industry sectors[33] to receive more allowances in the EU ETS. Some compensation to industries faced with more costly abatement measures or large sunk costs may be justified, but if signalling high abatement costs leads to higher future allocation, then investment in abatement measures may be delayed or guided to suboptimal technologies. Harstad and Eskeland (2007) show that, under conditions with high allowance prices and frequent revisits of the allocation,[34] the distortions can be greater than the gains from trade, implying that non-tradable emission allowances may be better.

Most of the potential pitfalls associated with grandfathering were already known before the EU ETS was launched, but two principal justifications were typically put forward for its use, regardless. First, it increased the chances that participants would agree to the trading system in the first place. Grandfathering would decrease the financial burden on participating firms and would offer a situation closer to the status quo than an auction, thus reducing resistance from incumbent emitters.

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[28] Efficiency, in this context, is defined as the ability to reduce emissions to a predetermined level at minimum abatement cost. This can be interpreted in a static sense, e.g., assuming fixed reduction targets and available technologies, or in a dynamic context, where second order effects and incentives are also considered.

[29] See Montgomery (1972) and a related paper by Baumol and Oates (1971), which demonstrate that a correctly defined tradable allowance system under specific conditions, including a sustainability constraint, can maximise the value received from the resource.

[30] In fact, in the EU ETS, the value of those assets is much greater than the costs that the firms face for compliance. See figure 1 in Åhman et al. (2007).

[31] However, as described by Harrison et al. (2007), certain conditions, such as negligible transaction cost, perfect competition, and low costs of emissions (relative other costs and the overall value of output), are also necessary for this ideal situation to hold.

[32] Notable exceptions are the U.S. SO_2 allowance program, and the Regional Greenhouse Gas Initiative, which rely on auctions to allocate a portion of the allowances.

[33] See, for example, "Position Paper of the Alliance of Energy Intensive Industries on 'further guidance on allocation plans for 2008 to 2012'", February 2, 2006 (http://www.cembureau.be/Cem_warehouse/AEII-FINAL POSITION-GUIDANCE ON ALLOCATION PLANS-2008 TO 2012.PDF [accessed May 2009]).

[34] See the next subsection, "Updating, New Entrants, and Closures" for further discussion.

Second, free distribution based on historical experience arises from a public policy rationale or desire to compensate incumbent installations affected by the regulation. Schultze (1977) argues that people feel that government should 'do no direct harm' when imposing new public policy. This rationale implies a specific amount of compensation proportional to the change in the economic value of installations caused by the program.

Both of these arguments carry some weight. Auctions are (and are still) opposed by important sectors of industry, as well as by some member states. The steel and cement industries, in particular, have actively voiced their concerns over the additional costs an auction would force on them. Both individual companies and their business associations argue that auctions would be economically detrimental to them, referring to the international competition that they face from firms outside the EU ETS.[35] Considering the lobbying power and economic importance of these industries in Europe, it would be difficult politically to introduce auctions for all allowances in the first phase of the EU ETS. The argument for compensation is also correct in principle, but begs the question 'how much is enough'. The answer depends on how the policy affects the profitability of the firm, which in turn depends on the change in firm (total) revenues and costs. Empirical evidence suggest that the amount of revenue needed, in the form of free allocation, to avoid losses to firm shareholders, is only a fraction of the total revenues returned from auctions (Bovenberg et al. 2000; Bovenberg and Goulder 2001; Burtraw et al. 2006; Hepburn et al. 2006b).

The economic literature broadly supports auctions as a more efficient way to distribute allowances, compared to free allocation.[36] Auction revenues can be recycled in ways that may enhance the efficiency of the economy as a whole, for example, by reducing distortionary taxes.[37] Sometimes, it is argued, that there is a 'double-dividend', meaning that not only can the trading system achieve the environmental objective, but the efficiency gains made possible by the recycled auction revenues can make the net cost of the policy negative.[38] Even though the support for the double-dividend argument in the literature is ambiguous, it is clear that auctions give the regulator more flexibility to reduce other distortions in the economy or increase investments in areas important for climate policy (e.g., research and technology).

Auctions also promote innovation, relative to grandfathering, since the incentives to innovate (and thereby reduce abatement costs and ultimately allowance prices), are higher if firms do not receive any rents from free allowances.[39] The effect is true in the aggregate and the difference between auctions and free allocation decreases as the time between the innovation and the allocation grows. This is because, as Cramton and Kerr (2002) point out, the incentive to innovate depends on who owns the allowances at the time of innovation.

In addition, if markets are not fully competitive, free allocation can move consumer prices away from the marginal social cost of production and, therefore, may direct (via distortion)

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[35] See, for example, "EUROFER position paper on ETS, October 2008" (http://www.eurofer.org/index.php/eng/content/pdf/776); the press release from the Swedish Steel producers association, Elisabeth Nilsson, "Gratis utsläppsrätter är än så länge en förutsättning för stålindustrins globala konkurrenskraft" [Free allowances are still a prerequisite for the steel industry's global competitiveness], *Jernkontoret*, May 4, 2009 (http://www.jernkontoret.se/jernkontoret/pressmeddelanden/2009/vdkommentar_090504_utslappsratter.pdf [accessed June 2009]); and the Cembureau position paper, "Climate Change: CO2 Emissions Trading—Points of Convergence within the Cement Industry" (http://www.cembureau.be/Cem_warehouse/POINTS%20OF%20CONVERGENCE%20WITHIN%20THE%20CEMENT%20INDUSTRY.PDF [all accessed May 2009]).

[36] See, for instance, Cramton and Kerr (2002), Hepburn et al. (2006a), Dinan and Rogers (2002), and Lange (2005) for discussions about carbon emissions trading and the EU ETS.

[37] See, for example, Parry (1995) and Parry et al. (1998).

[38] The incidence of the cost of the trading system depends crucially on how the revenues are distributed, as shown by Burtraw et al. (2009).

[39] Milliman and Prince (1989), Fischer et al (2003)

resource allocation away from an efficient outcome. This effect may be more significant in electricity markets, which are key to reducing carbon emission, but often are not fully competitive.[40] Furthermore, an auction may improve administrative transparency and the perception of fairness, compared to grandfathering, which are crucial to the formation of a new market for an environmental commodity.

In sum, it seems clear that the grandfathering applied in the first two trading periods of the EU ETS has affected not only the distribution of costs, but also the economic efficiency and the environmental effectiveness of the system. A transition to auctions can resolve many of these problems since the case for continued grandfathering, in its current form, is weak. Although the proportion of allowances auctioned in the EU ETS is still very low (<5%), small but important steps have been taken to phase out free allocation. In several member states, the energy sector still receives a significantly reduced allocation in the second phase—an indication that free allocation is not necessary in the system.[41]

Updating, New Entrants, and Closures

In principle, pure grandfathering should be a 'one time only' gift that does not affect firm behaviour, but experience in the EU ETS and previous trading systems shows that such a model is politically difficult to implement. Instead, explicitly or implicitly, various forms of updating are usually applied, implying that future allocation is based on current firm behaviour (for example, output, input, or emissions). It is clear that such an allocation mechanism does influence firm incentives.

The official position of the EU Commission has, from the outset, been that the EU ETS should not contain any updating. This view has not always been shared by all member states or industries. For example, Germany included updating in its first NAP, seeking to adjust the allocation to firms whose emissions changed significantly during the trading period. This was contested by the EU Commission in a legal process that forced Germany to change its NAP and remove the updating clause.

The efficiency characteristics of updating depend on whether the market is open or closed to outside participants and on whether the regulator has control over the allocation of all emissions in the market.[42] There is significant support for an open-system approach in the analysis of allocation in the EU ETS. Member states face a dilemma when the allowance price, determined by the total allocated volume of all member states, can be regarded as exogenous. Furthermore, since the EU ETS only covers a fraction of the economy, there is interplay between the allocation to firms under the EU ETS and the emissions that the rest of the economy is allowed to produce. The EU ETS also has an explicit link to the outside through the CDM and JI. Finally, experience shows that it is tricky for a regulator to disregard the past when future caps are set and, thus, the cap is likely to be flexible rather than absolute.

In this setting, distortion from updating moves incentives away from optimum. Consider a firm that in period t produces output q_t as a function of input h_t. Production generates emissions e_t, and the firm has abatement opportunities a_t. The firm sells the produced good at price p_t^q.

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[40] For a detailed analysis of this in a U.S. context, see Burtraw et al (2001).

[41] The reduction of free allowances to the energy sector is also closely linked to the discussion of windfall profits, see separate discussion below. The total share of auctioning is low in phase II - approximately 4 % of the total volume of allowances – although that number does not include the zero-allocation to some energy firms (they have to buy allowances on the secondary market rather than in an auction.

[42] For example, Böhringer and Lange (2005) suggest that if there is an absolute cap on emissions and the market is closed to trading with the outside, basing the allocation on past emissions with updating can also produce a first best solution. Rosendahl (2008), however, argues that even under closed market conditions an efficient solution may be infeasible and permit prices will significantly exceed marginal abatement costs.

The firm's aggregated cost function is $c_t = c_t(q_t, h_t, a_t)$. At the start of each time period t, the firm receives an allocation $\bar{e}_t$ and the firm can comply with the cap either by buying allowances at price p_t^e or by reducing emissions through abatement. Solving the first order conditions for the firm's maximising problem gives $p_t^q = c_{t,q_t}' + p_t^e e_{t,q_t}'$, where c_{t,q_t}' is the marginal cost of production and $p_t^e e_{t,q_t}'$ is the opportunity cost of the marginal emissions. Sterner and Muller (2008), by applying this model[43] in a two-period setting, show that updated allocation methodologies have a suboptimal price (and output) effect. For example, if the allocation in the second period, $\bar{e}_{t+1}$, depends on the production in the current period, q_t, solving the first order conditions of the firm's profit maximising problem yields the solution $p_t^q = c_{t,q_t}' + p_t^e e_{t,q_t}' - \dfrac{\varepsilon}{1+r} p_{t+1}^e$, where ε is the emission intensity of production and r is the discount rate. Put differently, the expectation of increased allocation in period $t+1$ increases the value of output in period t. The allocation, thus, acts as an output subsidy, lowering the price of the produced good and diminishing the output effect of the policy.

Nevertheless, updating is both explicitly and implicitly part of the design of the EU ETS—explicitly through the renewed allocation for each trading period, and implicitly through the treatment of new entrants and installations that close. The debate on this issue has been focused on whether new entrants should receive any allowances at all and, if so, what specific allocation rules are appropriate. A parallel discussion on whether a closed installation should retain its allocation exists, although it receives less political attention. In addition, the definitions of a 'new entrant' and 'closure' are not clear cut.[44]

All member states have set aside a reserve of free allowances to allocate to new entrants; it is clearly a political priority. The sizes of the reserves differ, as do exact allocation methodologies, although most member states use some set of benchmarks to allocate allowances to new entrants. For installations that close, the prevailing policy is to withdraw the allocation. In some member states, the allocation can be transferred to a new installation or for an increase in capacity in an existing one.

The energy sector is worth a special note. Most member states allocate more allowances to high emitters than low emitters, for both existing and new facilities. For instance, a coal-fired power plant often receives more allowances than one that runs on natural gas. This reduces incentives to develop low-carbon technologies, particularly by new entrants and closures. Åhman et al. (2007) use a two-period model, similar to Sterner and Muller (2008), but provide additional context in order to illustrate the potential effects of the rules in the EU ETS. By introducing representative fixed and variable costs of electricity generation for different plant types, they calculate costs going forward for firms considering investment and for firms contemplating closing installations. The findings show that the treatment of closures and new entrants in the electricity sector, during the first two trading periods, can affect what types of new plants are built and which old plants are retired, and that high-emitting plants have an advantage over low-emitting ones.[45]

Thus, both theoretical analyses and applied research indicate that the current rules on new entrants and closures create distortions between member states, between new and existing installations, and among technologies. The methodologies differ greatly among member states,

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[43] In their model, Sterner and Muller also include an input that is unrelated to emissions in order to capture potential effects on the relative factor use under different allocation schemes. That part of the analysis is not central here and has been omitted from this discussion.

[44] The formal definitions of new entrants and closures applied in the EU ETS are found in the EU ETS directive.

[45] For additional analyses of this issue, see, for example, Neuhoff et al. (2007) and Åhman and Holmgren (2007).

and prevent a level playing field across the internal market. The policy of withdrawing the allocation from installations that close constitutes an implicit subsidy to existing installations still in operation,[46] thus putting new entrants at a disadvantage. In many cases, the allocation methodologies to new entrants do not encourage low-carbon technologies.

Again, many of these problems would be solved if free allocation were replaced by auctions. But, if free allocation continues, harmonising the rules for new entrants and closures will be a first step towards a more efficient system. Preferably, installations that close should retain their allocation at least long enough to reduce the incentive to keep an inefficient installation in operation just to receive the allocation. This would reduce the implicit subsidy to existing installations and allow a more stringent allocation to new entrants—or even no free allocation at all.

EU ETS and the Electricity Sector

An important objective of cap and trade is to alter relative prices throughout the economy by including the social cost of pollution in product prices. At the same time, higher retail prices for goods, such as electricity, may be a concern to policy makers and politically controversial. Energy-intensive industries are important to several member state economies and possess strong lobbying power. To many of those industries, the indirect effects of increased electricity price are more important than the direct costs of allowances. For example, studies of the British (Hourcade et al. 2007), German (Graichen et al. 2008,) and Swedish (Zetterberg and Holmgren, 2009) industries show that, in the aluminium sector, the paper industry, and the inorganic chemical sectors, the indirect costs in the form of higher electricity prices are significantly higher than direct costs for emission allowances.[47] In addition, low-income households typically spend a higher proportion of their disposable income on energy, which means that changes in electricity prices tend to be regressive,[48] adding to the political sensitivity.

As a result, the debate over effects of the EU ETS on electricity prices has several thorny dimensions.[49] Concerns over negative effects on energy intensive industries and criticism related to equity and fairness have featured prominently, while some observers have questioned whether prices have been—or should be— affected at all, given that allowances in most cases were allocated free of charge.

Two kinds[50] of 'windfall profits' are recurring themes in this context. Firms make direct windfall profits when they are given more allowances than they need and then sell their excess on the market. Indirect windfalls may be even more important, however. Because electricity prices in competitive markets are determined by marginal production—which in Europe is dominated by fossil fuel—electricity generators can charge a higher price for all their generated electricity, including that from nuclear, biomass, and hydro.

As discussed earlier, economic theory argues that, at least in competitive markets, retail prices will include the economic value of the allowances, and hence increase under an emissions trading system, whether polluting entities received allowances for free or not. Nevertheless, many people disagree and disapprove when firms raise product prices, especially if it means

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[46] One could also impose, equivalently, a tax on closures. This effect is analogous to the output effect studied formally by Sterner and Muller (2008), as discussed previously.

[47] As these sectors are subject to international competition and have limited ability to pass on increased costs to their customers, this is highly relevant to the discussion on competitiveness of European industry and carbon leakage. See sub-section, 'Competitiveness and Carbon Leakage', below.

[48] This effect is part of a broader debate around 'fuel poverty' that has been particularly intense in the UK, where specific measures have been implemented to compensate low-income households for increasing energy prices.

[49] See Åhman et al. (2008) for an overview of the debate and illustrative examples for the Nordic electricity market.

[50] In fact, both of these effects are present in other sectors as well, but are most pronounced in the electricity sector.

they reap substantial windfall profits. Although the argument may be discounted by economists, free allocation is frequently put forward as a means of reducing downstream price effects. Occasionally, industry has reinforced this view. For example, one energy company official told Point Carbon, that 'if EUAs are auctioned that will only lead to 100 percent of the carbon price being priced into the electricity price, and thus increase it'.[51]

Similar arguments have also been raised by governmental authorities. For instance, the German Federal Cartel Office (Bundeskartellamt) in 2006 stated that the industrial electricity prices charged by RWE[52] in 2005 were abusive, as the company had passed on more than 25% of the value of its CO_2 emission allowances to consumers in higher electricity prices.[53] In simplified terms, the Bundeskartellamt recognized that, in principle, the opportunity costs should be taken into account as a business calculation. However, the authorities took the view that, since the allowances were necessary for the electricity generation of RWE, they were not actually for sale and, thus, would not have a full opportunity cost associated with them. Because the allowances had been allocated to the firm free of charge, the firm should not be entitled to include their full value in the electricity price.[54]

The list of such examples can go on and on, but there remain strong, opposing views regarding how the EU ETS interacts—or should interact—with electricity markets. Pricing in electricity markets is further complicated by the special character of these markets, particularly their network externalities, and by their role as players with considerable market power. Furthermore, many of the large European energy companies are publicly owned, which increases the possibility that their pricing strategies may deviate from pure profit maximisation. In sensitive issues, such as this one, public relations and the interests of the utility owners are likely to be particularly important.

Studies on price effects of emissions trading include the econometric time series analyses of Bunn and Fezzi (2007) in the U.K. electricity market and Fell (2008) in the Nordic market. Sijm et al. (2006; 2008) have performed simulations of a number of European markets, as well as some econometric analyses. These studies all have similar findings: at least in relatively competitive markets, 60–100% of the CO_2 price is passed through to electricity consumers, more likely at a higher amount the more competitive the market is. This suggests that consumers pay for a significant portion of the value of emissions allowances.

However, econometric studies do not remove the ambiguity about how much pass through is actually observed, which is important for concerns about fairness, compensation, and potential windfall profits.[55] The ambiguity stems from the heterogeneity of technologies within the market. In the electricity sector, technologies differ both with respect to the marginal cost of producing electricity and to their emissions rate. In most wholesale power markets (except where power purchase agreements are in place), each electricity producer receives the same price per unit produced. The price depends solely on the bid of the last generator, i.e., the one with the highest marginal cost needed to meet demand. Figure 2 gives a schematic illustration of pass-through effects in electricity, showing that at peak demand the pass through will be lower

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[51] See "Member states look to deal with windfall profits," *Carbon Market Europe, Point Carbon*, October 14, 2005, 6.

[52] RWE is one of the largest electricity producers in Germany and has received a major share of their allowances for free, according to a grandfathering procedure.

[53] See Bundeskartellamt press release, December 20, 2006 (http://www.bundeskartellamt.de/wEnglisch/News/Archiv/ArchivNews2006/2006_12_20.php [accessed June 2009]).

[54] The Belgian energy market regulator CREG stated that the country's authorities must prevent utilities from making windfall profits by passing on the cost of CO_2 emission allowances to consumers (Reuters, January 21, 2009). CREG referred to estimates that electricity producers in the Belgian market made about €1.2 billion (US$ 1.68 billion) in profits between 2005 and 2007 by charging clients for CO_2 emissions allowances they had received for free.

[55] Wråke et al. (2008) apply an experimental approach, with experiment subjects acting as firms in a controlled economic environment in a laboratory, to analyse the issue of pass-through behaviour.

than at off peak. The reason is that natural gas, which most often is at the margin in European electricity generation, at peak demand has lower carbon intensity than coal. There are also dynamic effects that are harder to represent in a simple figure. For example, because demand for allowances will be higher at peak demand for electricity, it acts as a driver for higher allowances prices. However, even though electricity companies play an important role in the EU ETS, non-perfect correlation in demand patterns across the EU and the large volume of allowances in the market will make it unlikely that the effect on allowance prices, due to electricity demand spikes, would be a dominant factor.

Figure 2   Pass Through of Allowance Cost in Electricity
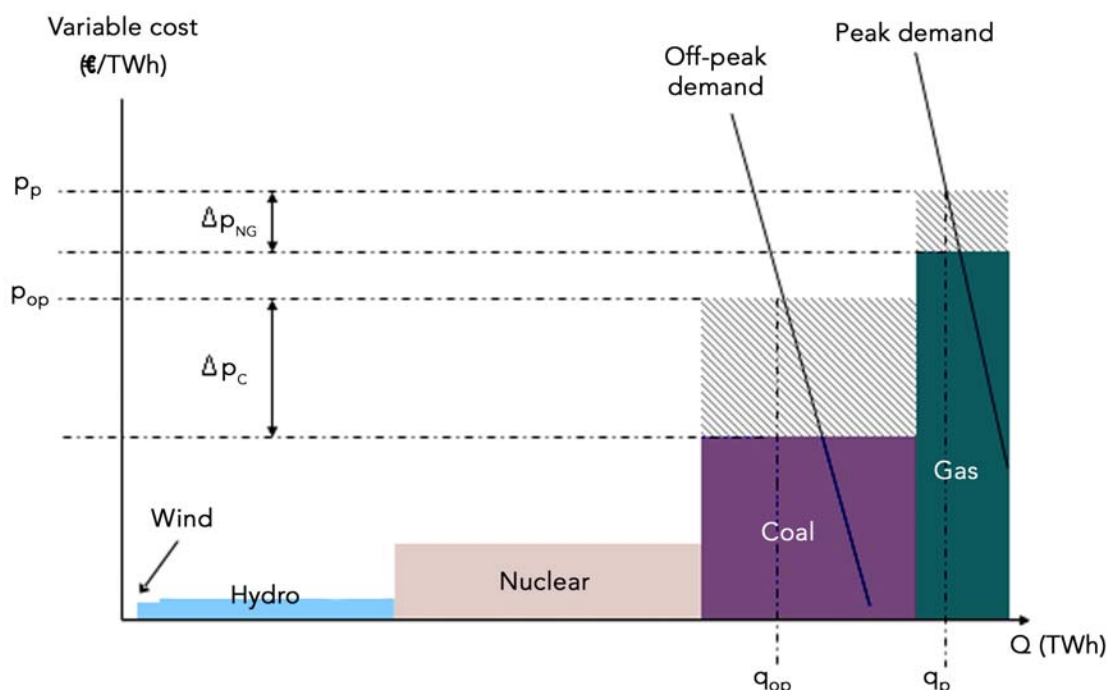
The figure shows a schematic merit-order curve for electricity. At peak demand, the clearing price would be p_p, including the cost of emission allowances Δp_{NG} required for the marginal unit which is natural gas. Thus at peak demand an operator of a coal-fired plant will not recuperate 100% of the additional variable cost of allowances. At off peak the clearing price p_{op} is lower, but the pass through of allowance cost Δp_C is higher.

Sijm et al. (2006) refer to the pass-through behaviour of the individual generator as 'add-on' and the increase of the bid for the marginal unit, which will determine the electricity price, as 'work-on'. The work-on rate can be calculated from market observations, but the add-on cannot be calculated without knowing which kind of technology is on the margin at any given point in time. A marginal pass-through rate (work-on rate) of 100% does not indicate whether the industry is earning revenue that is more or less than 100% of its cost[56]. The cost to industry

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[56] Indeed, the "60-100% pass through rate mentioned earlier refers to average effect under specific assumptions of marginal capacity and does not say much about the work on.

depends on the emission rates of infra-marginal generators, which may on average be more than or less than the emissions rate of the marginal unit.

Uncertainty and Price Volatility

There is extensive economic literature on what type of policy is most efficient when there is uncertainty about both benefits and costs of regulation.[57] Much of this literature revolves around the relative merits of cap and trade versus emissions taxes. Here, we focus on the EU ETS and do not deal with the choice of trading versus taxes, although much can be learnt from the more general discussion on policy design under uncertainty.

There is an inherent trade-off between flexibility and certainty in any policy design. On the one hand, there are benefits of retaining the options of adjusting policies to changing priorities and information, for instance, new developments in climate science and in the international climate policy negotiations. On the other hand, there is a need to provide certainty to market actors. Uncertainty over prices in products or inputs will, on average, delay investments, compared to a situation under certainty.[58] The greater the level of policy uncertainty, other things held equal, the less effective the climate change policies will be at providing incentives for investment in low-emitting technologies. The closer in time a change in policy is expected, the higher the option value will be for a company. This is particularly relevant in capital-intensive sectors where investment cycles may stretch over several decades.

Critics of the EU ETS have argued that the market for emission allowances is artificial, in the sense that there is no underlying physical commodity being traded. Fluctuations are closely linked to variations in policy, which are difficult to predict. In addition, price volatility in inputs can be amplified by the allowance market.[59] A potential consequence could be that the trading system, instead of offering incentives for long-term investments based on expectations of higher future prices of carbon emissions, only stimulates changes based on short-term marginal costs. This would make it less effective in driving investment in low-carbon technologies and in research and development of new technologies.

Real option models are frequently used to assess the effect of uncertainty on firm behaviour, and many studies indicate that uncertainty will reduce the efficiency of climate policy.[60] For example, IEA (2007) shows that the price of carbon required to make an investment in a hypothetical carbon capture and storage (CCS) facility viable is 37% higher if policy is set only 5 years into the future, compared to policy that is certain 15 years ahead. This is particularly relevant since 5–15 years is a typical time over which a firm needs to recoup the majority of major capital investments—and it also resembles the current cycles in international climate negotiations. Fuss et al. (2009) conclude that scenarios with small and frequent revisions of an emissions cap will result in higher cumulative emissions than if policies are altered less frequently but more drastically. Laurikka and Koljonen (2006), using an extended discounted cash flow model, show that uncertainty regarding the allocation of emission allowances was

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[57] For example, see Weitzman (1974), Roberts and Spence (1976), Kolstad (1996), Pizer 2002, Hoel and Karp (2002), Montero (2002), and Mandell (2008). The key point made by Weitzman and elaborated by others is that the expected efficiency of the policies will depend on the relative slopes of the curves for marginal costs and marginal benefits of emissions reductions, as well as the associated uncertainties in these curves.

[58] See, for example, IEA (2007), Philibert (2006), and Laurikka (2006). Simply put, the option value is 'the value of waiting'.

[59] See, for instance, Bunn and Fezzi (2007) and Fell (2008) for analyses of price interactions between electricity prices and the EU ETS.

[60]The models build on the option value theory (see, for instance, McDonald and Siegel 1986; and Dixit and Pindyck 1994), which predicts that firms will consider shifting investments in time in order to gain better information. If the (future) information is worth more than owning the asset, investment will be postponed. Real option modelling can be applied both in short-term frameworks focused on operational decisions and in long-horizon valuations of investments over decades or even centuries.

critical in a quantitative investment appraisal of fossil fuel-fired plants in Finland. These findings are consistent with the rich literature that exists for other environmental areas. A multitude of studies show that the hurdle rates for investments may be multiplied several times when uncertainty is considered compared to evaluations using standard net present value calculations.[61]

Assessments of climate policy-induced uncertainty should, however, be made in light of other market factors, which affect investments. For most firms, the carbon price would have to be significantly higher than today to have the same impact on investments and cost variability as, for instance, variations in fuel prices, demand for energy and commodities, currency fluctuations, and political turmoil. IEA (2007) concludes that it is unlikely that climate policy uncertainty would pose a serious threat to overall capacity levels in most electricity markets in the long run. If climate policy is set over sufficiently long time scales, the total risk will be dominated by fuel price risk, with climate policy contributing relatively little to the total risk profile of the investments.[62]

Nevertheless, several proposals have been put forward that address both short-term and long-term aspects of price variability and regulatory uncertainty associated with the EU ETS. Some proposals are aimed primarily at reducing overall costs and, thus, lower the risk of very high prices of emissions. An example is strategic public investments intended to reduce the cost impact of emissions trading by lowering marginal abatement costs. Other measures target cost volatility more directly, while still others attempt to address both aspects in parallel. For example, offset mechanisms (such as the CDM) seek to reduce the overall cost of reaching an emissions target and create a backstop price of emissions by making low-cost abatement opportunities available outside the trading system. Short-term cost variability could also be reduced by expanding the role of the offset market in response to sudden price increases in the domestic market for emissions.[63]

A relatively simple measure to avoid drastic market corrections of the allowance price would be to improve transparency in monitoring and frequency of emissions reporting. Had firms' emissions been published more frequently, dramatic price falls, such as EU ETS experienced in 2006, could probably be prevented. Increasing the length of the trading periods would reduce regulatory uncertainty,[64] but would also limit political manoeuvring room of the EU. Allowing firms to bank allowances, which was not permitted between phase I and phase II, has substantial economic benefits. It increases the inter-temporal flexibility of firms, allowing them to implement low-cost abatement options in one case and postpone higher cost measures in another, so as to minimise net present cost of investments. Furthermore, banking gives firms with a surplus of allowances a vested interest in keeping the allowance market active. Borrowing improves firms' opportunities to rationalise investments over time, similar to banking. However, it also introduces an element of moral hazard; firms that acquire an emissions debt have an incentive to work for a relaxation of the emissions cap or even a suspension of the trading system in order to wipe out that debt.

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[61] See, for example, Löfgren et al. (2008) for an overview of existing studies.

[62] Some studies, like Zhao (2003), even suggest that uncertainty in allowance markets may help maintain firms' investment incentives, compared with a scenario with fixed emissions charges. Firms factor in a convenience yield for the value of holding allowances over and above the marginal cost of avoided abatement expenditures. This premium leads to additional investment in order to hedge against the uncertainty in the allowance market.

[63] Although, some observers have serious doubts about both the CDM's potential ability to reduce overall compliance costs—the primary reason being the difficulty of ensuring that reductions are additional—and its ability to act as an effective cost-containment mechanism because of the constraints in delivering large volumes of reductions quickly. See, for instance, Wara and Victor (2008) for a discussion.

[64] As discussed by Åhman et al. (2007) and Sterner and Muller (2008), this would also reduce some of the distortions created by the allocation procedures.

One of the most debated cost management proposals is the so called 'safety valve', a guard against unexpectedly high allowance prices. The EU has been firmly opposed to such a mechanism and it is not part of the EU ETS. However, it features prominently in the U.S. discourse on cap and trade,[65] so—in part because of its implications for a future linking of the EU ETS and a US system—considerable attention has been given to the safety valve in the EU as well.

The basic idea of a safety valve is that additional allowances would be released into the market if prices exceed some pre-determined ceiling. If the number of allowances that can be released could is unlimited, the cap is in effect relaxed. If, instead, the allowance pool is limited, the short-term problem of price spikes may be mitigated, but not the long-term problems of escalating prices and costs to stay under a firm cap on emissions.[66]

The safety valve has been criticised from several perspectives. At a general level, such a mechanism may be promoted by parties who want it introduced solely as a way to constrain costs and emissions reductions below what they otherwise would be, given the set emissions reduction targets. In such a circumstance, the safety valve would lower the environmental integrity of the system. More importantly, evidence is that ex post actual costs of government regulation are more often lower than ex ante expected costs (Harrington et al. 2000). In an emissions trading system, this results in falling prices. To date, the problem in emissions trading systems has not been unforeseen price rallies, but rather much lower prices than expected.[67] For instance, in the US SO_2 trading program, market prices fell quickly to one-quarter of the price projections predicted by the US EPA before they launched the system. The EU ETS' prices collapsed in 2006, and even though phase II prices are higher, they are still lower than what many observers projected them to be. Lower-than-expected costs are, of course, not a problem if the cap is the optimal one. However, often the intention is to gradually tighten the cap, striking a balance between increasing stringency and limiting costs. This is the case with climate change, where both the EU and elsewhere expect to reduce emitted volumes significantly in the future. In this situation, a safeguard not only against higher-than-expected costs but also against lower-than-intended prices may be called for.

Figure 3 illustrates the effect that cost management mechanisms will have on expected allowances prices. As expectations of allowance price levels shift, so will the expected returns on investments of various types. Consider the profit function for a firm that uses a non-emitting technology to produce an output q:

$$\pi = qp_q(Q, p_A) - c(q) , \qquad\qquad\qquad (1)$$

where p_q is the market price for the output q, Q is the aggregate quantity in the market, p_A is the price of allowances, and c is the cost function of the firm. The firm maximises profits by choosing

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[65] Some kind of safety valve has been included in an overwhelming majority of the proposals for a federal U.S. trading system that have been put before Congress to date.

[66] See Pizer (2002), Kopp et al. (2002), Murray et al. (2008).

[67] One exception is the RECLAIM program for NOx emissions in California. In 2001, in hot and dry conditions, a sudden increase in demand for NOx credits was created because coal-fired utilities had to compensate for a lack of hydropower. As a result, NOx credits jumped from less than US$ 1 per pound to over $30 in six months. Due to this, the program was re-examined and amended, resulting in major changes. The first major change was the mandated installation of emissions control technology in major power plants to reduce emissions. The second was legislation to have businesses that produced over 50 tons of NOx annually develop five-year emissions plans based on historic emissions levels from the year 2000, thus improving market transparency. RECLAIM's price levels have since dropped back to prices reflecting those seen before the price spikes.

quantity. Assuming competitive markets $\frac{\partial p_q}{\partial q} = 0$ and the firm maximises profits, such that marginal revenues equal marginal costs:

$$p_q(Q, p_A) = \frac{\partial c}{\partial q} \quad . \tag{2}$$

As aggregate quantity and allowance price are uncertain variables, thus making market price uncertain, the firm's profit maximising would equal marginal cost to *expected* marginal revenues:

$$E(p_q) = \frac{\partial c}{\partial q} \quad . \tag{3}$$

With a safety valve, prices over a given level are precluded, resulting in a lower expected market price $E(p_q^{SV})$ than without the safety valve. Thus, our non-emitting firm would choose a level of output under the safety valve such that:

$$E(p_q^{SV}) = \frac{\partial c}{\partial q^{SV}} < \frac{\partial c}{\partial q} = E(p_q) \quad , \tag{4}$$

resulting in $q^{SV} < q$. Thus, the consequence of the safety valve is a reduction in investment in low-emitting facilities.

Burtraw and Palmer (2006) and Palmer et al. (2008) show how a political commitment to prices above a certain level would work in the opposite direction and analyse how such a mechanism would affect investment incentives. Because a guaranteed minimum price precludes prices below a certain level, the expected price will be increased, compared to a situation without such a price floor. If a safety valve and a price floor are combined and made symmetric, it would bring the expected prices levels back to conditions without the cost management mechanisms, so that $E(p_q^{SV+PF}) = E(p_q)$, resulting in maintained investments and output $q^{SV+PV} = q$.

Figure 3   Distribution of Allowance Prices under a Cost Management Mechanism

Panel A
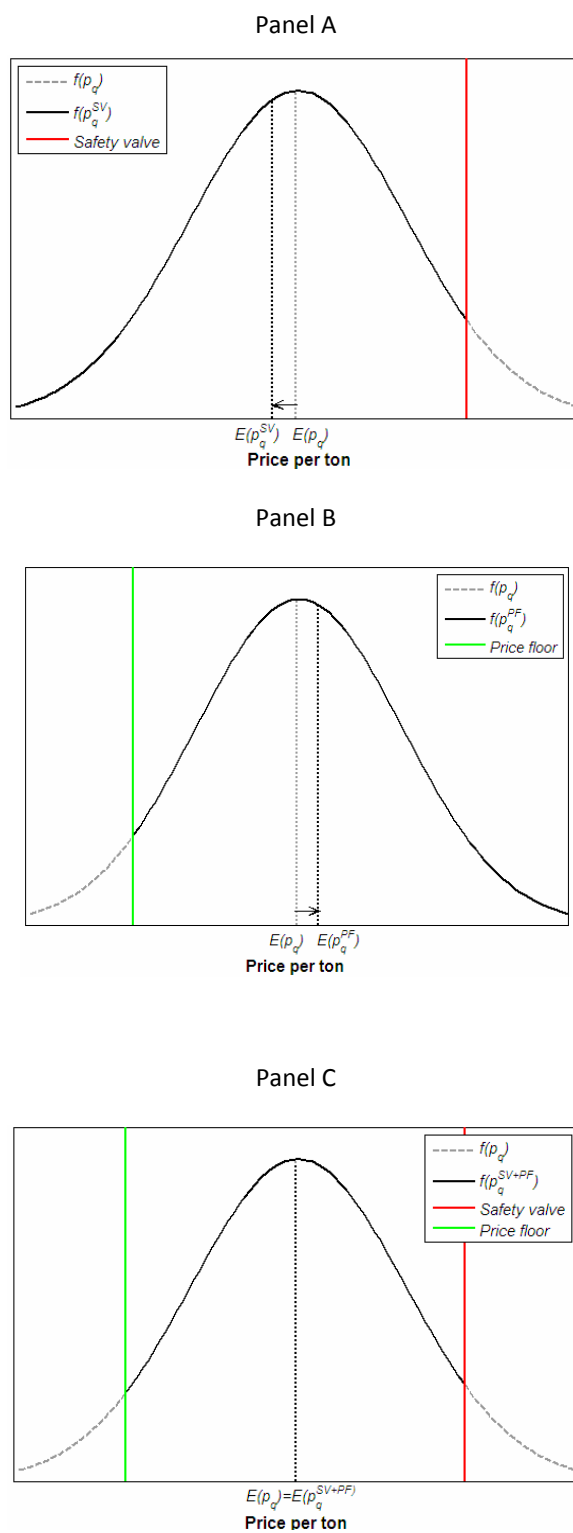
Panel B

Panel C

Panel A shows a safety valve, panel B shows a price floor, and panel C, the balancing effect of a symmetric cost management mechanism on the expected value of the allowance price.

In the limit, if the level of the safety valve is lowered and the guaranteed minimum price is increased so that the two coincide, the trading system has, in effect, turned into an emissions tax. The original uncertainties in prices and abatement costs are gone, replaced by uncertainties in emissions and damages to the environment. Of course, this brings us back to the discussion on the relative merits of price versus-quantity type regulations.

To summarise, the level and nature of uncertainty are key factors in the design of climate policy and important determinants for the efficiency of the policy. Expectations of high-compliance costs and the interaction of allowance markets with natural price variations are recurring arguments against stringent policies. At the same time, it is imperative to have credible investment incentives that are high enough to bring about the changes needed. If investors perceive climate policy measures as short-sighted and volatile, pursuing traditional high-emitting technologies will be a less risky strategy than investment in new and, in some cases, unproven technologies. Efficient mechanisms to manage uncertainty in incentive structures, overall costs, investor expectations and short-term price fluctuation would strengthen both the political case for climate policy and the efficiency of such policies.

The EU ETS and the Transport Sector

The potential inclusion of the transport sector in the EU ETS has been a frequent topic of discussion. The most obvious reason for including it is that the transport sector alone accounts for more than 20% of EU greenhouse gas emissions (EEA 2009); and some argue that including transport in the EU ETS would be an effective way to curb emissions by the sector.

Fuel taxes in the road transport sector vary in levels and structure across member states. Often they have several components with different names: energy tax, petrol tax, CO_2 tax, etc. Overall, fuel taxes in the EU are high, compared internationally, and far above the current price of CO_2 in the EU ETS. Empirical evidence strongly suggests that these fuel taxes have had a large impact on emissions from road transport across the EU, indicating that the long-term elasticity in fuel demand is significant.[68] However, because demand is more inelastic in the short run, the transport sector (were it in the EU ETS) would most likely be a large buyer of allowances, particularly if inclusion in the EU ETS replaced a large proportion of fuel taxes. This would result in significantly higher allowance and electricity prices for industry, and a radically lower pressure on the transport sector to reduce emissions.[69] Dynamic and secondary effects are difficult to predict, but it is clear that including road transport in the EU ETS is associated with substantial uncertainty and risk regarding both emissions from transport and economic impact on other sectors.

In contrast to road transport, international aviation and maritime shipping are virtually exempt from all climate policies, even though emissions from these sectors are growing rapidly. The European Environment Agency (EEA 2006) forecasts that emissions from EU international aviation will grow by 150% in the period 1990–2012, which alone would offset more than one-quarter of the Community's reductions target under the Kyoto Protocol. Progress on regulating these emissions have been virtually absent in international negotiations, and neither of them are covered by the Kyoto Protocol. In an effort to break the trend of growing emissions, the EU has decided that all flights taking off or landing in the EU will be included in the EU ETS, starting in 2012. This decision has been contested by the International Civil Aviation Organisation (ICAO) on the grounds that it breaches international agreements.[70] Whether ICAO's claim will hold up in

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[68] Sterner (2007).

[69] See Holmgren et al. (2007) for an illustrative discussion.

[70] Current legislation on international aviation is based on the concepts in the Convention on International Civil Aviation (also known as the Chicago Convention), that came into force in 1947. The preamble to the convention states

a judicial review remains to be seen, but a ruling for ICAO could severely limit countries' ability to introduce domestic environmental legislation on international aviation.

The International Maritime Organisation (IMO) has been charged with leading the international efforts to curb maritime emissions. As with international aviation, little progress has been made. Although no formal proposal has been put forward to include shipping in the EU ETS, the EU has stated that if no international agreement is reached under the leadership of the IMO or the UNFCCC by the end of 2011, it will propose including maritime shipping in the EU ETS, as of 2013. A longstanding argument against regulating aviation and maritime transports is that it would be technically and legally difficult to do so. This argument has been discounted by recent research and the absence of international policies and measures seems due to institutional issues and political barriers rather than to technical shortcomings.[71]

Competitiveness and Carbon Leakage

Modern history is full of examples where proposals for environmental legislation have been accompanied by intense debate over their effects on industry, and the EU ETS is no exception. Earlier sections of this paper touch upon how the trading system has affected competition between technologies and new versus existing installations, and how member states may engage in strategic behaviour in order to attract investments. This section deals primarily with the competitiveness of European industry vis-à-vis the outside world.

The literature on the impact of environmental policy on trade flows is extensive. A number of competing theories have been put forward to explain how firms respond to tightening environmental regulation. Copeland and Taylor (1994) helped sort out the conflicting evidence and arguments. They made the distinction between the 'pollution haven effect', which implies that, all else being equal, tightening environmental regulation will drive firms to countries where it is more lenient, and the 'pollution haven hypothesis', which says that this effect is a dominant force for firm location.

Another theory, with roots in the business literature, is the 'Porter hypothesis',[72] which argues that stringent environmental policy will prompt productivity and efficiency improvement of firms to such an extent that the net costs to firms will be negative and their competitiveness enhanced.

The term 'competitiveness', often used in relation to effects of the EU ETS, should be interpreted with care. At a microeconomic level, the definition is relatively straightforward, at least in the short term. For example, pollution regulation that affects a firm's costs of production will also alter its competitiveness. A uniform cost increase of emitting carbon, as imposed by the EU ETS, will impact all firms that emit, but the importance of these costs to each firm will differ greatly, depending on the carbon intensity of its production. In the short run, a firm's competitiveness will be negatively affected if it faces a higher cost for polluting than its competitors or if it has higher carbon intensity than its competitors.

On a macroeconomic scale, the term 'international (or national) competitiveness' usually refers to the ability of firms to sell their goods and services on the international market. A potential measure of this is net exports; if they are high, a country or a region has a high international competitiveness. However, in the long run, such differences will be factored into currency exchange rates and labour cost, eventually balancing out gains in competitiveness

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that civil aviation should be established on the basis of 'equal opportunity', a priority that still dominates talks on regulation of international aviation.

[71] See Åhman (2008) for a discussion.

[72] The hypothesis is named after its proponent, Michael Porter. See Porter and van Linde (1995).

defined in this way. Consequently, other measures have been put forward[73] that may be more accurate, but the bottom line is that competitiveness will have different meanings to different stakeholders, at different levels of the economy, on different time scales, and in different contexts. Consider the simple example of a European firm investing in new capacity in China. A European policy maker may interpret this as a sign of decreasing competitiveness of the European economy. For the firm, however, the same investment may result in lower costs, improved competitiveness, and increased market share.

One concern voiced by European industry in particular is the issue of *carbon leakage*. It is important to distinguish between intended and unintended effects that follow from a cap on carbon emissions. From a regulator's perspective, it may not be a problem if output in one sector is reduced in favour of another, or if existing (dirtier) goods are replaced by new (and cleaner) goods. However, if industry activities (emissions production) are simply shifted outside of the EU, it would make the emissions trading system less effective and raise the overall cost of reaching the environmental objective, and could result in reduced employment at home.

The term carbon leakage, itself, has an ambiguous definition. Usually it is defined as the ratio $\Delta E_1 \big/ \Delta E_2$, where ΔE_1 is the emissions increase in countries outside the policy regime, and ΔE_2 is the reduction in emissions in the region under the policy. (Changes in emissions are driven by the policy.) However, this implies that if an inefficient installation is closed due to the EU ETS and its market share is captured by a more efficient plant in another region, it is also defined as leakage, even though total emissions have decreased.

Firm relocation is probably the driver of leakage most commonly referred to in the public debate. The basic argument is simple: given the asymmetries in carbon prices between Europe and the rest of the world, it is rational for European firms, all else being equal, to look for opportunities to shift their activities elsewhere. Empirical evidence suggests that the cost of complying with environmental regulation is generally a small share of a firm's total cost structure. Other factors, such as the cost of capital, trained personnel, etc., also affect a firm's location choice. However, on the margin, it would be rational for firms to relocate production in response to environmental stringency. This is particularly important, given the ambitious emission targets being set by the EU in relation to some of its trading partners. A more subtle version of this is altered patterns of reinvestments; even if firms keep existing capital stock in place, they may prioritise expansions and reinvestments in other places.

Loss of market share to firms outside of the EU is another channel for leakage. In the short run, firms facing higher variable costs will have to raise prices, resulting in declining sales, or reduce their prices, thus eroding their profit margin.[74] In the long run, if cost asymmetries persist, it will affect the use of existing capacity and the returns on investment.

There is also a general equilibrium effect that has the potential to generate carbon leakage. The large-scale reduction in demand for carbon-intensive commodities, such as fossil fuels, in the EU would prompt global prices on those goods to fall. As the prices of these goods fall, other parts of the world economy with less stringent climate policies would increase their consumption of these cheaper goods, thus offsetting some of the European reductions.

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[73] See Brännlund (2008) for an overview.

[74] In the very near term, one could argue that firms do not even have the choice of lowering prices or adjusting production, as does Morgenstern et al. (2007).

Most empirical studies[75] of the EU ETS have focused on identifying what sectors are at risk for leakage. Two factors have received particular attention: exposure to international trade and what values associated with the EU ETS are at stake. High exposure to international trade can reduce a firm's ability to pass on the cost of carbon to its customers. If, in addition, the carbon costs are high, relative to the value added of the firm, there is a greater risk of leakage. Using only these two determinants for analysing the risk of carbon leakage will certainly offer an incomplete picture, but it will likely give at least a rough indication of which sectors are most vulnerable.[76]

In studies of the value at stake, defined as the ratio between the added cost of carbon and the value added by the firm, the cement industry generally rank among the highest. Hourcade et al. (2007) estimates increased costs,[77] due to carbon pricing, of more than 30% of value added in the UK cement industry. Graichen et al. (2008) puts the equivalent number for the German cement industry at over 60%.[78] In Sweden and the Netherlands, the values at stake are generally found to be significantly lower across all sectors, but the cement industry comes out high in those countries as well: 9% in Sweden (Zetterberg and Holmgren 2009) and 8% in the Netherlands (de Bruyn et al. 2008).

However, if trade intensities are considered, the picture changes considerably. For instance, the German cement industry's trade intensity with non-EU countries was approximately 3% in 2005,[79] and only the lime industry ranked lower. Similar results have been reported for the UK. At a global level, only 6% of cement is traded internationally (Reinaud 2005a). In fact, due to its relative insulation from international competition, the cement industry is likely less affected, and domestic substitution is a more relevant threat to the sector than international trade (de Bruyn et al. 2008). Instead, it is the aluminium, iron and steel, and fertilizer industries that are consistently most vulnerable to increasing costs of carbon. The chief reason is the high proportion of international trade in these sectors; in 2007, 40% of global steel production[80] and 77% of global aluminium production[81] was traded internationally.[82]

The next step, quantifying how much carbon leakage will result from the EU ETS, is even more complex. Evidence of carbon leakage includes changes in trade and investments. The multitude of forces driving such activities makes it difficult to establish a counterfactual scenario needed to estimate the effect of climate policy or control for all relevant variables in a statistical model. An accurate analysis would require knowledge of how European and foreign firms respond to fluctuations in carbon prices, what technologies and associated emissions intensities dominate in different regions, cross-elasticities between substituting goods and international trade flows, to name just a few parameters. Furthermore, capital-intensive sectors, such as those identified as most at risk, are typically characterised by a high inertia due to long investment cycles and significant fixed costs. This further highlights the need for detailed and disaggregated data.

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[75] For example, see Demailly and Quirion (2007), Reinaud (2005a; 2005b; 2008), Smale et al. (2006), McKinsey (2006), Gilbert, Bode, and Phylipsen (2006), Hourcade et al. (2007), Graichen et al. (2008), de Bruyn et al. (2008), and Zetterberg and Holmgren (2009).

[76] These two parameters have also been identified by the EU Commissions as most important in determining which sectors could eligible for free allocation in the EU ETS phase III. See separate section below.

[77] This includes both direct costs for abatement or purchase of allowances, which would depend on the method of allocation (the numbers cited here pertain to 100% auctioning) and indirect costs, resulting from increased electricity prices.

[78] Both studies assume a €20-EUA price. Hourcade et al. (2007) uses 2004 industry data, and Graichen et al. (2008) uses 2005 production data.

[79] German Statistics Office, quoted in Graichen et al. (2008).

[80] Reinaud (2008a).

[81] Baron et al. (2007).

[82] For a more detailed breakdown of trade flows in exposed sectors, see Mohr et al. (2009).

Hence, it is not surprising that ex-ante studies[83] of both the EU ETS and other planned or potential climate policies display a wide range of results. These theoretical studies are often based on general or partial equilibrium models, which may not be deliberately designed for carbon leakage estimates. Demailly and Quirion (2008a) estimate the leakage rate in the iron and steel industry to be 0.5–25 %, with a median value of 6%. Some findings are contradictory in light of analysis of sectors vulnerable to leakage. Ponssard and Walker (2008) estimate leakage rates in the EU cement sector to be around 70% with EUA prices at €20, while Demailly and Quirion (2008b) report 20% leakage rates for the cement sector, assuming a €15 carbon tax in annex B countries (except for the United States, Australia, and New Zealand). Ex post studies based on empirical observations are still scarce, but the results are much more consistent: they show little, if any, evidence of carbon leakage resulting from EU ETS.[84] Naturally, not enough time has elapsed since the EU ETS was implemented for any robust time series of these effects, so any findings should be interpreted with care.[85]

There are some important caveats, however. Modelling studies on sectoral competitiveness is usually based on the assumption that sector characteristics are homogenous. Factors, such as like abatement costs and technologies used, are modelled as identical across firms. At sub-sector or firm levels, this is unlikely to be true and leakage effects may well be different. Further, most studies look at short-term effects and use carbon prices in the range of 10–50€/ton CO_2.

In the long run, conditions can change considerably, which brings us back to a distinction between the pollution haven *effect*, and the pollution haven *hypothesis.* The empirical studies quoted here have not been able to confirm the pollution haven hypothesis. This indicates that any pollution haven effect has not been a dominant force for firm location and trade flows. Should carbon prices increase dramatically, however, their importance will increase and they could potentially become a major factor. Further, many firms have long-term contracts for electricity that have insulated them from increasing carbon costs so far. As these contracts expire, effects of the EU ETS will become more visible. In sum, there are good reasons to revisit the issue of leakage, both empirically and theoretically, over the coming years.

3. The Road Ahead: Conclusions and Unresolved Issues

The first years of the EU ETS have demonstrated that it is possible to design and implement a large-scale trading system in a relatively short period of time. Phases I and II have provided opportunities for institutional learning, development of market infrastructure, and empirical assessments, which will be critical to future improvements of the system.[86] Clearly, considerations of political feasibility, special interests, and perceived fairness have been key parameters in the design of the EU ETS, and they will no doubt continue to be so in the future. . A simpler trading system with few distorting elements would be more economically efficient, but pose greater political challenges to implement, in part because it would leave less room for pursuing other policy objectives than least cost emissions reductions, such as stimulating certain technologies, developing new fuels, or including additional industries.

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[83] See Gielen and Moriguchi (2002), OECD (2003), Demailly and Quirion (2006; 2008a; 2008b), and Ponssard and Walker (2008).

[84] Looking for effects on trade flows, Lacombe (2008) finds no significant changes in petroleum products, Reinaud (2008b) reports no significant effects in aluminium trade, and Demailly and Quirion (2008a) find no changes in trade flows in Iron and steel,

[85] I have found no studies that report leakage rates exceeding 100%. Thus, neither the theoretical nor the empirical literature supports suggestions that a cap on European emissions would result in *increased* global emissions.

[86] In fact, the first phase was originally a trial period. In addition to the large body of academic research and consulting reports that has been published on the EU ETS, the EU ETS Directive mandated the EU Commission to carry out a review of the trading system, which it did in 2006 and 2007. Documents from this review can be found at http://ec.europa.eu/environment/climat/emission/review_en.htm (accessed June 2009).

In response to some of the criticism of its current design, the EU is reviewing the EU ETS. In January 2008, the EU Commission published "The EU Energy Package", an extensive proposal for a new, integrated climate and energy policy for Europe.[87] After less than a year of negotiations (which in this context has to be considered remarkably short), the Parliament and the Council struck an agreement in December 2008, and the Energy Package was formally adopted on April 6, 2009. It contains three principal elements:[88]

1) Legally binding greenhouse gas emissions-reduction targets for sectors not covered by the EU ETS (These targets—the so-called 'effort-sharing agreement'—range from -20% to +20%, and in total amount to 10% reduction below 2005 levels.)

2) Differentiated targets for renewable energy sources and a flat rate of 10% biofuels

3) A revision of the EU ETS, including a 21% emission reduction target for the sectors covered compared to 2005

These targets together aim to achieve an overall 20% reduction in total greenhouse gas emissions below 1990 levels by 2020 and a 20% share of renewable sources in final energy consumption.

For the EU ETS, this means that substantial changes will come, as of January 1, 2013. The most fundamental change is centralising much of the allocation process. Instead of each member state drawing up a NAP, the cap will be set at the European level. This change will reduce the risk of a repeat 'race to the bottom', seen in the first two allocation rounds. The cap in 2013 will start at the average total quantity of allowances allocated by member states in 2008–2012,[89] decreasing linearly to a 21%-reduction below 2005 levels by 2020. It is worth noting that the annual reduction rate[90] is legally binding beyond 2020, unless a new decision is made. The EU has, in fact, laid out a default emissions reduction path, not only for the short term, but also further into the future. Should an international agreement be signed that triggers an EU move to an overall 30% reduction target, the cap of the ETS will be adjusted downward proportionally.

The reductions imposed on the traded sectors are larger than what an equal burden between sectors (in terms of absolute emission reductions to reach the 20% overall target) would imply. The underlying rationale is that the EU expects the traded sectors to have lower abatement costs, compared to the non traded sectors. The approach used in phase I and II was rather the opposite, where traded sectors received a relatively generous cap, compared to sectors outside the system. This is further evidence of the political pragmatism that influenced central elements of phase I and phase II, with the EU seeking buy-in of the system from major industry stakeholders through a generous allocation of allowances.

Another central change is that auctions will distribute approximately 50% of the allocations in the revised EU ETS, up from about 4% in phase II. Electricity producers will, by and large, receive no free allocation.[91] In other sectors, 20% of allowances will be auctioned in 2013,

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[87] The "Energy Package" builds on the Commission proposal of January 2007, *Energy Policy for Europe*, and its twin communication, *Limiting Global Climate Change to 2°C*. Both of these proposals were endorsed by the spring Council in 2007.

[88] The package contained a number of additional documents, including a directive on the geological storage of CO_2 (an 'enabling document' focusing on legal matters), new state aid guidelines, and Commission communications and impact assessments of the proposals. Also, see http://ec.europa.eu/commission_barroso/president/focus/energy-package-2008/index_en.htm#key, for important documents (accessed May 2009).

[89] The cap will be adjusted for changes in the coverage in the system. In 2012, the aviation sector will be included and in 2013 aluminium production and parts of the chemical industry will also be covered. Further, nitrous oxide from fertilizer production and perfluorocarbon emissions from aluminium production will also be included.

[90] Equalling 1.74% per year in the traded sectors.

[91] Certain member states are allowed an optional and temporary exemption from the rule that no allowances are to be allocated free of charge to electricity generators, as of 2013. This option is available to member states which fulfil certain conditions related to the interconnectivity of their electricity grid, the share of a single fossil fuel used in

increasing to 70% in 2020, 'with a view to reaching 100% in 2027'.[92] The broader use of auctions in phase III is likely to improve the economic efficiency of the EU ETS. The specifics of how the auctions will be structured and implemented are still to be settled, however, and there are potential pitfalls which could undermine some of the positive effects. Making sure that auctions are not used for national interests,[93] reducing the risk of collusion among firms, and minimising administrative costs should be priorities. A reserve price in the auctions would act as a price floor in the market and increase incentives for investments in low-carbon technologies. How the revenues are used will also impact efficiency, as will the way costs, imposed on the economy by the EU ETS, are distributed among member states, industries, and households.[94]

There is an important exception to phasing out free allocation. Installations that 'are found to be exposed to a significant risk of carbon leakage' would receive 100% of their allocated allowances for free.[95] The Directive does not specify to which industries this provision will apply. Instead, the EU Commission will assess the risk of carbon leakage, based on direct and indirect cost increases, in relation to the gross value added for the sector, and on the trade exposure for the sector. Although quantitative criteria[96] have been established for the assessment, it will be sensitive to underlying assumptions. Future prices of allowances, trade flows, technological development, investments in new electricity-generation capacity, and currency exchange rates are all factors that will affect the result of the assessment, to name but a few. By leaving an option open for free allocation and open to discussion, the EU has encouraged lobbying by industry and member-state governments interested in protecting industries that are important to their economies. No matter the outcome of the process, it seems plausible that it will be contested by some dissatisfied stakeholders.

The third trading period will be 8 years instead of 3 (phase I) or 5 (phase II). This will diminish some of the distortions created by the previous allocation methodologies, as discussed above. It will also, however, affect efforts to reduce carbon leakage.

The primary measure proposed by the EU to mitigate carbon leakage is the free allocation of allowances to firms at risk.[97] This is likely to take some of the heat out of this sensitive discussion and silence some of the most vocal opposition to a stringent cap. However, as discussed previously, free allocation does not, in itself, alter the economic incentives that firms face at the margin. Only an expectation that future allocations will be affected by production decisions will increase the incentives for firms to maintain their activities in the EU. That is, there has to be an element of updating in order for free allocation to do more than strengthen the balance sheets of firms, for instance in the form of output-based and updated allocation. The

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electricity production, and GDP per capita in relation to the EU-27 average. In addition, the amount of free allowances that a member state can allocate to power plants is limited to 70% of CO_2 emissions of relevant plants in phase I and declines annually thereafter. Furthermore, free allocation in phase III can only be given to power plants that were operational or under construction no later than the end of 2008.

[92] As stated in the revised EU ETS Directive. The original proposal from the Commission went further, phasing out free allocation completely by 2020.

[93] At the time of this writing, it seems very likely that each member state will holds its own auctions.

[94] The Directive stipulates that a certain percentage of auction revenues be redistributed among member states, with poorer countries getting a slightly larger share. There are no requirements regarding how member states make use of revenues, although the Directive recommends that at least 50% be used to promote climate change-related activities or investments.

[95] I.e., their share in the annually declining total quantity of allowances. The share of these industries' emissions is determined in relation to total EU ETS emissions from 2005 to 2007.

[96] The EU Commission states that assessments will be based on 'inter alia, whether the direct and indirect additional production costs induced by the implementation of the EU ETS Directive, as a proportion of gross value added, exceed 5%, and whether the total value of its exports and imports divided by the total value of its turnover and imports exceeds 10%. Further, if the result for either of these criteria exceeds 30%, the sector would also be considered as having a significant risk of carbon leakage'.

advantage of using such a mechanism should, however, be weighed against the efficiency losses in terms of reduced incentives for conservation it would carry.

The ETS Directive also leaves open the option for border adjustments. For example, the possibility of requiring importers to surrender allowances is explicitly mentioned. There is a large and growing body of research that analyses the efficiency and desirability of such policies from economic, legal, and political science perspectives. The picture that emerges is ambiguous. Using border adjustments in the context of climate change has still not been tried legally, so whether such measures would be compatible with, for instance, the WTO is not clear. The political implications of using border adjustments, even assuming they are legal, are difficult to predict. If they result in less political will to cooperate multilaterally, the measures could prove counterproductive. Analyses of the economic incentives resulting from various kinds of border adjustments require detailed information on firm characteristics, trade sensitivities, substation elasticities between products, etc. Further, as noted by Fischer and Fox (2009), the environmental effectiveness of import adjustments depends on how well they reflect the actual emission intensities of products (sometimes referred to as 'embedded emissions'), while the competitiveness depends on how large the adjustments are for imported goods that may substitute those produced domestically. Finally, import adjustments do nothing to support domestically produced goods that are exported. Export rebates could do this, but that option is not explicitly mentioned in the ETS Directive.

A comprehensive analysis of potential measures to mitigate carbon leakage is beyond the scope of this article, but clearly current EU climate policy expects that major trading partners will implement comparable policies. A situation with significant asymmetries in the price of emissions, and a plethora of policies introduced to adjust for them, is obviously second best. Political signals for more progressive climate policies emerging from countries, such as the United States and China, suggest that measures to mitigate carbon leakage should be transitional rather than long term. Assessments of the efficiency and appropriateness of such measures should be made in this light.

The initial years of the EU ETS have provided a large-scale testing ground for trading a new environmental commodity. The lessons learned are diverse and not all experiences are positive. Further, the future development of the EU ETS is closely tied to the international climate policy regime, and linking the EU ETS to other trading systems could require changes in its design. Nevertheless, invaluable information has been gained from the EU ETS. Policy makers would be wise to make use of it, be they supporters of emissions trading or sceptics of such policies.

References

Åhman, M. 2008. Why international transport need a sectoral approach. In *Beyond Bali*, ed. C. Egenhofer. Brussels: Centre for European Policy Studies.

Åhman, M., D. Burtraw, and K. Palmer. 2008. Interactions between the EU ETS and Nordic Electricity: Setting the scene. Elforsk Report 08:29. Stockholm: Elforsk Market Design.

Åhman, M., Burtraw, D., Kruger, J. & Zetterberg, L., 2007, "A Ten-Year Rule to Guide the Allocation of EU Emission Allowances". *Energy Policy* 35:1718-1730.

Åhman, M., and K. Holmgren. 2006. New entrant allocation in the Nordic energy sectors: Incentives and options in the EU ETS. *Climate Policy* 6: 423–40.

Baron, R., with J. Reinaud, M. Genasci, and C. Philibert. 2007. Sectoral approaches for greenhouse gas mitigation: Exploring issues for heavy industry. IEA Information Paper. Paris: IEA.

Baumol, W.J., and W.E. Oates. 1971. The use of standards and prices for protection of the environment. *Swedish Journal of Economics* 73: 42–54.

Bovenberg, A.L., L.H. Goulder. 2001. Neutralizing the adverse industry impacts of CO_2 abatement policies: What does it cost? In *Behavioral and Distributional Effects of Environmental Policy*, ed. By C. Carraro, and G.E. Metcalf. Chicago: University of Chicago Press.

de Bruyn, S., D. Nelissen, M. Korteland, M. Davidson, J. Faber, and G. van de Vreede. 2008. Impacts on competitivenesss from EU ETS: An analysis of the Dutch industry. Delft, the Netherlands: CE Delft.

Brännlund, R. 2008. Miljöpolitik utan kostnader? En kritisk granskning av Porterhypotesen [Environmental policy at no cost? A critical review of the Porter hypothesis]. Report to the Expert Group on Environmental Studies 2007:2. Stockholm: Swedish Ministry of Finance.

Bunn, D.W., and C. Fezzi. 2007. Interaction of European carbon trading and energy prices. FEEM Working Paper, no. 63.2007. Milan, Italy: Fondazione Eni Enrico Mattei.

Burtraw, D., K. Palmer, R. Bharvikar, and A. Paul. 2001. The effect of allowance allocation on the cost of carbon emission trading. RFF Discussion Paper 01-30. Washington, DC: Resources for the Future.

Burtraw, D., K. Palmer, and D. Kahn. 2006. CO_2 allowance allocation in the regional greenhouse gas initiative and the effect on electricity investors. *Electricity Journal* 19, no. 2 (March 2006): 79–90.

Burtraw, D., R. Sweeney, and M. Walls. 2009. The incidence of U.S. climate policy: Alternative uses of revenues from a cap-and-trade auction. RFF Discussion Paper 09-17. Washington, DC: RFF.

Böhringer, C., and A. Lange. 2005. On the design of optimal grandfathering schemes for emission allowances. *European Economic Review* 49: 2041–2055.

Christiansen, A.C., and J. Wettestad. 2003. The EU as a frontrunner on greenhouse gas emissions trading: How did it happen and will the EU succeed? *Climate Policy* 3(1): 3–18.

Demailly, D., and P. Quirion. 2006. CO_2 abatement, competitiveness, and leakage in the European cement industry under the EU ETS: Grandfathering versus output-based allocation. *Climate Policy* 6: 93–113.

———. 2008a. European emissions trading scheme and competitiveness: A case study on the iron and steel industry. *Energy Economics* 30: 2009–2027.

Dixit, A.K., and R.S. Pindyck. 1994. *Investment under Uncertainty*. Princeton, NJ, USA: Princeton University Press.

Gilbert, A., J.-W. Bode, and D. Phylipsen. 2004. "Analysis of the national allocation plans for the EU emissions trading scheme." Ecofys Report YUES4003. London: Ecofys.

EEA (European Environment Agency). 2006. Annual European Community greenhouse gas inventory, 1990–2007, and inventory report 2009. Technical Report, no. 4/2009. Copenhagen: EEA.

———. 2009. Annual European Community greenhouse gas inventory report 2009. Accessed through the Greenhouse gas data viewer. http://dataservice.eea.europa.eu/PivotApp/pivot.aspx?pivotid=475. Accessed June 2009.

Ellerman, D., and B. Buchner. 2006. Over-allocation or abatement? A preliminary analysis of the EU ETS based on the 2005 emissions data. FEEM Working Paper No. 139.06. Milan, Italy: Fondazione Eni Enrico Mattei.

———. 2007. The European Union emissions trading scheme, origins, allocation, and early results. *Review of Environmental Economics and Policy* 1(1): 66–87.

Ellerman, D., B. Buchner,and C. Carraro, eds. 2007. *Allocation in the European Emissions Trading Scheme: Rights Rents and Fairness*. Cambridge University Press.

European Commission. 2000. Green paper on greenhouse gas emissions trading within the European Union. COM (2000) 87 final.

———. 1998 (June). Climate Change: Towards an EU Post-Kyoto strategy. Commission Communication to the Council and the Parliament. COM (98) 353.

———. 2001. Proposal for a directive of the European Parliament and of the Council establishing a framework for greenhouse gas emissions trading within the European Community and Amending Council Directive 96/61/EC (23 October).

———. 2007, Limiting global climate change to 2 degrees Celsius: The way ahead for 2020 and beyond. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions. SEC(2007) 7, SEC(2007) 8.

European Commission Directorate General for Environment, McKinsey & Company, and Ecofys. 2006. EU-ETS Review: Report on International Competitiveness. http://ec.europa.eu/environment/climat/emission/pdf/etsreview/061222compreport.pdf. Accessed June 2009.

European Union. 2003. Directive 2003/87/EC of the European Parliament and the Council of 13 October 2003 establishing a Scheme for Greenhouse Gas Emission Allowance Trading within the Community and amending the Council Directive 96/61/EC.

———. 2009a, Directive of the European Parliament and of the Council amending Directive 2003/87/EC, so as to improve and extend the greenhouse gas emission allowance trading scheme of the Community.

Fell, H. 2008. EU-ETS and Nordic electricity: A CVAR analysis. Elforsk report 08:57. Stockholm: Elforsk Market Design.

Fischer, C., and A.K. Fox. 2009. Comparing policies to combat emissions leakage: Border tax adjustments versus rebates. RFF Discussion Paper 09-02. Washington, DC: Resources for the Future.

———. 2007. Output-based allocation of emissions permits for mitigating tax and trade interactions. *Land Economics* 83 (4): 575–99.

———. 2004, Output-based allocations of emissions permits: efficiency and distributional effects in a general equilibrium setting with taxes and trade. RFF Discussion Paper 04-37. Washington, DC: Resources for the Future.

Fischer, C., I.W. Parry, and W.A. Pizer. 2003. Instrument choice for environmental protection when technological innovation is endogenous. *Journal of Environmental Economics and Management* 45: 523–45.

Fuss, S. , D. J.A. Johanssonb, J. Szolgayovaa and M. Obersteiner, 2009. Impact of climate policy uncertainty on the adoption of electricity generating technologies, *Energy Policy* 37(2):733-743

Goulder, L.H., I.W. Parry, R.C. Williams, and D. Burtraw. 1999. The cost-effectiveness of alternative instruments for environmental protection in a second-best setting. *Journal of Public Economics* 72(3): 329–60.

Gielen, D., and Y. Moriguchi. 2002. CO_2 in the iron and steel industry: An analysis of Japanese emission reduction potentials. *Energy Policy* 30: 849–63.

Graichen, V., K. Schumacher, F. Matthes, L. Mohr, V. Duscha, J. Schleich, and J. Diekmann. 2008. Impacts of the EU emissions trading scheme on the industrial competitiveness in Germany. Report prepared for the Federal Environment Agency. UBA Climate Change 10/08. Dubai, UAE: FEA.

Harstad, B., and G.A. Eskeland. 2007. Trading for the future: Signalling in permit markets. CMS-EMS Discussion Paper, no. 1429. Evanston, IL, USA: Northwestern University, Center for Mathematical Studies in Economics and Management Science. http://www.kellogg.northwestern.edu/research/math/papers/1429.pdf. Accessed June 2009.

Hepburn, C., M. Grubb, K. Neuhoff, F. Matthes, and M. Tse. 2006a. Auctioning of EU ETS phase II allowances: How and why? *Climate Policy* 6: 137–60.

Hepburn, C., J. Quah, and R.A. Ritz. 2006b. Emissions trading and profit-neutral grandfathering. Oxford University Department of Economics Discussion Paper, no. 295.

Hoel, M., and L. Karp. 2003. Taxes versus quotas for a stock pollutant. *Resource and Energy Economics* 24: 367–84.

Emissions Trading: The Ugly Duckling in European Climate Policy?

Holmgren, K., M. Belhaj, J. Gode, E. Särnholm, L. Zetterberg, and M. Åhman. 2006. Greenhouse gas emissions trading for the transport sector. IVL Report 1703. Stockholm: IVL Swedish Environmental Research Institute Ltd.

Harrison, D., P. Klevnas, D. Radov, and A. Foss. 2007. Complexities of allocation choices in a greenhouse gas emissions trading program. Report prepared for the International Emissions Trading Association (IETA) by NERA Economic Consulting. Boston, MA, USA: NERA.

Hourcade, J.-C., D. Demailly, K. Neuhoff, M. Sato, M. Grubb, F. Matthes, and V. Graichen. 2007. Differentiation and dynamics of EU-ETS industrial competitiveness impacts. Climate Strategies series. Paris and Cambridge: CIRED and University of Cambridge Electricity Policy Research Group.

IEA (International Energy Agency). 2007. *Climate Policy Uncertainty and Investment Risk.*, Paris: IEA.

Kolstad, C. 1996. Learning and stock effects in environmental regulation: The case of greenhouse gas emissions. *Journal of Environmental Economics and Management* 31(1): 1–18.

Kopp, R., R. Morgenstern, W. Pizer, W. and F. Ghersi. 2002. Reducing cost uncertainty and encouraging ratification of the Kyoto Protocol. In *Global Warming in the Asian-Pacific*, ed. R. Mendelsohn and D. Shaw. Cheltenham and Camberly, UK: Edward Edgar Publishing.

Kramton, P., and S. Kerr. 2002. Tradeable carbon permit auctions: How and why to auction, not grandfather. *Energy Policy* 30: 333–45.

Lacombe R. 2008. Economic impact of the European Union Emission Trading Scheme: Evidence from the refining sector. Masters thesis. Cambridge, MA, USA: MIT.

Laurikka, H. 2006. Option value of gasification technology within an emissions trading scheme. *Energy Policy* 34: 3916–28.

Laurikka, H., and T. Koljonen. 2006. Emissions trading and investment decisions in the power sector: A case study in Finland. *Energy Policy* 34: 1063–1074.

LETS Update. 2006. Decision makers summary: LETS/LIFE Emissions Trading Scheme. Report produced for LETS Update Partners by AEA Technology Environment and Ecofys. London: LETS Update.

Löfgren, Å., K. Millock, and C. Nauge. 2008. The effect of uncertainty on pollution abatement investments: Measuring hurdle rates for Swedish industry. *Resource and Energy Economics* 30: 475–91.

Mandell, S., 2008. Optimal mix of emissions taxes and cap-and-trade. *Journal of Environmental Economics and Management* 56: 131–40

McDonald, R., and D. Siegel. 1986. The value of waiting to invest. *Quarterly Journal of Economics* 101: 707–728.

Milliman, S.R., and R. Prince. 1989. Firm incentives to promote technological change in pollution control. *Journal of Environmental Economics and Management* 17: 247–65.

Montero, J-P. 2002. Prices versus quantities with incomplete enforcement. *Journal of Public Economics* 85: 435–54.

Montgomery, W.D. 1972. Markets in Licenses and Efficient Pollution Control Programs. *Journal of Economic Theory* 5(3): 395–418.

Mohr, L., V. Graichen, and K. Schumacher. 2009. Trade flows and cost structure analysis for exposed industries in the EU-27. Working paper, draft report. Cambridge: Climate Strategies.

Morgenstern, R., J. Aldy, E. Herrnstadt, M. Ho, and W. Pizer. 2007. Competitiveness impacts of carbon dioxide pricing policies on manufacturing. RFF Issue Brief 7. Washington, DC: Resources for the Future.

Murray, B., R. Newell, and W. Pizer. 2008. Balancing cost and emissions certainty: an allowance reserve for cap-and-trade. RFF Discussion Paper 08-24. Washington, DC: Resources for the Future.

Newell, R., and W. Pizer. 2003. Regulating stock externalities under uncertainty. *Journal of Environmental Economics and Management* 45: 416–32.

Palmer, K., D. Burtraw, and M. Wråke. 2008. Managing Cost Variability in Emissions Allowance Markets. Elforsk Report 08-56. Stockholm: Elforsk Market Design.

Parry, I.W. 1995. Pollution taxes and revenue recycling. *Journal of Environmental Economics and Management* 29(3): 64–77.

Parry, I.W., R.C. Williams, and L.H. Goulder. 1998. When can carbon abatement policies increase welfare? The fundamental role of distorted factor markets. *Journal of Environmental Economics and Management* 37(1): 52–84.

Pizer, W. 2002. Combining Price and quantity controls to mitigate climate change. *Journal of Public Ecnomics* 85(3): 409–34.

Philibert, C. 2006. Certainty versus ambition: Economic efficiency in mitigating climate change. IEA Working Paper Series. Paris: IEA.

Ponssard, J-P., and N. Walker. 2008. EU Emissions Trading and the cement sector: A spatial competition analysis. *Climate Policy* 8(5): 467–93.

Porter, M.E., and L. van der Linde. 1995. Toward a new conception of the environment-competitiveness relationship. *Journal of Economic Perspectives* 9: 97–118.

Reinaud, J. 2005a. European refinery industry under the EU Emissions Trading Scheme: Competitiveness, trade flows and investment implications. IEA Information Paper. Paris: IEA.

———. 2005b. Industrial competitiveness under the European Union Emissions Trading Scheme. IEA Information Paper. Paris: IEA.

———. 2008a. Issues behind competitiveness and carbon leakage: Focus on heavy industry. IEA Information Paper. Paris: IEA

———. 2008b. Climate policy and carbon leakage. IEA Information Paper. Paris: IEA

Roberts, M., and M. Spence. 1976. Effluent Charges under uncertainty. *Journal of Public Economics* 5: 193–208.

Rosendahl, K.E. 2008. Incentives and prices in an emissions trading scheme with updating. *Journal of Environmental Economics and Management* 26: 69–82.

Rogge, K., J. Schleich, and R. Bets. 2006. An early assessment of national allocation plans for phase 2 of EU emission trading. Working Paper Sustainability and Innovation, no. S1/2006. Karlsruhe, Germany: Fraunhofer ISI.

Schultze, C.L. 1977. *The Public Use of Private Interest.* Washington, DC: Brookings Institution.

Sijm, J., K. Neuhoff, and Y. Chen. 2006. CO_2 cost pass-through and windfall profits in the power sector. *Climate Policy* 6: 49–72.

Sijm, J.P.M, S.J. Hers, W. Lise, and B.J.H.W. Wetzelaer. 2008. The impact of the EU ETS on electricity prices: Final report to Directorate General of Environment of the European Commission. ECN report E-08-007. Brussels: EU.

Skjaerseth, J.B., and J. Wettestad. 2008. EU emissions trading: Initiation, Decision-Making, and Implementation. Ashgate Publishing Ltd. Aldershot, Hampshire, UK.

Smale, R., M. Hartley, C. Hepburn, J. Ward, and M. Grubb. 2006. The impact of CO_2 emissions trading on firm profits and market prices. *Climate Policy* 6(1): 31–48.

Sterner, T. 2007. Fuel taxes: An important instrument for climate policy. *Energy Policy* 35: 3194–3202.

Tietenberg, T. 2002. The tradable permits approach to protecting the commons: What have we learned? FEEM Working paper 36.2002. Milan, Italy: Fondazione Eni Enrico Mattei.

UNFCCC (United Nations Framework Convention on Climate Change). 1998. Kyoto Protocol to the United Nations Framework Convention on Climate Change. New York: United Nations. http://unfccc.int/resource/docs/convkp/kpeng.pdf. Accessed June 2009.

Victor, D.G., and J.C. House. 2006. BP's emissions trading system. *Energy Policy* 34: 2100–2112

Walker N. 2006. Concrete evidence? An empirical approach to quantify the impact of EU emissions trading on cement industry competitiveness. Working Paper PEP06/10. Dublin, Ireland: University College, School of Geography, Planning and Environmental Policy.

Wara, M., and D. Victor. 2008. A realistic policy on international carbon Offsets. Working Paper, no. 74. Stanford, CA, USA": Stanford University.

Widerberg, A., M. Wråke. Forthcoming 2009. The impact of the EU ETS on CO^2 intensity in electricity generation. IVL Report. Stockholm: IVL Swedish Environmental Research Institute Ltd.

Wråke, M., D. Burtraw, S. Mandell, and E. Myers. 2008. Pricing strategies under emissions trading: An experimental analysis. RFF Discussion Paper DP 8-49. Washington, DC: Resources for the Future.

Zhao, J. 2003. Irreversible investment under cost uncertainties: Tradeble emission permits and emissions charges. *Journal of Public Economics* 87: 2765–89.

Zapfel, P., and M. Vainio. 2002. Pathways to European greenhouse gas emissions trading: history and misconceptions. FEEM Working Paper 85.2002. Milan, Italy: Fondazione Eni Enrico Mattei.

Zetterberg, L., and K. Holmgren. 2009. Sectoral agreements and competitive distortions: A Swedish perspective. IVL Report B1835. Stockholm: IVL Swedish Environmental Research Institute Ltd.

Zetterberg, L., K. Nilsson, M. Åhman, A.-S. Kumlin, and L. Birgersdotter. 2004. Analysis of national allocation plans for the EU ETS. IVL Report B1591. Stockholm: IVL Swedish Environmental Research Institute Ltd.

Paper II

Pricing Strategies under Emissions Trading:
An Experimental Analysis

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Abstract

An important feature in the design of an emissions trading program is how emissions allowances are initially distributed into the market. In a competitive market the choice between an auction and free allocation should, according to economic theory, not have any influence on firms' production choices nor on consumer prices. However, the debate around current emissions trading systems shows that parts of industry, the general public and policy making community expect the method of allocation to affect product prices. This paper reports on the use of experimental methods to investigate behavior with respect to how prices will be determined under a cap-and-trade program with different allocation methods. Participants initially display a variety of pricing strategies. However, given a simple economic setting in which earnings depend on behavior, we find that subjects learn to consider the value of allowances and overall behavior moves toward that predicted by economic theory.

> **Key words:** *Carbon dioxide, climate change, Emissions trading, distributional effects, electricity, allocation, auctions*

> **JEL Classification:** *C91, D21, D44, D61, Q54*

Introduction

Predicting effects of policies is not a straightforward task. As an example, expectations of what effects a cap-and-trade program may produce have clearly differed among consumers, policy makers and industry. In particular, how allocation of emissions allowances has influenced downstream product prices has surprised many observers.

At the heart of this problem lies the theoretically simple, but intuitively difficult, concept of opportunity cost. We use an experimental setting to explore the understanding of opportunity cost in an emissions trading market, and how that understanding evolves over time as market

participants learn to profit maximize in the presence of opportunity cost. We also assess how differences in policy design and market environment may influence pricing strategies.

The theory of incentive-based environmental regulations such as cap-and-trade is one of the most important contributions from the field of environmental economics to public policy. How emissions trading programs are implemented and actually work remain a subject of much interest. A cap-and-trade program sets a limit on the number of emissions allowances available. An important objective is to alter relative prices throughout the economy by including the social cost of pollution in product prices. At the same time, higher retail prices for goods such as electricity may be a concern to policy makers and politically controversial, particularly if price effects are regressive; *i.e.* they have a disproportional affect on the poor. To many observers, an initially obvious solution would be to distribute the allowances to the polluting entities for free so that the direct costs of production remain the same, resulting in no change in retail prices.

However, economic theory would argue that at least in competitive markets, retail prices will increase reflecting the economic value of the allowances whether polluting entities received them for free or not see (*e.g.*, Sijm et al. 2006, Burtraw et al. 2002). While using a free allowance may not incur any direct costs to firms, it is an *opportunity cost* of production that will be reflected in retail prices.

Nevertheless, many disagree with the reasoning that firms will, or even have the right to, raise product prices to include the value of emissions allowances received for free. Although the view may be discounted by economists, free allocation is frequently put forward as a means of reducing the downstream price effects of the EU Emissions Trading System (EU ETS). Occasionally, industry has reinforced this view. For example, one energy company official told Point Carbon "If EUAs (emissions allowances in the EU ETS) are auctioned that will only lead to 100 per cent of the carbon price being priced into the electricity price, and thus increase it."[1]

Similar arguments have been raised by governmental authorities. For instance, the German Federal Cartel Office (Bundeskartellamt) stated in a "warning letter"[2] to RWE[3] in December 2006 that the industrial electricity prices charged by RWE in 2005 were abusive as the company had passed on more than 25 percent of the value of its CO_2 emission allowances within its electricity prices. In simplified terms, the Bundeskartellamt recognized that opportunity costs should in principle be taken into account in a business calculation. However, the authority took the view that since the allowances were allocated for free and were necessary for the electricity generation of RWE, they were not actually available for sale and thus would not have a full opportunity cost associated with them.

In a more recent example, the Belgian energy market regulator CREG stated that the country's authorities must act to prevent utilities from making windfall profits by passing on the cost of CO_2

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[1] "Member States look to deal with windfall profits," *Carbon Market Europe, Point Carbon*, October 14, 2005, p. 6.

[2] See press release December 20, 2006, URL: www.bundeskartellamt.de/wEnglisch/News/Archiv/ArchivNews2006/2006_12_20.php

[3] RWE is one of the largest electricity producers in Germany. RWE is subject to the EU ETS and, as all entities participating in EU ETS, has received a major share of their allowances for free according to a grandfathering procedure.

emission allowances to consumers.[4] CREG referred to estimates that electricity producers in the Belgian market generated about 1.2 billion euros ($1.68 billion) in profits between 2005 and 2007 by charging clients for CO_2 emissions allowances they had received for free.

The list of examples may be extended, but the bottom line is that there clearly exist opposing views regarding the applicability of economic theory to emissions trading. The intuition that only direct costs should be included in product prices seems deeply ingrained among parts of the public, the political sphere and industry. Thus, the fundamental question of what price effects different allocation mechanisms may yield is an empirical one.

This paper examines the behavior of firms through the use of experiments in a laboratory setting. It allows us to study aspects that are hard or impossible to capture econometrically based on real market data. For example we can observe the cost pass-through directly and not only market prices, which are influenced by other factors as well. We control and isolate various aspects of the decision problem including the structure of costs and emission rates, etc. In our experiments, individuals make decisions in a structured setting resembling the decisions faced by firms in real markets, in this case a market for emission permits. Participants make production or pricing decisions for products that require two inputs: fuel and an emissions allowance. In some treatments they receive emissions allowances for free and in others they must purchase them.

Our methodology offers a deeper understanding of some of the dynamics of market performance. We aim to capture three distinct aspects of the overarching question of how allocation affects downstream prices. First, we obtain an idea of what pricing strategy is the most intuitive for actors without direct experience in this type of market by observing the initial behavior of subjects. Second, we seek to understand whether different allocation mechanisms lead to different behavior over time. Third, the experiments enable us to study the learning process; how the transition from initial to final behavior unfolds and how this process is influenced by factors such as the allocation mechanism, market transparency and cognitive ability.

The remaining paper is structured as follows: In section 2, we formulate a series of conjectures. Procedures and results are set forth in sections 3-4. Two choice environments are investigated. In one, subjects are given a market price and must choose the quantity that they wish to produce and in the other subjects choose prices that reflect the lowest amount that they would be willing to accept to produce a unit of a good. We supplement the price choice with response data on a cognitive reflection test. Section 5 contains conclusions and further discussion.

2. Conjectures

We conduct an experiment where participants play the role of producers that have to purchase inputs and surrender allowances for production. We proceed with five conjectures about expected behavior.

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[4] As quoted by Reuters news, January 21 2009.

Economic theory suggests that the subjects in the experiment will take the opportunity cost of allowances into consideration in their production decisions. However, as there are very different viewpoints in evidence about what to expect, we cannot take it for granted that economic intuition is shared by all participants. Thus, we expect that:

1) *Subjects display a variety of production/pricing strategies in initial rounds of the experiment.*

As we describe below, the experiments are structured to provide participants with a financial incentive to maximize profit– doing so can lead to substantially greater take-home earnings. However, it may require repeated rounds of the experiment before subjects learn which strategy maximizes their payoffs. We expect that, at least for some subjects, an iterative process may be needed in order to reach the strategy that maximizes the payoff (the efficient strategy):

2) *Subjects improve their payoff over multiple rounds of the experiment.*

There are at least two potential explanations for why behavior under free allocation may differ from when subjects have to pay for allowances. The simpler is that if the subjects pay for allowances they may take this into consideration in their production decision in the same way that they consider the cost of other production factors. That is, subjects do not need to grasp the idea of opportunity cost to arrive at the efficient strategy. A somewhat more complex explanation may be that paying for allowances provides a signal about their market value, making it easier for the subjects to correctly take the opportunity cost into consideration. Previous experience in allowance markets indicates an auction has played an important role in the discovery of the price of allowances (Ellerman et al. 2000). In the experiments we have focused on the former explanation by explicitly stating the market price of allowances. We conjecture:

3) *Subjects can more easily identify the payoff-maximizing pricing strategy under an auction than under free allocation.*

In real life, one of the ways that firms learn, especially in a new market environment, is by watching the performance of other firms. Therefore we consider the role of information about relative performance. We conjecture that if participants receive information about their payoffs relative to the performance of others they will adapt their strategy more quickly. Thus:

4) *Information about relative performance helps subjects to identify the payoff-maximizing pricing strategy and will speed up the transition toward payoff-maximizing pricing.*

Finally, we have argued that the concept of opportunity cost is cognitively challenging. In the context of the price-choice environment explained below we evaluate the performance of subjects according to a cognitive exercise, with the following conjecture:

5) *Subjects that perform better on the cognitive reflection test will have relatively higher payoffs.*

We evaluate these conjectures first in the quantity-choice environment, and then in the price-choice environment.

3. Procedures

In the experiments, participants had the role of producing firms. The experiment was divided into ten 'rounds', each representing one time period. Participants had the capacity to produce up to three units of a product in each round. Each unit produced required one unit of fuel and one

allowance. Fuel cost increased for each unit produced, capturing increasing marginal costs. In some treatments three allowances were given to the producer for free and in other treatments allowances had to be purchased. There was a single market price for allowances and thus their opportunity cost was independent of the number of units produced. If subjects did not produce all three capacity units, they automatically received the market price for any unused allowances that they possessed. The production costs and allowance price were announced at the beginning of each round.

Experiments were run in two different production choice environments. As subject pools and experimental set up changed between the two environments, we do not combine data for the analysis and results from the two environments are not directly comparable.

In the first set up, referred to as the **quantity-choice environment**, subjects were informed of the market price for their product and they had to decide whether or not they wanted to produce each of their three production units. The market prices were randomly determined for each round so that previous production decisions did not affect future prices. The incentives for subjects were straightforward; they should produce a unit as long as it is more profitable than not producing.

The second environment is the **price-choice environment** in which subjects were asked to specify the lowest price that they would be willing to accept (WTA) in order to produce each of their three possible units. If the WTA for an individual unit was less than or equal to a randomly determined market clearing price, that unit was produced and the subject received the market price. Otherwise, the unit was not produced. This mechanism for eliciting WTA is known as the Becker-DeGroot-Marshak (BDM) mechanism (Becker et al., 1964), which is incentive compatible, *i.e.*, individuals have the incentive to truthfully reveal their WTA.[5]

Nevertheless, the BDM method is complicated by the fact that subjects have to consider hypothetical situations that are analogous to proxy bidding. In the experiment, subjects are told that if the randomly determined going market price turns out to be higher than the price they choose, then they will sell for the going market price, but if the going market price is lower, they make no sale. In thinking about this, a subject might reason "if the market price were $$p$, would I be willing to sell for that amount or anything higher." Subjects must do this mental experiment for various price levels until they find a lower bound on what they are willing to accept. It is only in considering such hypothetical scenarios that they can avoid the temptation to choosing a higher price in the (false) hope that doing so cause the "going market price" to be higher.

The BDM set up is attractive because it resembles the pricing mechanism in many electricity markets, which are highly relevant in this context. The BDM mechanism also gives a higher reward to a good understanding of opportunity cost than does the quantity choice environment, thus facilitating the study we set out to do.

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[5] Irwin et al. (1998) find also that in an experimental environment it is cognitively transparent and neutral, meaning that it does not give the subject any unintended feedback.

Pricing Strategies under Emissions Trading: An Experimental Analysis

For each of the two experimental designs, we ran treatments to investigate the effect of relative performance information on production decisions. After each round, subjects were told what their profits were. In some treatments, subjects were informed how their profits ranked compared to the other subjects in the room. All subjects faced the same prices and costs, and the subjects knew that any differences in profits were strictly attributable to differences in production decisions.

Twelve subjects were recruited for each experimental session.[6] Market clearing prices were determined randomly in the interval $1-$10; the price changed between rounds, but was the same for all subjects within a round. We ran treatments using a 2x2x2 experimental design. For each of the two production choice environments, we ran sessions with and without relative information and with and without free allocation, for a total of 8 design treatments. We used two different seeds for the random price and cost draws in each design treatment, for a total of 16 sessions overall. By using the same random draws across treatments we ensure that the differences that we find are attributable to treatment differences rather than the individual market clearing prices. Additionally, multiple seed values were used to ascertain that the results can be generalized for more than one set of prices.

Subjects were recruited among undergraduate and graduate students at the University of Virginia in Charlottesville (for the quantity choice environment) and at the Royal Institute of Technology in Stockholm (price choice). Subjects' earnings in experimental dollars were converted to real currency and given to them at the end of the experiment.[7]. The representativeness of student subjects and the generalizability of results obtained in a laboratory setting is a recurring concern in experimental studies, and generalizing result outside the subject pool should always be done with care. This has been discussed extensively in the literature[8], and factors found to influence subject behavior include presence of moral considerations, whether a subject's behavior can be scrutinized by other participants, the decision context, potential self selection of subjects and the stakes in the experiment. The experiments in this study were set up so as to minimize the influence of these factors (please see Appendix C for a discussion of laboratory procedures and Appendix D for a copy of the experimental instructions) and in order to obtain treatment effects within the subject pool the critical parameter is appropriate randomization of subjects. Further, as one objective of this study is to capture the intuition held by a 'wider public', using students acting as firms rather than trained professionals also carries benefits.

There were some differences in how we ran the experiments for the two different production choice environments, as summarized in Table 1.[9] In the quantity-choice environment, the fuel input costs were $1, $3, and $5 for production units 1, 2, and 3 respectively while the allowance prices

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[6] Two of the sessions using the BDM mechanism contained 11 subjects and one contained 13 subjects.

[7] In Charlottesville (US), the exchange rate between experimental dollars and US dollars was 0.1. In Stockholm (Sweden) an exchanged rate of 1 SEK for 1 experimental dollar was used. At the time of the experiment US $1 approximately corresponded to 7 SEK.

[8] See for instance Harrison and List (2004), Danielson and Holm (2007), Levitt and List (2007), Levitt and List (2009)

[9] In both cases the subjects played multiple rounds and our results are based on the first 10 rounds. In the quantity-choice environment the experiments lasted for 10 rounds only, while in the price-choice environment subjects completed 20 rounds but with a change of treatment from round 11 onwards.

were invariant at \$4 per allowance. In the price-choice environment, fuel input costs and the allowance prices were stochastic and thus changed between rounds. However, the expected values correspond to those in the quantity-choice environment. The fuel input prices ranged from \$0.50-\$1.50, \$2.50-\$3.00, and \$4.50-\$5.50 for units 1, 2 and 3 respectively. The allowance prices ranged from \$3.50 and \$4.50. In each round, the four input values were randomly determined with each \$.50 increment in the range being equally likely. In addition, at the end of the price choice treatments subjects filled out a questionnaire with several cognitive questions. This was not done in the quantity choice environment.

The experiments conducted are similar to those in a study by Plott (1983), which reports on a laboratory study of behavior when limited and tradable licenses were required for economic activity that imposed an externality on others. The value of the licenses was created by their scarcity and reflected in a market in which licenses could be bought or sold. In this setting, even when licenses were distributed initially for free, the price of production reflected opportunity costs. Moreover, the licenses ended up being used in an approximately efficient manner by producers who valued them most. However, one aspect of the experiments was not general in that licenses

Table 1   Payoff Structure For The Two Treatments

Sequence	Quantity Choice	Price Choice
Initial information	$\left\{ E_a, w_a, w_f^i, p_g^r \right\}$	$\left\{ E_a, w_a^r, w_f^{r,i}, \tilde{p}_g \right\}$
Decision	$\hat{q}$	$\hat{p}_g^i$
Realization of uncertain variables		p_g
Production level	$q = \hat{q}$	$q = \max\{ i \leq 3 \mid \hat{p}_g^i \leq p_g \}$
Profits	$\sum_{i \leq q} \left(p_g - w_f^i \right) - \left(q - E_a \right) w_a$	$\sum_{i \leq q} \left(p_g - w_f^{r,i} \right) - \left(q - E_a \right) w_a^r$

Where:

E_a = endowment of allowances

w_a = price of allowances

w_f = price of fuel

p_g = price of produced good

$\hat{q}$ = quantity bid (in quantity choice experiment)

q = quantity produced

$\tilde{p}_g$ = random price of produced good (in price choice experiment)

$\hat{p}_g$ = bid price of produced good (in price choice experiment)

and superscripts denote:

$$i = \text{production opportunities}, \ i \in \{1, 2, 3\}$$

$$r = \text{round}, \ r \in \{1, \ ...,10\}$$

were not distributed in approximate proportion to their ultimate use, which has been the case in previous actual allowance markets. Rather, licenses were distributed initially so that the maximum efficiency of the original allocation was 46%, had no trading occurred. Consequently, producers had to engage actively in the secondary market for licenses in order to maximize profits from production, and this was likely to contribute to the discovery and recognition of the market-driven opportunity costs of the licenses. Plott acknowledges that the efficiency of a license policy "might be affected by the initial distribution of licenses by placing them in the 'right hands' initially" (p. 118). In our experiment when allowances were distributed for free subjects had sufficient allowances to cover production, perhaps making it more difficult for subjects to recognize their opportunity cost and encouraging them to use different decision rules in determining their production activities.

4. Experiment Results

Results are reported for each of the experimental settings, beginning with the quantity-choice environment and followed by the price-choice environment.

4.1 The Quantity-Choice Environment

In the quantity-choice environment we ran 8 sessions, each of which had 12 participants making 10 individual production decisions. As described above, we ran sessions with free allocation of emissions allowances and sessions where allowances had to be purchased. We denote free allocation as FA. Sessions that provided relative earnings information are denoted RI.

Result #1: *We confirm conjecture #1 in the quantity choice environment. Participants display a variety of production strategies in the quantity choice environment in the early rounds of the experiment.*

Figure 1 depicts the proportion of erroneous decisions[10] made by the 12 participants in each of the 10 rounds. Each data point is the average of the values for the two replications that we did using different seeds. In the first round there are about twice as many errors made in treatments with grandfathering than in treatments with no grandfathering. As shown in figure 1, many participants make production decisions that fail to maximize profits in early rounds of the experiment. This effect diminishes over time, as discussed below.

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[10] An erroneous decision is defined as when the subject decided to produce and sell at a market price below marginal cost, or not to produce and sell when marginal cost was below market price.

Figure 1 Proportion of Decisions That Are Erroneous in Each Round by Treatment Type in the Quantity Choice Experiment

Result #2: *We confirm conjecture #2 in the quantity choice environment. Subjects improve their payoff over multiple rounds of the experiment. Greater improvement occurs among participants that receive emissions allowances for free.*

A central question is whether learning is facilitated by having to pay for allowances and/or the presence of relative information. A more formal test of this is performed by looking at each individual's behavior in the first three rounds (1-3) compared to the last three rounds (8-10). If there is a learning effect, subjects would adopt a more efficient strategy in the latter rounds and, thus, for a majority of the subjects we expect to see a positive difference between the number of errors in early compared to late rounds. Let e^j denote subject j's error, which is defined as the absolute value of the difference between subject j's quantity bid choice and the choice that would maximize j's payoff.[11] We use individual participants as observations, and compare e^j for rounds 1 to 3 with rounds 8 to 10. Table 2 reports the result of the Wilcoxon Matched-Pairs Signed Rank test, for all four possible combinations of treatments.

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[11] That is, for subject j in a given round the error is: $e^j = \left| \hat{q}^j - \breve{q} \right|$, where $\breve{q} = \max i : p_g \geq w_a + w_f$.

Table 2   Learning Effects in the Quantity Choice Environment

	W+	W-	p-value	n
Free Allocation, No Relative Info	186	4	0.0000267***	19
Free Allocation, Relative Info	137.5	15.5	0.00209***	17
No Free Allocation, Relative Info	78	0	0.0004883***	12
No Free Allocation, No Relative Info	70.5	7.5	0.009272***	12

Note: Results from a Wilcoxon matched-pairs signed rank test.

*significant at the 10 % level, **significant at 5% level and ***= significant at 1 % level.

The number of observations with a positive difference in deviation between early and late rounds (W+ in Table 2) dominates those with negative difference (W-). This is consistent with a learning effect. The difference is large enough to be statistically significant for all treatment types, indicating that subjects learned to profit-maximize in all of the treatments.

Result #3: *We confirm conjecture #3 in the quantity choice environment. Subjects more easily identify the payoff-maximizing production strategy under an auction than under free allocation, especially in the early rounds.*

In order to formally test whether different treatments have a different impact on behavior we perform the non-parametric Wilcoxon rank sum test to examine the number of errors committed. We use individual participants as observations, and e^j summed over rounds 1 to 5 and rounds 6 to 10 respectively as a measure of performance. Table 3 contains the results of the test of a null hypothesis that treatment effects do not matter.

Table 3   Treatment Effects in the Quantity Choice Environment

	Rounds 1 to 5			Rounds 6 to 10	
	W	**p-value**		**W**	**p-value**
Effect of Free Allocation	1840	p<=.000354***	Effect of Free Allocation	2243	p<=.5358
Effect of Relative Information	2271.5	p<=.6816	Effect of Relative Information	2178	p<=.2733

Note: Results from Wilcoxon rank sum test.
*=significant on the 10 % level, **=significant on 5% level and *** = significant on 1 % level.

In Table 3, the Wilcoxon test shows a statistically significant difference between FA and no FA for rounds 1-5, but not for rounds 6-10. The difference in early rounds suggests that the profit-maximizing strategy may be more comprehensible when the cost of an emissions allowance is a direct cost rather than an opportunity cost of production. The fact that the difference disappears in the last 5 rounds suggests that subjects in free allocation treatments learn to incorporate the opportunity cost over time.

Result #4: *We cannot confirm conjecture #4 in the quantity-choice environment. Information about relative performance does not speed the transition toward payoff-maximizing production.*

Table 3 shows that the results of the Wilcoxon tests for treatments with RI vs. no RI were insignificant, indicating that relative information did not appear to improve individual performance in early or late rounds of the experiments. Comparisons within allocation environments also yielded no differences between the relative information and no relative information treatments (FA RI vs. FA no RI, W = 533.5 p<=.2655; no FA RI vs. no FA no RI, W=580.5 p<=.8852). A potential explanation is that the quantity choice environment by design will result in several subjects performing equally well and thus receiving identical ranking in each round. Such 'ties' in ranking are likely to decrease the relative information's signal to subjects to alter pricing strategy.

4.2 The Price Choice Environment

In the quantity choice environment, subjects are presented with a price and they have to make a binary decision to produce or not. In the price choice environment, the decision is more open ended; participants choose a price from a broad range. The data therefore allow us to conduct richer analyses in that we may not only observe the numbers of errors made but also the size of the error by calculating the deviation between bid price and optimal price. In a given round, such a deviation, made by subject *j* for production unit *i*, is defined by the absolute value of the difference between the bid price and the costs of allowances plus fuel: $d^{i,j} = \left| \hat{p}^{i,j} - \left(w_a + w_f^i \right) \right|$. A minority (17 %) of the observations is associated with asking prices above the optimal price. Also, at the end of these treatments subjects filled out a questionnaire with several cognitive questions, adding another dimension to our investigation.

Figure 2 illustrates the pricing behavior. It plots the average absolute value of the deviation (*d*) from optimal pricing behavior for each round and each of the four different treatments in the price choice environment.[12]

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[12] The sessions included a second treatment in which the allowance allocation mechanism changed in rounds 11-20. We do not report the data here as performance in the second treatment was influenced by the first treatment. However, it is noteworthy that we did observe that at the beginning of the second treatment there were substantially increased deviations relative to the end of the second treatment, especially for the groups who had to pay for allowances in the first treatment and were now receiving them for free. Since, by this time participants were familiar with the set up, we feel the change beginning in round 11 is attributable to their production strategy rather than potential misunderstanding of the experimental design.

Figure 2   Average Absolute Deviation From Theoretical Optimum By Round

Result #5: We confirm conjecture #1 in the price-choice environment. Participants display a wide range of pricing strategies in early rounds of the experiment.

Similar to the quantity choice experiment, subjects initially apply pricing strategies that deviate from what would be economically optimal.

Result #6: We confirm conjecture #2 in the price-choice environment. Subject performance moves closer to maximizing payoffs and shows smaller variation over rounds of the experiment.

With the possible exception of the group who had to pay for allowances and received no relative information (No FA, No RI), all groups exhibit rather clear learning affects in that the lines in figure 2 all show downward sloping trends.

Looking at all choices taken in round 1 through 5, the standard deviation of *d* is 1.68. This may be compared to a standard deviation of 1.55 for rounds 6 to 10. Thus, there is a smaller variation in latter rounds, but the difference is not large. The share of bid prices that is more than 2 experimental dollars below (above) optimum is 37 % (2 %) in round 1 through 5 and 25 % (2 %)

for rounds 6 through 10.[13] This supports the general impression that the subjects price in a more optimal way in latter rounds, *i.e.*, there is a learning effect.

Result #7*: We confirm conjecture #3 in the price-choice environment. Participants bid closer to the profit maximizing price when they have to buy allowances rather than receiving them for free.*

Again, we use the non-parametric Wilcoxon two tail rank sum test to formally test whether different treatments have a different impact on pricing behavior. Table 4 contains the results both for round 1 to 5 and for round 6 to 10.

Free allocation yields absolute deviations that are statistically higher than the alternative where the subjects must buy allowances, as seen from the first row in Table 4. This is the case both for the first 5 rounds and for the last 5 rounds. That is, having to pay for allowances helps the subjects in pricing optimally. The second row in Table 4 shows that relative information has a statistically significant impact on absolute deviation for rounds 6-10, but not for rounds 1-5. This suggests that relative information helps the subjects in their pricing decisions in later rounds indicating that relative information affects learning. This result differs from what we found in the quantity choice environment, where RI had no significant impact, and perhaps is because the BDM mechanism used in the price choice environment is not as straightforward as the binary decision mechanism in the quantity choice environment, giving a larger role to relative information about how other subjects are performing.

Table 4   Treatment Effects in Price Choice Environment

Rounds 1 to 5			Rounds 6 to 10		
	W	*p-value*		*W*	*p-value*
Effect of Free Allocation	1943	p<=.001405***	Effect of Free Allocation	2069	p<=.01731**
Effect of Relative Info	2575	p<=.3745	Effect of Relative Info	2704.5	p<=.006421***
Note: Results from a Wilcoxon two tail rank sum test.					
* significant on the 10 % level, ** significant on 5% level and *** significant on 1 % level.					

[13] The corresponding values that are more than 1 experimental dollar below (above) optimum is 49 % (4 %) for round 1-5 and 35 % (5 %) for round 6-10.

Result #8: *We confirm conjecture #4 in the price-choice environment. The rate of learning improves when participants receive information about their relative performance.*

We assess learning in the price choice environment using the Wilcoxon Matched-Pairs Signed Rank Test by comparing performance for each subject in the first three rounds with the last three rounds. Table 5 shows a rather clear and interesting picture in that there is a highly significant learning effect in the two treatments that include relative information, as seen from row 2 and 3. However, neither of the treatments without relative information shows a significant learning effect. This result differs from our findings in the quantity choice environment where all treatments exhibit significant learning. Again, this may be due to the decision environment being more complex, so fewer subjects learned the profit-maximizing strategy and relative information had a stronger effect on behavior.

Table 5   Learning Effects in Price Choice Environment

	W+	W-	p-value	n
Free Allocation, no Relative Information	192	84	.1037	23
Free Allocation, Relative Information	506	55	.00005816***	33
no Free Allocation, Relative Information	150	3	0.00007629***	17
no Free Allocation, no Relative Information	160	71	0.1262	21

Note: Results from a Wilcoxon matched-pairs signed rank test, all subjects.

* significant on the 10 % level, ** significant on 5% level and *** significant on 1 % level.

Controlling for Cognitive Ability

The results reported in Table 4 and Table 5 suggest there is an influence both from the choice of allocation approach and from the relative information in the price choice environment. They also suggest that relative information facilitates learning, while the allocation process has an absolute impact on pricing behavior. As the task the subjects perform in the experiment is obviously not trivial, it would be interesting to see whether subjects that are trained for, or in other ways are skilled in, performing more analytical assignments manage to price in a more optimal way than other subjects.

In order to measure problem-solving skills, the subjects were asked to answer several questions directly after the last round in the experiment. For this analysis, we focus on three questions (see Appendix A) that make up a Cognitive Reflection Test (CRT) as presented in Frederick (2005), with wording adapted from a baseball to hockey motif for the Swedish audience. As defined by Frederick, "CRT measures 'cognitive reflection'—the ability or disposition to resist reporting the response that first comes to mind." A high performance on the CRT test is strongly correlated with high performances on other tests of cognitive abilities. We found that participants with low CRT scores (0-1) deviated further from profit-maximizing pricing than did participants with high CRT scores (2-3) (Wilcoxon rank sum, W = 2570, $p \leq .000003128$).

Result #9: *We confirm conjecture #5 in the price-choice environment. High performance on the cognitive test is associated with payoff-maximizing bidding behavior.*

In one of our treatments (no FA, RI), participants performed particularly well on the CRT test with 20 out of 23 receiving a high score. Because this sample is somewhat biased, it was particularly important to control for high scores in some of our analyses. Repeating the Wilcoxon Matched-Paired tests comparing performance in rounds 1-3 to rounds 8-10 for the High CRT-group yields mainly similar results but with a few notable differences (see Appendix B). First, the impact from relative information is no longer significant, neither for all ten rounds nor for the last five rounds, see Table B1. This suggests that this group does not benefit as much from relative information. Second, the High CRT-group shows significant learning in the FA no RI treatment (corresponding to the first row in Table B2). We return to these findings later in this section.

Regression Analysis

To analyze the data further, we estimated the following error components model with random effects and Huber-White standard errors, which are robust to the presence of heteroskedasticity:

$$Absolute\ Deviation = \alpha + \beta_1 * Round + \beta_2 * Round^2 + \beta_3 * FA$$
$$+ \beta_4 * FA * Round + \beta_5 * RI + \beta_6 * RI * Round + \varepsilon + \mu \quad (1)$$

The variable *Round* denotes which round (1 through 10) data refers to and is used to capture learning effects. As seen from (1) we allow for learning to be non-linear by also including *Round squared*. *FA* takes on the value 1 if the subjects receive allowances for free during these rounds and zero otherwise. *RI* takes on the value 1 if the subjects receive relative information. We allow for the choice of allocation and the presence of relative information to influence learning[14]. This will be captured by β_4 and β_6, respectively. Any absolute effect from allocation choice or relative information will be captured by β_3 and β_5, respectively. The idiosyncratic component of the error is captured by ε and μ represents random effects.

As discussed above, we expect to see learning in all treatments, but probably at a decreasing rate. That is, as the explained variable is measured in absolute terms, β_1 is expected to be negative. If the learning effect decreases over rounds, which is a reasonable expectation, β_2 should be positive but, in absolute terms, less than β_1. Furthermore, we expect that having to pay for allowances facilitates optimal pricing. That is, when *FA*=1 (allowances are received for free) absolute deviations are expected to be higher than otherwise, and thus the expected sign of β_3 is positive. If the subjects who receive allowances for free initially show a higher deviation from optimal pricing but learn to price more optimally over time β_4 will be negative. A negative value of β_5 implies that relative information facilitates optimal pricing behavior. Finally, we expect β_6 to be negative, *i.e.*, with relative information learning per round is greater than without.

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[14] We only allow for these effects to be linear. Also including the variables *GF*Round^2* and *RI*Round^2* results in a negligible increase in the model's explanatory power but yields no further insights than the model given by (1).

The results are given in Table 6, which contains three regressions. The first model uses all data, the second only data from subjects who scored high on the cognitive test and the third uses data for subjects with low scores on the cognitive tests. [15]

As seen in Table 6, the explanatory power of the model is low. Subjects seem to use a lot of trial-and-error, which causes noise in the data. This conjecture is further strengthened by the observation that the explanatory power is higher for the high CRT score group than for the group with low scores.

Looking at the All Subjects model, we see that it supports our expectations as described above. First, there is a clear evidence of learning in that *Round* has a highly significant negative coefficient. It is also evident that the rate of learning decreases in the number of rounds played, since *Round squared* has a positive and highly significant coefficient. Note, however, that learning happens throughout all ten rounds as the positive impact on deviation from *Round squared* never outweighs the negative impact from *Round*.[16] The impact on pricing from the choice of allocation mechanism is also as expected. Deviations from optimal pricing are significantly higher when allowances are received for free, as seen by the coefficient for *FA*. The effect is rather large. In the first round deviation is almost 0.8 experimental dollars higher under free allocation, which corresponds to one fifth of the expected value of an allowance. However, this effect decreases during the experiment, *i.e.*, subjects learn to take the opportunity cost into consideration. This is seen from the negative and highly significant coefficient for *Round*FA*. After ten rounds almost half (45 %) of the

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[15] We also ran regressions using all subjects with the above explanatory variables as well as binary variables for each question in the cognitive test: "1" for a correct answer and "0" for an incorrect answer. The coefficients for questions 2 and 5 were significant, suggesting that correct answers to these questions had the highest correlation with pricing strategies that maximized earnings in the experiment.

[16] At Round = 10 we have that 10*-0.182+10^2*0.0131=-0.51.

Table 6 Regression Results

	All Subjects		High CRT Score		Low CRT Score	
	Coefficient	*Sdt. error*	*Coefficient*	*Std. error*	*Coefficient*	*Std. error*
Constant	2.017***	.326	1.6554***	.435	2.7522***	.467
Round	-0.182***	.0572	-0.1830***	.0668	-0.2286**	.0986
Round^2	0.0131***	.0131	0.0101**	.00473	0.01737**	.00735
FA	0.8313***	.265	1.0373***	.340	-0.1121	.413
Round*FA	-0.0389*	.0211	-0.0707***	.0246	0.06336	.0398
RI	-0.0982	.267	-0.2425	.306	0.7274*	.377
Round*RI	-0.0723***	.0215	-0.00187	.0263	-0.2024***	.0348
R-squared	0.092		0.105		0.068	
n	3150		1860		1290	

Note, *=significant on the 10 % level, **=significant on 5% level and, *** = significant on 1 % level.

effect seen in the initial round has disappeared. Much as expected, relative information has no initial impact on pricing behavior (the coefficient for *RI* is not significant) but it shows clear evidence of facilitating learning as *Round*RI* is highly significant and negative. The effect of relative information after 10 rounds is of the same magnitude as the initial effect of having to pay for allowances.

Overall, information about relative performance has a higher impact on the pricing behavior of participants that have a low score on the cognitive test than those that have a high score. For the group with high CRT scores, most effects are of the same magnitude as for the entire group. The main difference lies in the impact from relative information. This group shows a small initial effect from having relative information barely significant even at the 10 % level. More interestingly, the presence of relative information does not help this group's learning process, which is seen from that *Round*RI* is not significant. A possible explanation of this lies in the nature of the information. As this group generally performed better, *i.e.*, showed less deviation from optimal pricing, they probably ranked high. A low ranking sends a clear signal that you should change behavior, a high ranking however signals that you should probably stick to your strategy. In that sense, this result is not surprising.

Looking at the highly significant coefficients for *Round* and *Round squared* for the group with low CRT score we observe a higher initial learning effect. Interestingly and rather surprisingly, the rate decreases faster over rounds than for the entire group. By round 10, the impact on learning from playing another round is of the same magnitude as for the entire group. The low CRT group shows no effect from having received allowances for free or not. The learning effect shows only weak significance and has an unexpected sign. One possible interpretation of this is that this group immediately and correctly took the opportunity cost into consideration. However, the data suggests that, on average this low CRT group priced far from optimal in both allocation treatments. The

relative information, on the other hand, seems to have a substantial impact on learning for this group. There is an initial effect that, even though it is only weakly significant, has an unexpected sign. However, the learning effect from relative information over rounds has the expected sign and is rather large such that after four rounds it outweighs the initial effect. Even though the sign of the initial effect is surprising and hard to explain, the general result that this group is highly influenced by the presence of relative information seems very plausible. This group is likely to rank low in relative earnings and the relative information thus sends a clear signal to change strategy to the next round.

5. Conclusion and Discussion

This paper reports on the use of experimental methods to investigate behavior with respect to how prices will be determined under a cap-and-trade program. Previous studies on price effects of emissions trading include econometric time series analyses performed by Bunn and Fezzi (2007) in the UK electricity market and Fell (2008) in the Nordic market. Both studies attempt to determine the influence on electricity price of changes in the price of factors of production including fuel and emissions allowances. These studies find that in these relatively competitive markets electricity prices respond to a change in the price of emissions allowances, suggesting that consumers pay for a significant portion of the value of emissions allowances even when the industry has received them for free.

However, the econometric studies do not remove the ambiguity about how much pass-through we actually observe. The ambiguity stems from the heterogeneity of technologies within the market. In the electricity sector, which has been at the core of previous emissions trading programs, technologies differ both with respect to the marginal cost of producing electricity and with respect to their emission rate. In most wholesale power markets (except where power purchase agreements are in place), each electricity producer receives the same price per unit produced. The price depends just on the bid of the last generator, *i.e.* the one with the highest marginal cost needed to meet demand. Sijm et. al. (2006) refer to the pass-through behavior of the individual generator as 'add-on' and the increase of the bid for the marginal unit, which will determine the electricity price, as 'work-on'. Econometric studies have estimated the price that is observable in the market, which is essential data for calculating the 'work-on' rate, but one cannot actually calculate that rate without knowing which kind of technology is on the margin at any given point in time.

An additional source of ambiguity affects the public discussion, where the pass-through issue is driven by concerns of perceptions of fairness, compensation, and potential windfall profits. These perceptions are determined by the average pass-through of costs incurred across the industry. A marginal pass-through rate (work-on rate) of 100 percent does not indicate whether the industry is earning revenue that is more or less than 100 percent of its cost. The cost to industry depends on the emission rates of infra-marginal generators, which may on average be more than or less than the emissions rate of the marginal unit.

In contrast to econometric studies, the experimental approach applied in this paper allows us to study the pass-through decision directly. Furthermore, as Binmore and Klemperer (2002) note, the use of experiments to explore economic behavior has persuasive power in addition to investigative value.

The initial bidding behavior of many subjects confirms the intuition that the method of allocation could influence downstream prices. Our findings also confirm that the concept of

opportunity cost is intuitively difficult. Participants initially exhibit a range of behavior, encompassing full or nearly full pass-through of allowance value regardless of how allowances are distributed by some participants, and little or no pass-through when allowances are given away for free by other participants. However, we find learning to be a central process for understanding firm behavior and market performance. Over time participants make decisions that are closer to profit maximizing levels in treatments both with and without free allocation.

We find that learning as well as market performance is influenced by policy design and market environment. Participants make decisions that are closer to economic theory when they have to purchase allowances than when they receive them for free. Further, when subjects receive information on their earnings relative to others, *i.e.*, under more transparent market conditions, there is stronger improvement in profit-maximizing behavior over time.

These results contribute to the debate around the design of emissions trading programs, which from a purely theoretical point of view may seem trivial at first glance. Our findings also hint at some of the dynamics of the discussion. Economic analysis of emissions trading policy broadly supports the use of an auction for the initial distribution of emissions allowances. One reason is that an auction reduces the incentives for regulated parties to engage in costly rent-seeking behavior in order to gain a more generous future allocation. Another is that in regulated cost-of-service markets, which characterize more than half of the United States and other parts of the world, free allocation can move consumer prices away from the marginal social cost of production and therefore distort resource allocation away from an efficient outcome. [17] Compared with other approaches, an auction helps maintain administrative transparency and the perception of fairness,[18] which are important principles for the formation of a new market for an environmental commodity. One particularly insidious aspect of free allocation that has been ubiquitous in the EU ETS for carbon dioxide (CO_2) is the adjustment to allocation rules for new emissions sources and for old sources that retire. [19] Finally, an auction makes revenue potentially available for reducing pre-existing taxes, which would lower the social cost of the program, [20] or for program-related investments in research and technology that many argue to be important for climate policy.

Nevertheless, in the first phase of the EU ETS from 2005-2007 approximately 99% of the allowances were distributed for free to incumbent emitting firms, and in the second phase from 2008-2012 approximately 96.5% will be distributed for free. In December 2008, EU decided in favor of the use of an auction for the majority of allowances going to the power sector beginning in 2013, and an expansion of the auction from 20 percent in 2013 to the entire market by 2027 for the industrial sector, with some exception for sectors at risk of competition from industries abroad that

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[17] Beamon et al. 2001. Burtraw et al. 2001. Burtraw et al. 2002. Parry 2005.

[18] Binmore and Klemperer, 2002.

[19] Åhman et al. 2007; Åhman and Holmgren 2006.

[20] Bovenberg and Goulder, 1996; Bovenberg and de Mooij, 1994; Golder et al. 1999; Parry et al. 1999. Smith et al. 2002.

do not have climate policies in place. [21] This decision has been opposed by parts of industry as well as some Member States, referring to the effects which auctioning may have on downstream prices.

This paper offers an explanation of the discrepancy between public ex-ante expectations and the ultimate outcome with regards to price effects of cap-and-trade with free allocation. Widely-held intuition suggests that free allocation would decrease downstream price effects compared to an auction, while we find little evidence that in the long run the choice of allocation methodology will affect pricing strategies in a competitive market.

References

Åhman, M., Burtraw, D., Kruger, J. & Zetterberg, L., 2007, "A Ten-Year Rule to Guide the Allocation of EU Emission Allowances". *Energy Policy* 35:1718-1730.

Åhman, M. & Holmgren, K., 2006, "New entrant allocation in the Nordic energy sectors: current principles and options for the EU ETS." *Climate Policy* 6:423-440.

Beamon, J. Alan, Tom Leckey, and Laura Martin. 2001. "*Power Plant Emission Reductions Using a Generation Performance Standard* [Draft]". Washington, DC: U.S. Department of Energy, Energy Information Administration.

Binmore, K. and P. Klemperer, 2002. "The Biggest Auction Ever: The Sale of the British 3G Telecom Licenses." *The Economic Journal* 112: C74-C76.

Bovenberg, A.L., and Goulder, L.H. 1996. "Optimal Environmental Taxation in the Presence of Other Taxes: General Equilibrium Analyses." *American Economic Review* 86: 985–1000.

Bovenberg, A.L., and de Mooij, R. 1994. "Environmental Levies and Distortionary Taxation." *American Economic Review* 84: 1085–89.

Bunn, D.W. and C. Fezzi. 2007. "Interaction of European Carbon Trading and Energy Prices." *The Fondazione Eni Enrico Mattei Note di Lavoro Series*. Fondazione Eni Enrico Mattei, 63.2007.

Burtraw, D., Palmer, K., Bahrvirker R. And Paul, A., 2002," The affects on asset values of the allocation of carbon dioxide emission allowances", *Electricity Journal* 15(5), 51-62

Burtraw, Dallas, Karen Palmer, Ranjit Bharvirkar, and Anthony Paul. 2001. "*The Effect of Allowance Allocation on the Cost of Carbon Emissions Trading*". RFF Discussion Paper 01-30. Washington, DC: Resources for the Future.

Danielson, A. and Holm, H. 2007, *Do you trust your brethren? Eliciting trust attitudes and trust behavior in a Tanzanian congregation.* Journal of Economic Behavior & Organization.Vol. 62 (2007) 255–271

Ellerman, A. Denny, Paul L. Joskow, Richard Schmalensee, Juan-Pablo Montero and Elizabeth M. Bailey, 2000. "*Markets for Clean Air: The U.S. Acid Rain Program*", Cambridge: Cambridge University Press.

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[21] Competition from regions that have not adopted climate targets is expected to result in a loss of European output in some industries without any gains in terms of emission reductions. For an extensive discussion of this topic, see for instance Hourcade et. al. (2008) and Reinauld, (2008)

Fell, Harrison, 2008. "EU-ETS and Nordic Electricity: A CVAR Analysis", Resources for the Future Discussion Paper 08-31.

Fischer, C. and A.K. Fox (2007) "Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions", *Land Economics*. 83: 575-599.

Frederick, Shane, 2005. "Cognitive Reflection and Decision Making", *Journal of Economic Perspectives* 19 (4): 25-42.

Goulder, L.H., Parry, I.W.H., Williams III, R.C., and Burtraw, D. 1999. "The Cost-Effectiveness of Alternative Instruments for Environmental Protection in a Second-Best Setting." *Journal of Public Economics* 72(3): 329–360.

Irwin, Julie R., Gary H. McClelland, Michael McKee, William D. Schulze and N. Elizabeth Norden, 1998. "Payoff Dominance vs. Cognitive Transxparency in Decision Making," *Economic Inquiry*, 36(April):272-285.

Harrison, G. and List, J., 2004, *Field Experiments*, Journal of Economic Literature Vol. XLII (December) pp. 1009–1055

Hourcade, J-C., Demailly, D. Neuhoff, K., Sato, M., Grubb, M, Matthes, F., Graichen, V., 2008. "*Differentiation And Dyamics of EU ETS Industrial Competitiveness Impacts*", Climate Strategies report

Levitt, S. and List, J, 2009, *Field experiments in economics:The past, the present,and the future*, European Economic Review 53 1–18

Levitt, S. and List, J, 2007, *What Do Laboratory Experiments Measuring Social Preferences Reveal About the Real World?* Journal of Economic Perspectives Vol 21 (2) 153–174

Parry, Ian W.H. 2005. "Fiscal Interactions and the Costs of Controlling Pollution from Electricity", *Rand Journal of Economics* 36(4): 850-870.

Parry, I.W.H., Williams, R.C., and Goulder, L.H. 1999. "When Can Carbon Abatement Policies Increase Welfare? The Fundamental Role of Distorted Factor Markets." *Journal of Environmental Economics and Management* 37(1): 52–84.

Plott, Charles, 1983. "Externalities and corrective Policies in Experimental Markets," *The Economic Journal* 93 (369 March): 106-127.

Reinaud, J., 2008. "*Issues Behind Competitiveness and Carbon Leakage*", International Energy Agency information paper.

Sijm, Jos, Karsten Neuhoff and Yihsu Chen, 2006. "CO_2 Cost Pass-Through and Windfall Profits in the Power Sector", *Climate Policy*, 6:49–72.

Smith, A.E., Ross, M.T., and Montgomery, W.D. 2002. "*Implications of Trading Implementation Design for Equity-Efficiency Trade-Offs in Carbon Permit Allocations.*", Washington, DC: Charles River Associates.

Appendix A – Questions in the Cognitive Reflection Test

1. A hockey stick and puck cost 110 Canadian dollars in total. The stick costs 100 more than the puck. How much does the puck cost?

2. In a lake, there is a patch of lily pads. Every day, the patch doubles in size. If it takes 48 days for the patch to cover the entire lake, how long would it take for the patch to cover half of the lake?

3. If it takes 5 machines 5 minutes to make 5 widgets, how long would it take 100 machines to make 100 widgets?

Appendix B – Non-parametric Tests on the High CRT-Score Group

Table B1   Results from the Wilcoxon Rank Sum Test: Subjects with High CRT-scores

Rounds 1 to 10			Rounds 6 to 10		
Treatment 1	*Treatment 2*	*p-value*	*Treatment 1*	*Treatment 2*	*p-value*
FA no RI	FA RI	p<.7921	FA no RI	FA RI	p<.79
FA no RI	no FA no RI	p<.9264	FA no RI	no FA no RI	p<.5588

Note, *=significant on the 10 % level, **=significant on 5% level and *** = significant on 1 % level.

Table B2   Wilcoxon Matched-Pairs Signed Rank Test: Subjects with High CRT-scores

Wilcoxon Matched-Pairs Signed Rank				
	W+	*W-*	*p<=*	*N*
FA No RI	76	2	.001465***	12
FA RI	124	12	.002136***	16
No FA RI	102	3	.0006103***	14
No FA No RI	36	9	0,1289	9

Note, *=significant on the 10 % level, **=significant on 5% level and *** = significant on 1 % level.

Appendix C. Experimental Procedures

Pilots: Two sets of pilot sessions were conducted. One set, using the price choice environment, in Stockholm with 46 subjects over 6 sessions. The other set was run in Charlottesville using the quantity choice environment with 24 subjects in 2 sessions. The results of these experiments were used to calibrate the price and cost parameters and payoff conversion factors. The language in the instructions and the quiz questions were also improved based on feedback from the subjects in the pilots.

Recruiting: Subjects were recruited among undergraduate and graduate students at the University of Virginia in Charlottesville and at the Royal Institute of Technology in Stockholm. Each session contained 12 subjects save one session with 11 subjects and one with 13. For the price choice environment, we recruited 24 students for several time slots and twice ran two sessions at once. For the relative information environment, it was imperative that all participants in the session were present at the same time (so that in every session they were learning about 11 other people's decisions). In the environment with no relative information, there is no interaction with any other subjects in the session, so we sometimes split the participants from any given session between time slots.

Laboratory Conditions: Each participant had their own computer and we placed them in the room so that that they could not see the screens of other participants.

Experimental instructions: The instructions were read out loud and the participants read along on their screens. The instructions contained a quiz (see Appendix D) that tested comprehension of the experimental procedures and payoff determination. Subjects had time to work through the quiz individually and the quiz was then read out loud again, followed by explanations of the correct answers. Participants were allowed to ask questions at any time, and if the questions were about the procedures the answer were addressed to the entire group. Please refer to Appendix D for the exact instructions.

Experimental Procedures: The experimental software was set up so that everyone in a session moved at the same pace; everyone had to finish the first round before anyone could move onto the second round and so on. This was important in the relative information treatment because we wanted each subject to receive information from all other subjects in the session before they could

move on to the next round. It also encouraged careful deliberation, since there was no reward for racing through the procedure. For the sessions with no relative information, subjects also moved at the same pace. This worked well because they all began the post-experiment survey at the same time, and they finished around the same time which made payment more convenient. If individuals were moving too slowly, a general announcement that we were waiting for "one more decision" was made, but we did not pressure anyone individually, as we did not want people to feel rushed or perceive peer pressure in their decision making process.

Cognitive Survey: In Sweden we had students take a cognitive survey after the completion of the experiment. We told subjects that they had a survey to fill out, but were careful not to explain the motivation of the survey so as not to bias the results.

Payment: After subjects finished the cognitive survey, we called each person to the front of the room individually and paid them privately.

Appendix D. Instructions

These instructions were read out loud to subjects. Below the instructions for the price choice environment are shown as they appeared on subjects computer screens. Instructions for the quantity choice environment are available on request.

Instructions (ID =), Page 1 of 6

- **Overview**: This is a market simulation in which you will be given some information about costs and production capacity, and you will then make price decisions. Your task is to use the cost information to make decisions that determine your earnings, and as a reward, we will pay you in cash, immediately afterwards, an amount that depends directly on your total earnings in this simulated market.

- **Rounds**: There will be a series of market periods or "**rounds**."

- **Earnings:** The decisions that you make will determine your earnings. The computer will calculate your earnings in terms of "experiment dollars." After you finish the final round, you will be paid in real money by the organizers, based on your total earnings in the experiment.

- **Price Decisions:** You will be a seller in a market, and you will choose a price that affects how much you sell.

- **Production Capacity:** In each round, you will have the capacity to produce up to 3 units of a product.

- **Costs:** As explained on the next page, each product unit that you produce and sell in a round will cost you a number of experiment dollars. (Even if you have the capacity to produce 3 units of output, the actual number of units that you produce and sell will depend on the price that you charge and on going market price, as explained later.) Some of these costs may change randomly from one round to the next, and you will be shown these costs before choosing a price.

Instructions (ID =), Page 2 of 6

- **Inputs Used:** To be able to produce and sell units in the market, you will need 2 types of inputs.

- **Input A Cost:** Each additional product unit you sell in a round requires that you purchase another unit of Input A, as shown in the table below. For example, if you sell 1 product unit, the associated cost of the input A would be $*.**, if you sell a second unit the additional cost would be $*.**, for a total of $*.** +

$*.**, etc. These costs will be randomly changing from one round to the next.

Input A Costs:	**Unit 1**	**Unit 2**	**Unit 3**
	$*.	**$*.**	**$*.**

- **Input B Requirements:** In addition to the costs for input A, a regulation requires that you have one unit of input B for each product unit that you sell. Input B can be bought in a market at a known price of **$5.00**.

Instructions (ID =), Page 3 of 6

Price and Sales Quantity: In each round, you will choose a price for each unit. The price must be between **$1.00** and **$10.00**. The actual number of product units that you sell will depend on your price and on the (randomly determined) market conditions.

Market Price: The computer will randomly determine a "going market price" between **$1.00** and **$10.00**, with all amounts in this interval being equally likely. You will not sell a unit if this market price is lower than the one you select for that unit, but if your price is lower, you will sell the unit **and you will receive the going market price**.

Your Price Choice: You may choose different prices for each of the 3 units that you have the capacity to produce. The price you choose should represent the least you would be willing to accept for selling that particular unit. The amount that you receive for a sale **will be the market price**, which generally will be higher than the price you

offered.

Non-decreasing Prices: Your price for unit 2 must be at least as high as your price for unit 1, your price for unit 3 must be at least as high as your price for unit 2, etc.

Instructions (ID =), Page 4 of 6

- **Earnings:** Your profit or earnings for a round is the difference between your revenues and costs.

- **Costs:** You will have costs from the purchase of inputs in each round.

- **Revenues:** Money inflows result from sales of units of the product.

- **Total Earnings:** The program will keep track of your total (cumulative) earnings. Earnings can be negative if revenues do not cover costs. Earnings can be positive if you make some profitable sales. Positive earnings in a round will be added, and negative earnings will be subtracted.

- **Working Capital:** You begin with an initial money balance, **$20.00**, so that gains will be added to this amount, and losses will be subtracted. This initial working capital will show up in your total earnings at the start of round 1.

Instructions (ID =), Page 5 of 6

Input A Prices: $2 for unit 1 $2 for unit 2 $5 for unit 3

Please answer the questions below to ensure that you understand key aspects of the payoff structure.

Question 1: You have 3 capacity units, and if you end up selling fewer than 3 units of output, you incur the input A cost

☐ a) only for those capacity units actually used in production.

☐ b) for all 3 capacity units, whether or not they are used in a given round.

Question 2: You are endowed with 0 input B. If you produce fewer units of output, the excess (unused) input B will be

○ a) automatically "banked" for future use.

○ b) automatically sold for you at $5 each.

Question 3: Suppose that you decide that you are willing to sell your first unit for any price above $P and you choose a price of exactly $P for that unit. If the market price turns out to be $10.00, which is higher than your price for the unit, then

○ a) you sell the unit and receive $P for it.

○ b) you sell the unit and receive $10.00 for it.

Instructions (ID =), Page 6 of 6

Input A Prices: **$2 for unit 1** **$2 for unit 2** **$5 for unit 3**

Question 1: You have 3 capacity units, and if you end up selling fewer than 3 units of output, you incur the input A cost

◉ **(a) only for those capacity units actually used in production.**

○ (b) for all 3 capacity units, whether or not they are used in a given round.

Your answer, **(a)** is **Correct**. You only incur input A costs for units actually produced.

Question 2: You are endowed with 0 input B. If you produce fewer units of output,

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the excess (unused) input B will be

○ (a) automatically "banked" for future use.

◉ **(b) automatically sold for you at $5 each.**

Your answer, **(b)** is **Correct**. Input B cannot be banked; unused input B will be sold automatically and the revenues will be returned to you.

Question 3: Suppose that you decide that you are willing to sell your first unit for any price above $P and you choose a price of exactly $P for that unit. If the market price turns out to be $10.00, which is higher than your price for the unit, then

○ (a) you sell the unit and receive $P for it.

◉ **(b) you sell the unit and receive $10.00 for it.**

Your answer, **(b)** is **Correct**. If you sell a unit, you receive the market price, so there is no risk to you of choosing the lowest price for which you are willing to sell a unit. But if you price below the least you are willing to accept, then you run the risk of selling at a price that is too low for you if the market price turns out to be too low.

Instructions Summary (ID =)

- There will be a number of market periods or "rounds."

- You will begin each one by finding out some information about your costs

- Then you will choose prices for each of your 3 capacity units.

- The going market price will be randomly determined and will between **$1.00** and **$10.00**.

- If your price for a unit is less than or equal to the market price, you will sell the unit **but the price you receive is the market price**.

- Thus your price is like a "limit order" that need only be lowered enough to

"meet the market," so you should choose a price for each unit that represents the lowest you would be willing to sell that unit for.

- For each product unit that you sell, you will have a cost for the input A, and in addition, you must use a unit of input B.

- Input A costs may be different for each unit produced. You only have to pay these costs for units that you actually end up selling in the round.

- You begin each round with no units of input B. Units of this input that you use must be purchased. Purchases of units of input B will be at the same "market" price for units of input B.

- Earnings are revenues from sales of the product and sales of unused units of input B (if any), minus what you spent on units of input A for each product unit sold, and minus the costs (if any) of needed units of input B.

- **Special Earnings Announcement:** Your cash earnings in Swedish kronor will be determined at the end of the experiment by multiplying your total earnings in "experiment dollars" by 4. Thus one experiment dollar equals 4 Swedish krona. You will be paid in cash today immediately after you fill out the receipt form.

Paper III

A Ten-Year Rule to guide the allocation of EU emission allowances

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Abstract

Member States in the European Union (EU) are responsible for National Allocation Plans governing the initial distribution of emission allowances in the CO_2 Emission Trading System, including rules governing allocations to installations that close and to new entrants. The European Commission has provided guidelines to discourage the use of allocation methodologies that provide incentives affecting firms' compliance behavior, for example by rewarding one type of compliance investment over another. We find that the treatment of closures and new entrants by Member States is inconsistent with the general guidelines provided by the EU. We propose stronger EU guidance regarding closures and new entrants, a more precise compensation criterion on which to justify free allocations, and a Ten-Year Rule as a component of future EU policy that can guide a transition from current practice to an approach that places greater weight on efficiency.
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Keywords: Emission trading; Closures; New entrants

1. Introduction

One of the most important and contentious issues in the design of an emission trading program is how to initially distribute the tradable emission allowances. This has certainly been true for the European Union emissions trading system (EU ETS), which began operation in January 2005. At play in the EU ETS is the creation and allocation of wealth in the form of tradable property rights with an annual value of approximately €38 billion (1.88 billion metric tons of CO_2 annually at a recent price just over €20). This value can be expected to grow substantially in coming years. Moreover, the way in which the emission allowances are distributed initially can provide incentives that affect economic behavior. This has important consequences for the costs of reaching emission targets, for the provision of compensation to incumbent installations and other parties, and for the stability of the trading program.

Annex III of the EU Directive lays out the rules of the trading system (European Union, 2004) and presents criteria for the initial distribution of emission allowances. A central feature of the EU guidelines is that ex post adjustments of allocations are disallowed. That is, regulators must decide prior to each trading period how many allowances will be allocated and to what installations, and the regulator cannot redistribute these allowances within the trading period. Hence, an approach that updates the allocation based on some measure or behavior within the same trading period is precluded. Allocation must be based on a "historic" measure. This ensures that the allocation does not provide incentives that would affect economic behavior. In addition, so as not to undermine the profitability of existing installations and to provide compensation for compliance costs, the EU rules specify that in the first trading period (2005–2007), only a small percentage of allowances may be auctioned, and at least 95% of allowances should be allocated for free. In the second period (2008–2012), the amount to be given away for free has to be at least 90%.[1]

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[1]This approach differs from that used in the CO_2 trading program targeting power plants in seven Northeastern states in the US, known as

Still, considerable freedom is left to the individual Member States to decide the magnitude of allowances to be allocated and how allowances should be distributed among participants in the trading scheme. Ultimately, each Member State develops its own National Allocation Plan (NAP),[2] which must be approved by the EU Commission ahead of each trading period.

Two of the most important issues in the NAPs are how to address installations[3] that close (closures) and how to treat new installations that enter the trading system (new entrants[4]) (Matthes et al., 2005). Both of these issues are connected to the short-run question of ex post adjustments to allocation within a period since decisions by Member States on these matters in fact do affect allocations within the trading period thereby undermining efficiency. We find that any approach that adjusts allocations based on the closure or entry will lead to private investment decisions that increase the social cost of the ETS.

These issues also relate directly to the overarching issue of how emission allocations are to change between periods. For instance, if an installation closes but continues to receive an allocation within a period, shall that allocation continue into the next period? And if a new installation begins operation, does it have access to a pool of free allowances? The EU Directive does not clarify how the transition from the status as "new entrant" into "existing installation" should take place. Ultimately, these questions lead one to ask whether free allocations based on a historic measure from the beginning of the century are to survive indefinitely as climate policy evolves, or should that measure be updated over time?

This paper analyzes how treatment of closures and new entrants should be handled in a cap and trade system, with a particular focus on the efficiency of the trading system and the inevitable connection to the changing of allocation rules between trading periods. In the EU guidelines, efficiency is balanced with the goal of compensation as represented by the core approach of free allocation of allowances to incumbent installations. However, the lack of a precise criterion on which to justify free allocation—specifically compensation for the loss of economic value due to the trading program—enables a number of self-

interested justifications to compete in the setting of policy by Member States.

We report how EU Member States have addressed closure and new entrants in their NAPs, and find the treatment is inconsistent with the general guidelines provided by EU. This treatment also presents an opportunity for strategic behavior among Member States. The investigation leads us to characterize a "Ten-Year Rule", which offers a way out of the dilemma of historic allocation and creates a framework to reconcile efficiency and compensation as the EU goes forward with its ETS.

2. The "Historic" dilemma

The limitation placed on use of an auction and the preclusion on adjusting allocations during the trading period has effectively forced Member States to allocate allowances for free to incumbent installations based on a historic measure of performance (often called "grandfathering"). In preparation for the second compliance period, the question will be reopened somewhat. However, the Commission has signaled its desire to continue to use base years for allocation that predate the first period of the EU ETS (Vis, 2005).

There is considerable discussion in the economics literature about the efficiency and equity disadvantages of free allocation of allowances compared to an auction.[5] One advantage of an auction is to raise revenue that can substitute for other taxes, thereby dramatically lowering the social cost of new regulation (Goulder et al., 1999). Secondly, in regulated markets, an auction tends to reduce the difference between price and marginal cost, again providing potentially dramatic cost savings (Burtraw et al., 2001; Parry, 2005).

Nonetheless, a historic approach with free allocation has three big advantages.[6] First, like an auction, the historic approach to allocation provides an underpinning of intertemporal consistency, which is a central consideration for efficiency. Most of our attention in this paper is directed toward how the treatment of entrants and closures undermines this consistency and the large efficiency consequences that result. A second advantage of the historical approach is that, unlike an auction, free allocation reduces resistance from industry to stringent targets. Experience with the US SO_2 program shows that the allocation of allowances at no cost to affected installations has been critical in gaining political acceptance for the emissions trading concept (Stavins, 1998; Ellerman, 2006).

Third, free distribution based on historic measures has a public policy rationale based on the desire to compensate

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incumbent installations that are affected by the regulation (Tietenberg, 2001; Harrison and Radov, 2002). Schultze (1977) argues that people feel that government should "do no direct harm" in creating new public policy. This rationale implies a specific amount of compensation proportional to the change in the economic value of installations due to the program. However, one should also note that CO_2 policies affect installations in various ways, creating winners as well as losers. Aggregated to the portfolio of a firm, the need for compensation so as to do no direct harm will be less than if one considers installations on an individual basis (Burtraw et al., 2006). Compensation could be provided through an allocation of an infinite stream of a share of allowances sufficient to achieve a specific net present value, or alternatively, the allocation could be front-loaded and decline over time so as to achieve the same value (Smith et al., 2002). Such reasoning figures prominently in finding a resolution to issues that attend a historic approach to distribution that we address—closures and new entrants—and in the proposed Ten-Year Rule to guide further evolution of the program.

From the standpoint of providing compensation, a crucial question is: How much is enough? The answer depends on how the policy affects the profitability of the firm, and that depends on the change in its revenues and costs. A cap and trade program provides installations with an increase in revenue that depends on the degree that producers can increase product prices to pass-through their increase in marginal costs.[7] An important but somewhat counter-intuitive aspect of emission trading is that when comparing free allocation based on historic measures with allocation through an auction, in a competitive market, the change in revenue does not vary. That is because the price of products depends on marginal cost, which does not vary between these approaches to allocation.[8]

The program increases the marginal cost of an installation in two ways. One is an increase in the resource cost that is incurred for compliance with the emissions cap.[9] The second is the regulatory cost embodied in using emission allowances.[10] Even if awarded initially for free, the allowances have value because they could be sold to other installations. Hence, they have an "opportunity cost" like other resource costs (labor, fuel, materials) that is reflected in marginal cost, but their opportunity cost does

not vary whether allowances are given away for free or sold in an auction initially.

In sum, in a competitive market, the allocation does not affect the product price or the marginal cost of an installation. However, the allocation directly and significantly affects the value of an installation, because the free allocation is a transfer of wealth to shareholders of the firm.

Numerous studies have found that free allocation of CO_2 allowances based on historic measures typically overcompensates firms by providing them with an increase in revenues that is greater than their increase in costs (Bovenberg and Goulder, 2001; Boemare and Quiron, 2001; Burtraw et al., 2002; House of Commons, 2005; Burtraw et al., 2006).[11] Overcompensation to firms when allowances are distributed based on historic measures has not emerged previously in emission trading programs because, compared to previous emission trading programs, the market for CO_2 is special due to the magnitude of the program and the size of the allowance market.

Fig. 1 provides a stylized illustration, with an aggregate linear marginal cost schedule for reducing emissions. Across the horizontal axis is the percent emission reduction to be achieved. This marginal compliance cost schedule illustrates that the first unit of emission reduction is virtually free, the next unit costs a little more, and so on. The marginal cost curve determines the price (P) of an emission allowance, because this represents the cost of achieving one more unit of reduction (or the savings from avoiding the last unit). The resource cost of reducing emissions up to a certain level is the sum of incremental costs, or as illustrated it is the triangle underneath the marginal cost schedule.

The US SO_2 trading program was intended to achieve roughly a 50% reduction in emissions. In that case, as indicated by the dashed lines in Fig. 1, the annual asset value of the federally created "intangible property right" embodied in SO_2 emission allowances (the rectangle) was roughly twice the resource cost of compliance (the triangle).[12] However, under a cap and trade program for CO_2 in the EU striving to achieve initially, say, 5% reduction in emissions from baseline, the industry must use emission allowances sufficient to cover 95% of its original emissions. The regulatory cost of emission allowances therefore is the shaded rectangle indicated by price (P) multiplied by the quantity of emission allowances. With linear marginal cost, the area of the regulatory cost

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[7]Another source of new revenue may be the sale of emission allowances to other installations, but this is just equal to the payment made by other installations. This mechanism rewards low-emitting installations on average at the expense of higher-emitting installations, leaving all installations neutral in the aggregate.

[8]This statement does not apply for types of allocation that depend on updated measures of performance rather than historic measures.

[9]Resource costs refer to costs of operation including fuel, labor, and capital including the cost of emission control strategies such as fuel switching and efficiency improvements.

[10]Regulatory costs refer to the opportunity cost of emission allowances, that is, the market value of those allowances.

[11]The same does not apply necessarily to other pollutants. The finding depends on pollutant, sector, and market structure (Burtraw and Palmer, 2003).

[12]The annual value of the allocation of SO_2 allowances in 2001 was about \$1.68 billion, based on a price of about \$188 per ton. Annual compliance costs are about \$1.33 billion (2005\$) (Carlson et al., 2000). One reason the difference was not an important issue at the time of authorizing legislation in 1990 is the electricity industry was regulated at that time. Allowances were valued at original cost (usually zero) in establishing electricity prices.
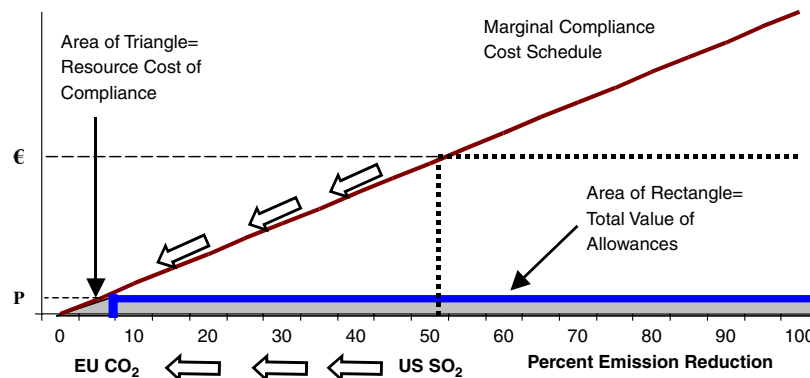
Fig. 1. Illustration of cost of compliance and value of allowances.

rectangle is 38 times the shaded area of the resource cost triangle.[13]

Free allocation based on historic measures creates a dilemma in the CO_2 program because the potential magnitude of overcompensation is enormous. The issue of allocation is more important under the CO_2 trading program than any previous emission trading program because the ratio of regulatory cost to resource cost is so much greater than in any previous program. Moreover, once launched, the inertia created by historic allocation may be difficult to reverse. For instance, in the new rule finalized in 2005 governing SO_2 emissions from power plants in the United States, known as the Clean Air Interstate Rule (CAIR), SO_2 emission allowances will continue to be awarded decades into the future according to a formula based on heat input at incumbent installations operating between 1985 and 1987.

The degree to which installations are compensated, or overcompensated, through free allocation is an empirical question that is beyond the scope of this paper, but the general question of compensation is relevant because it provides incentives that affect investment behavior for closures and new entrants.

3. Closures

The main argument why allocation should not change if an installation reduces economic activity or closes is that this preserves correct incentives for individual firms to consider private financial costs of resources that are equivalent to their social opportunity cost when making decisions about changes in economic activity, thereby minimizing overall social cost. An efficient decision will

result if the owner of an installation considers investment and operational decisions simply on the basis of opportunity costs—the marginal costs of production including the opportunity cost of using emission allowances. If it were profitable for the operator to close an installation and transfer or sell the allowances to a more efficient installation, this also would be the efficient solution and the intended effect of the trading scheme, everything else equal. The key feature is that the economic decision is not affected by or contingent on the award of compensation (free allowances).

In contrast, a policy that conditions the allocation of emission allowances on the continued economic operation of an installation has different incentive properties. The withdrawal of allocation based on reduced economic activity or closure makes the loss of the allocation into an additional opportunity cost affecting the production decision. In considering the marginal cost of operation, the firm will recognize that it receives the allocation *if and only if* it continues to operate. Consequently, the firm will not maximize its profits only with respect to the cost of production (including resource cost and the opportunity cost of allowances); in addition, it will take into account the value of the allowances that it will lose should it cease to produce output.[14] Imposing a condition that the allocation depends on continued operation of the installation transforms the allocation into a production subsidy.

The effect of the subsidy is illustrated in Table 1 drawing on industry and technology information from a simulation model of US power plants maintained by Resources for the Future. Each technology characterized in the table is expected to yield 2.85 million MWh of electricity generation per year. Consider a firm facing a choice about which of two existing electricity-generating installations to

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[13]The ratio of the regulatory cost (the allowance pool) relative to resource cost continues to increase up to about 15% emission reduction. Moreover, the marginal cost schedule is likely to be convex, making the case that allowance values are greater than resource costs a fortiori because at any given allowance price, the resource cost (area under the curve up to that price) is less than depicted by the triangle.

[14]Stavins (2005) finds similar perverse incentive properties in vintage differentiated conventional regulations, such as the US New Source Review Program. Gollop and Roberts (1985) find a similar effect in the electricity sector.

Table 1
Comparison of opportunity costs (€/MWh)

	Existing coal	Existing NGCC	New NGCC	New wind
Going forward operating cost	33	37	46	48
Emission rate (CO_2 metric tons/MWh)	1.05	0.49	0.38	0
Allowance burden	21	10	8	0
Going forward cost (Social opportunity cost)	54	47	53	48
Levelized allocation	21	10	8	0
Adjusted going forward cost (Private opportunity cost)	33	37	46	48

All monetary values are in 2005 Euros, calculated with an exchange rate of $1.23 = €1.00. Numbers may not add due to rounding. Calculations assume 2.85 million MWh per year of generation and an allowance price of €20/metric ton.

continue in operation and which to be shut down. One would expect the firm to continue to operate the installation with the lowest going forward operating cost, and the other installation would be closed. As an example, the choice between an existing pulverized coal installation and an existing natural gas combined cycle installation is illustrated in the first two data columns of Table 1.

The going forward operating cost of the existing pulverized coal installation includes short run variable costs and fixed operation and maintenance charges that total €33 per MWh. The plant produces 3.0 million metric tons of CO_2, which at an allowance price of €20 per ton yields a regulatory cost (allowance burden) of €21 per MWh. The going forward total cost of continued operation of this installation is €54 per MWh. In contrast, taking into account its specific costs and emission rates, the existing natural gas combined cycle plant has a going forward total cost of €47. The going forward cost as measured is society's total resource cost of electricity generation and hence it is equal to social opportunity cost.[15]

Imagine, however, that the allocation to an installation is withdrawn if an installation reduces operation. For illustration we assume the allocation is exactly equal to anticipated emissions. In this case the decision to close the pulverized coal installation implies the loss of allowances worth €60 million per year, or €21 per MWh. Accounting for the (private) opportunity cost of losing the allocation, the adjusted going forward total cost is €33 per MWh, which differs from the social cost. Meanwhile, the value of allocation to the natural gas plant is only €10 per MWh. Accounting for the (private) opportunity cost of losing the allocation at the natural gas plant, the adjusted going forward cost is €37 per MWh.

In other words, when the allocation is conditioned on the continued operation of a facility, the value of the allocation

behaves like a production subsidy, and is deducted from the overall operating costs to calculate the adjusted going forward cost. In this example, the subsidy to continued operation of the existing coal plant is greater than the subsidy to the existing natural gas plant so that the overall private cost of operation is lower for the coal plant. From the perspective of the firm, continued operation of the coal installation at €33 per MWh is the least cost option compared to the cost of the gas installation at €37, although the coal installation is not the least cost from a social perspective.

The example illustrates that withdrawal of the emission allocation alters the relative private going forward cost and can affect the investment and retirement decision.[16] Moreover, in this case, it leads to a private decision that does not align with social welfare, thereby undermining the efficiency goal of emission trading.

The third and forth columns of data in Table 1 present two new investment options. One is a new natural gas combined cycle plant with going forward total cost of €53 per MWh, and a wind installation with cost of €48 per MWh. Each of these technologies has a social cost that is lower than the cost of continued operation of the existing pulverized coal plant.

If allocation to existing installations is removed upon closure, then as noted above, the adjusted going forward total cost of the pulverized coal plant falls to €33 per MWh, which is below the cost of the new, more efficient technologies. Hence, we find that withdrawal of the emission allocation affects not only the relative comparison of existing installations, but it also affects the comparison with new installations. This explains why many observers (Betz et al., 2004; Harrison and Radov, 2002; Tietenberg, 2001) have argued that in the interest of economic efficiency the allocations should not be adjusted if an installation is closed. However, this has not been the strategy adopted by most Member States.

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[15]Social cost is the total of all the costs associated with an economic activity. It includes both costs borne by the economic agent and also all costs borne by society at large. It includes the costs reflected in the organization's production function and the costs external to the firm's private costs, for instance damages caused by emissions of CO_2. Thus, the example is equal to social cost under the assumption that the allowance price is a correct reflection of marginal damages caused by the emission. Under an emission cap, damages are constant and the allowance price represents avoided costs at other facilities (Burtraw et al., 2003).

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[16]An important assumption in this example is that the firm is able to pass through in electricity price 100% of its change in marginal cost. If the pass-through is less than this, then the value of the allocation subsidy to the firm is less than illustrated.

4. Current treatment of closures in the NAPs

Three approaches to the treatment of closures are applied by Member States currently.[17] By far the most common is that allocation is lost if an installation is closed. However, in a number of Member States the owner of a closed installation may transfer the allowances to a new installation instead of losing them altogether. A third option is to leave the allocation unaffected, but this has only limited application.

In Germany, an installation that is closed will receive no allowances the following year. Closure is defined as when an installation emits less than 10% of its average annual baseline emissions. Further, if an installation emits less than 60% of its average annual emissions, the quantity of allowances will be reduced by the same proportion as the reduction in utilization of capacity compared to the reference period. Thus, an installation has an incentive to keep operating *and* to emit a certain volume of CO_2 compared to the reference period. If applied strictly, this punishes both adjustments in production and mitigation measures such as switching from coal to biomass.

Also in Germany, any allowances recalled or not issued will be placed in the new entrant reserve. If the operator of the decommissioned installation commissions a new installation in Germany within 3 months, producing comparable products, the allocated annual allowances of the old installation can be transferred to the new installation. The 3-month deadline may be extended up to 2 years under special circumstances. Similar transfer rules exist in Italy, Austria, and Poland.

In contrast, in Finland and Spain, the transfer of allowances is not allowed. If an installation is closed, it will lose its greenhouse gas emissions permit and consequently lose subsequent allocations.

Sweden and the Netherlands are the only Member States that apply the policy of letting operators keep the allocation in the case of closure. However, emission allowances are only allocated for one trading period at a time (Germany has introduced an exception to this, see below). That is, the operator of a closed installation may find himself without any allocation in the next trading period, should the regulators decide to update the allocation scheme in this way. Hence, in reality, the difference between the policies of Sweden and the Netherlands and other Member States may be small.

Finally, the above examples of how closures are treated in the EU ETS illustrate the point that "closure" is just an extreme form of output variation. Thus, the distinction is vague between what is considered ex post adjustment or updating and thereby disallowed, and what is considered a legitimate withdrawal of allocation to closed installations under EU regulation.

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[17] All National Allocation Plans, as well as EU Commission decisions on them can be found and downloaded from the official EU website, http://europa.eu.int/comm/environment/climat/emission_plans.htm.

5. Discussion: treatment of closures

We have shown there is a strong efficiency case to be made against withdrawing allocations after closures of installations because such a policy creates an inefficient subsidy for the continued operation of existing installations. Nonetheless, in the political sphere, one finds prominent arguments that appear persuasive for withdrawing allocation to installations that no longer operate. One argument is that allocation of a stream of allowances in perpetuity contradicts fairness and common sense, from the perspective of some observers. The argument against permanent allocation is an easy one to make in the face of a question such as: "Why give allowances to somebody who does not need them?" This is an especially potent argument if it appears an installation would have shut down "anyway" in the absence of the program. Indeed, should allocation not be removed but remain unchanged indefinitely, it is possible that at some point all allowances are allocated to the owners of installations that no longer operate. As this paper shows, this is not a problem from an efficiency point of view, but advocates argue this means the possibilities for authorities to use allocation in order to drive changes in behavior, to compensate for losses, or to give rewards to desirable behavior are lost. In most of the NAPs that we have analyzed, Member States have withheld, or require transfers of allowances from closed installations. The efficiency argument appears counterintuitive in the context of a desire to treat participants in a trading scheme in what may appear to be a fair manner by avoiding the award of a perpetual property right to closed installations. This provides motivation for practical limits to the indefinite allocation of allowances to closed installations. In the next section, we will examine how the efficiency and fairness issues associated with closures interact with a related issue—the treatment of new entrants.

6. New entrants

Just as in the case of closures, an allocation to new entrants upsets the symmetry between the private opportunity cost of the firm and social opportunity cost. The allocation represents a production subsidy that lowers the entrant's cost. The subsidy clearly will lead to more production; the question is whether it leads to a more or less efficient outcome. Unfortunately, the answer depends on the manner in which emission allowances are distributed to other facilities. In general, however, allocation to new entrants can only be justified on the basis of other imperfections in markets or in the design of the emission trading system. Inevitably, even with such justification, allocation to new entrants takes the program further away from an efficient design.

Under competitive conditions, we argued previously, the allocation to existing sources based on a historic measure does not affect the opportunity costs going forward of an

installation as long as the allocation is not affected by the decision to continue operation. The symmetry between private and social opportunity costs is reflected in Table 1 by comparing the rows indicating going forward cost from a social perspective with the private perspective. If the existing coal plant receives an allocation whether or not it continues to operate (a point we revisit in a moment), then its private going forward total cost is equal to the social cost. Moreover, if there is no allocation to a new facility so that its private cost equals social cost, then the firm will choose to invest in the new wind installation with a going forward (social) cost of €48. The symmetry between social and private cost preserves the desirable efficiency properties of a market-based emission trading system—e.g., by minimizing private cost, the firm's decision serves to minimize social cost.

However, an allocation conditional on new economic activity (entry) introduces asymmetry. Let us assume the allocation to new entrants is based on expected future emissions expressed as fuel-specific benchmarks. The natural gas plant would receive an allocation worth €8 per MWh. The wind plant receives no allocation since its expected emissions are zero. As a consequence of this policy, the new natural gas plant has an adjusted (private) going forward total cost of €46 per MWh, which is less than the new wind plant. From the private perspective of the firm, the natural gas plant is the least cost option for new productive investment, although the wind plant is the least cost option from a social perspective.

The outcome of free allocation to new entrants leads to further unintended consequences when we compare the opportunity cost of new investment with continued operation of existing installations. Looking across all the options for electricity generation in Table 1 illustrates the case in which new wind is the least cost option for incremental generation. On the other hand, the new natural gas plant is more expensive than new wind or the existing natural gas plant. However, with the allocation to new entrants, the new natural gas plant dominates the other options from the private perspective of the firm.

A caveat is that these examples assume no special treatment of closures and that allocations were not withdrawn from facilities that close. However, as noted in Section 4, most Member States in fact do adjust the allocation to existing installations that decide not to operate. In this case, when considering the treatment of new entrants, we should do so in the context of pre-existing imperfections that create asymmetry between social and private costs for existing facilities.

The decision to withdraw emission allowances from an installation that closes alters the economic equation, placing incrementally more advantage to keeping the installation in operation. Section 3 illustrated that if new installations do not receive an allocation, then they are indeed at an economic disadvantage in the context of marginal retirement and investment decisions. Given that most Member States do adjust the allocation to existing

installations, one can ask: Is an efficient set of incentives preserved if allowances are simultaneously awarded to new installations? If so, then there could be two efficient policy equilibria—one with no adjustments in the case of closures and new entrants, another with adjustments—and the EU would be faced with the relatively simple problem of coordinating the choice of equilibrium.

Drawing on the example in Table 1, we know that an allocation based on a fuel-specific benchmarks reflecting their emission burden reverses the private cost ordering between the new natural gas and new wind facilities. Therefore, let us consider a uniform allocation to new entrants that does not depend on fuel type. Let us assume that wind also receives an allocation per MWh that has equivalent value to that for a new natural gas plant. Wind would have a cost of €48–€8 = €40 per MWh. Wind would remain the least cost option among new technologies, preserving an efficient decision when considering the new investment options.

Unfortunately, this strategy does not preserve the ordering between new and existing technologies. Given that allowances are withdrawn from facilities that close, the existing coal plant remains the least cost option from the firm's private perspective, at €33 per MWh. In fact, the existing natural gas facility also appears to be less costly than the wind plant from the private perspective, even though it is not from the social perspective.

It is apparent that the only way to preserve the relative ranking between new and existing installations while adopting special rules for closures and new entrants is if the value of withdrawals from installations that close and awards to new installations are equivalent. This observation suggests a strong condition: Adjustments for retirement and investment decisions should have the same value per MWh of production in order not to undermine the efficiency of the ETS. One approach to maintain the symmetry between private cost ordering of cost and the social cost ordering among new and existing installations would be to withdraw allocations upon closure and to award to new entrants based on a common, fuel-neutral benchmark.

The transfer rule used by many Member States and discussed in the context of closures—that is, the withdrawal of allowances from an installation that closes unless the allowances are transferred to a new installation operating within the same country—is one approach that approximates this prescription. The transfer rule leaves the allocation unaffected, as long as (a) the owner of an installation executes a transfer upon closure of the original installation, (b) the withdrawal from the retiring facility is equal to the allocation to the new facility, and (c) the transfer occurs between a closing and new installation within the same firm. However, the transfer rule provides little comfort to new installations owned by new investors who are different from the owners of incumbent facilities. Bode et al. (2005) have argued that this transfer rule discriminates against new investors and causes large profits for incumbent generators.

Taking a broader viewpoint, however, even a uniform allocation for closures and new entrants will not lead to an efficient outcome; it just appears more efficient than an asymmetric approach. The uniform approach remains inefficient because it invites strategic behavior over the operation of the *full set* of productive installations. For example, consider a firm with an inefficient installation that could be shut down because of sufficient capacity at other installations, but the firm would have its allocation adjusted if the installation were to close. If the firm were to shift production from an existing efficient installation to the inefficient one, it would incur higher costs of production but it would also avoid the adjustment to allocation. Consequently, any adjustments to allocation on the basis of closure, or similarly for new entry, will inevitably create potentially important inefficiencies in the trading system.

7. Current treatment of new entrants in the NAPs

Treatment of new entrants is one of the areas where policies among the Member States differ the most. The EU Commission only asks Member States to describe how new entrants can gain access to emission allowances. There are no rules on whether or not new entrants should be allocated free allowances. Still, *all* member states guarantee a certain volume of allowances will be available to new entrants at no cost, by setting aside allowances reserved specifically for new entrants. Allowances from these reserves are usually provided on a first-come, first-served basis.[18]

The most common methodology is to base allocation to new entrants on forecast activity levels and emission rates, specified by sector or product type, and forecasted activity. However, benchmarks differ significantly across Member States, even for identical products such as heat or power.

When sector-wide benchmarks cannot be defined, Member States often refer to Best Available Technology (BAT) as the benchmark to be used. The emission factors can be specific to an installation, or common for an entire sector. The latter is mainly used in the energy sectors, but for instance Italy also applies sector-wide benchmarks in the mineral and ceramic industries.

The definition of BAT also varies across Member States. Some Member States refer to existing official EU studies (for instance the BAT reference documents from the Joint Research Center, 2005). Others refer to national legislation or to the IPPC directive (European Union, 1996), which allows BAT to be defined on a case-by-case basis. In Sweden, for example, BAT is to be defined in accordance to environmental law on a case-by-case basis. For energy installations, only combined heat and power plants (CHP) are eligible for allocation from the new entrant reserve. A benchmark based on a fuel mix containing significant shares of renewables is used together with forecasts of generation in order to calculate the allocation. In contrast, Poland does not specify how BAT will be established.

Once benchmarks are selected, they are multiplied by a level of activity (e.g., output) to arrive at an allocation for sources. The most common method for estimating activity levels for new entrants is to use a forecast of future production. There are significant differences among Member States in how the forecasts are estimated and production calculated. In some cases (e.g., Sweden, Poland) allocation is based on production forecasts specific to the new installation. In other Member States (e.g., Denmark, Finland, Austria, and Italy), allocation is based on the size of the new installation, expressed as installed capacity, and general assumptions on utilization rates for specific technologies. However, even among these general methods, there are differences. For example, Finland and Denmark, whose energy systems to a large extent are integrated, use different utilization rates when calculating the allocation to new entrants.

This construction creates a range of potential problems. Basing allocation on forecasts provides an incentive for firms to exaggerate forecasts of future production. Since ex post adjustments are not allowed in the EU ETS, the only possibility for regulators to police incorrect forecasts is to update the allocation between trading periods. However, as will be discussed later in the paper, this has the potential to distort operational decisions made by firms.

Finally, there is at least one example of a Member State that explicitly guarantees an allocation to new entrants for a significant number of years. Germany guarantees free allowances to a new entrant for up to 14 years if it is a completely new installation. If the new installation is replacing an old installation and allowances are transferred from the old installation, allocation may be guaranteed for up to 18 years. It is not clear if this guarantee of property rights will be compatible with the revision of the EU directive on the ETS that will take place in 2006 and most probably again in 2012, where significant changes in allocation methodology at the EU level might be mandated.

8. Discussion: treatment of new entrants

In theory, to achieve efficiency goals, new entrants should not receive an allocation because this preserves the symmetry between private and social opportunity cost. However, when withdrawal of allocation is made for closures then making an allocation to new entrants can preserve the symmetry in private opportunity costs between retiring and new installations. The best way to do that is likely to be a uniform adjustment to closures and new entrants that does not vary with fuel. However, we find that even this approach is laden with the unintended

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[18]A limited number of Member States (e.g., **Poland** and **Italy**) also plan to purchase allowances from the market for new entrants if their new entrant reserves are oversubscribed.

consequence of providing an incentive to shift production among installations. The only approach that preserves efficiency is to make no adjustments to closures or new entrants.

Nonetheless every Member State chooses to allocate to new entrants. One reason is the perception that it is "unfair" to treat incumbent and new entrants differently. The notion of fairness in this context is poorly defined, since the primary rationale for free allocation is to provide compensation to incumbent installations with sunk capital investments that lose value under the program. Since new entrants have no sunk value, there is no basis on which to make a comparison of fairness.

A second reason that Member States choose to allocate to new entrants may be a more specific notion of fairness in the access to capital. Capital markets discriminate in the price they charge firms for acquiring new capital in response to observable accounting measures such as debt, liquidity, and cash flow and also due to uncertainties such as exposure to price volatility in factor inputs, including emission allowances. Since the cost of capital varies with the amount of capital needed, the free allowances reduce the need of the firm to go to the bank to borrow money to buy allowances, placing its cost of capital on a par with incumbent installations.

Given that all 25 Member States have set up provisions to give allowances to new sources, allocation to new entrants is clearly a political priority. Hence, we propose that new entrant reserves should be fuel neutral, and adjustments for retirement and investment decisions should have the same value per unit of production in order to reduce the extent to which it undermines the efficiency of the ETS.

9. Strategic behavior among member states

The preceding discussion examined closures and new entrants within a single domestic system. Decisions about allocations, in particular those on closures and new entrants, also involve considerations about national competitiveness. The government of a Member State is faced with incentives that lead to a decision that is not the efficient solution for the trading program as a whole—the well-known "prisoner's dilemma."

For instance, when a Member State decides on the rules for closures, it may consider the tax base and the job opportunities that installations provide. Thus, it may seem rational from a Member State point of view to take allowances away from installations that close, or at least condition the allocation on a transfer to a new installation in the country, in order to create incentives for continued production in one's own country. Similarly, in the interest of attracting new investments that hopefully will result in a larger tax base and increasing job opportunities for its own citizens, it is rational from a Member State perspective to be generous to new entrants, regardless of the effect on the system as a whole.

Consider the case of the energy sector regulations in two neighboring Member States connected by electricity transmission lines, for instance Denmark and Germany or Sweden and Finland. These Member States allocate emission allowances to new entrants, based on emission factors that differ significantly. This means allocation to a new entrant will be more favorable in one Member State than another.

Similarly, imagine two Member States where one imposes a greater penalty for closure than the other. If a firm was otherwise indifferent between closing an installation in one of these countries, it would choose to keep open the installation in the country with the greater closure penalty. The setting creates a contest between Member States to provide favorable incentives to retain and attract investment. The ultimate outcome may be economically inefficient and politically undesirable.

Given the EU-wide efficiency issues, the efficiency goal would be best served through the regulation of closures at the EU level. We suggest Member States should be required to let installations keep their allocation even in the event of closure, but if adjustments are made they should be imposed on a consistent system-wide basis. Further, if existing installations are no longer penalized for shutting down, this removes the subsidy for continued operation of these sources, which negates the best justification for providing allowances to new sources.

Moreover, allocation to new sources is universal in the Member States already. To enable such allocation while minimizing distortions away from economic efficiency in the short-run, the EU should consider a uniform allocation rule to all new sources that is fuel and technology neutral. This preserves efficient trade-offs among new sources and negates competition among Member States on the basis of allocation to new sources, but it does not resolve the efficiency issues that arise in the treatment of new sources compared to existing sources.

As we have noted, differential treatment of incumbents and new entrants creates a troublesome perception of unfairness. This perception is enabled by the failure to articulate a criterion for free allocation, specifically, that free allocation provides compensation to existing installations for lost economic value. This criterion would help in the development of guidelines to design and limit free allocation, and protect against strategic behavior by firms or Member States in attempts to capture rent or benefit from the allocation rules. In the long run, we propose a transition strategy for the EU to achieve reform, guard against strategic behavior, and place improved efficiency.

10. The Ten-Year Rule: a framework for moving forward

Adjustment for closures and new entrants is closely related to the issue of updating allowance allocations over time, but the Commission has chosen to view the issues differently. In general, the economics literature finds that
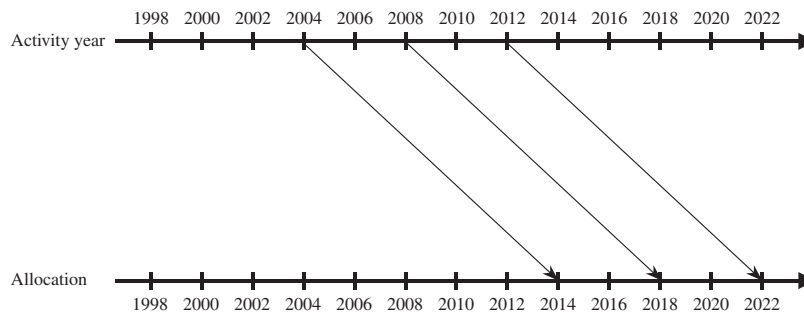
Fig. 2. Schematic illustration of the Ten-Year Rule.

an updating approach serves as an economically inefficient subsidy for production and lowers product prices for consumers.[19] These considerations have guided the Commission's prohibition on updating in the Member States' general allocation formula while guidelines allow for adjustments on the basis of closure and entry within a budget period. Nevertheless, in the general allocation formula, it is not clear whether the initial base year must be kept forever and whether sources should continue to receive allowances in perpetuity, as has been the case with the SO_2 trading system in the United States.

With a structure that requires new allocation plans for each new 5-year budget period, some sort of change in allocations seems inevitable in the EU system. Therefore, we propose a Ten-Year Rule to guide a transition to a more consistent and efficient approach to the treatment of closures and new entrants. The transition rule we propose resembles an updating approach, as does the current treatment of closures and new entrants. However, the rule we propose has a sufficiently long lag between allocations and the activity year upon which they are based that the allocation retains much of the incentive structure of a historic approach.

Under a Ten-Year Rule, installations would receive an allocation in a given year based on activities dating 10 years previous. Unlike the SO_2 program in the US, the activity year on which the allocation is based rolls forward over time. To illustrate, imagine that a Member State allocates to installations based on the average emissions from three reference years 2000–2002. Until 2011, existing installations would receive allocations based on this average, but in 2012, the allocation would be based on the average of 2001–2003, and so forth. Fig. 2 illustrates the concept of the Ten-Year Rule schematically. Averaging over several years, or whether allocation is based on emissions, output or input will not alter the concept or change the functioning of the rule.

This approach has the disadvantage that it provides an inefficient price signal. However, the ten-year lag weakens significantly the tendency of updating to produce unin-

tended incentives for actors on the market. The net present value in year b of an allowance with price (P) in year n is given by the formula $NPV_b = (1/1 + r)^{(n-b)} P_n$, where r is the real rate of financial discounting. The allocation decision in year n is based on behavior in year b. The previous literature has examined a relatively short lag of 0–3 years in estimating the production subsidy that is implicit in an updating approach. However, a discount rate of 10% with the lag inherent in the Ten-Year Rule would reduce the production subsidy by 60%, diminishing significantly the impact of updating on the behavior of firms.

The Ten-Year Rule would provide a remedy to the conundrum of how to treat existing sources that reduce economic activity or close. However, to implement this rule, we foresee a need for greater guidance at the EU level to coordinate consistent policies by the Member States. Under the Ten-Year Rule, if an installation reduces production or closes, it would continue to receive an annual allocation until it no longer had economic activity in the updated base period 10 years prior. In other words, if an existing installation is shut down, it would continue to receive allowances for 10 years, thereby diminishing incentives to continue operation. However, eventually the group of installations receiving allowances would shift, as sources that shutdown would eventually stop receiving allowances.

The treatment of new sources would have parallel structure. Initially, new sources would receive allowances from a reserve according to an emission rate benchmark that could be standardized across industry sectors in Europe. Preferably, allocations for these installations would be based on installed capacity and standard utilization rates to avoid gaming and inconsistencies associated with case-by-case projections of expected utilization. The emission rate benchmark would remain constant for all new installations, and they would continue to receive an allocation on this basis for 10 years before allocations would begin to update based on a rolling measure of economic activity similar that of existing sources. For instance, an installation starting operation in 2005 would receive allocation based on forecasts until

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[19]Fisher, 2001; Burtraw et al., 2001; Haites, 2003.

2014. From 2015 onwards, allocation would be based on actual activity 10 years prior.[20]

The Ten-Year Rule is less of a specific policy recommendation than a metaphor for how to go about improving the approach to allocation in the ETS. The implementation of a transition rule could take a variety of forms. Allocation could be based on activities with a 10-year time lag or activities averaged over several years. The advantage of a 10-year framework is that the time period conveniently extends over two 5-year budget periods. Allocation could be based on emissions, output or energy input, none of which would alter the fundamental concept that an evolution in the approach to allocation is needed. We feel an incremental transition has virtue compared to a sudden change in the design of the trading system. Even though allocations for future periods have not been announced and have no legal standing, an incremental approach helps build confidence that property rights in the nascent trading program will be respected. An incremental transition represented by the Ten-Year Rule can accommodate various expectations about allocation that have begun to shape investment behavior and it can allow for those expectations to evolve. However, even more importantly, a transition rule provides a way to move toward an overall approach to allocation that is more efficient.

The Ten-Year Rule is designed to address the intertemporal inconsistency in the current practice for retiring and new sources. However, the EU may do well to ask whether incumbent sources should receive all of their allowances for free for their entire operational life, if not beyond that life. Since the existing fleet of CO_2-emitting installations is likely to be long-lived, the most significant characteristic of the Ten-Year Rule would be how it treats currently existing sources.

The Ten-Year Rule provides an opportunity to transition away from free allocation of emission allowances for all existing sources and toward an expanded role for an auction. For example, after 10 years of free allocations, there might begin a transition period with a steadily increasing auction of allowances.[21] The premise for this recommendation is the efficiency disadvantage of free allocation. Provision of compensation is the only compelling justification for free allocation. Since free allocation has a tremendous efficiency cost, specific compensation goals should be developed to limit and target compensation and to guide the general allocation formula. Criteria

should address the level of compensation desired, and these criteria could then serve as the basis for a transition to a more efficient approach to allocation in general. The Ten-Year Rule provides an institutional vehicle for that transition.

11. Conclusion

The European Commission has discouraged Member States from adopting allocation methodologies that provide incentives to firms to alter their behavior in an attempt to adjust their allocation. In general, it is anticipated that such incentives would move behavior away from economic efficiency and thereby raise the cost of the emission trading program.

In this paper, we demonstrate that the negative effects of adjusting allocation become apparent when analyzing the treatment of new entrants and closures in the EU ETS. There is a strong case to be made against withdrawing allocations after closures of installations. Paradoxically, the policy of withdrawal of allowances serves as a production subsidy because the allocation is received if and only if the installation continues to operate. This production subsidy for inefficient installations that otherwise would close has efficiency costs for the ETS and the economy.

However, we observe that a purely historic approach to the initial distribution of allowances presents a political dilemma. Member State policies reveal a desire to limit the indefinite allocation of allowances to closed installations. In most of the NAPs that we have analyzed, Member States have decided to withhold or require transfers of allowances from closed installations.

Similar considerations are apparent in the treatment of new entrants. In order to preserve economic efficiency, new entrants should not receive an allocation because to do so provides a subsidy for production that is received if and only if the installation operates. Hence, the allocation lowers the going forward cost of new entrants and gives them an advantage relative to the going forward cost of incumbents.

However, Member States uniformly choose to allocate to new installations. There is an understandable desire to treat participants in a trading scheme in a manner that is perceived as fair, and the perception is that new entrants should receive the same allocation as existing installations even though rules to this effect have different incentive properties for these two types of installations. Were new installations not to receive an allocation, then ultimately the Member States would find itself in an awkward position with two classes of facilities, those original incumbents receiving an allocation and the entrants who would not. Also, capital markets anticipate a risk associated with the acquisition of allowances at an uncertain price in the future. New entrants would face a cost associated with that risk that is not shared by existing installations. This would be true even if the operator were

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[20]We suggested previously that the only way to preserve the relative ranking between new and existing installations while adopting special rules for closures and new entrants is if the value of withdrawals from installations that close and awards to new installations are equivalent. Strictly speaking, the Ten-Year Rule would recognize that there was no production 10 years previous and there should be no allocation to a new installation until the installation is 10 years old. However, our suggested approach is an improvement over current practice and offers a transition to a more efficient allocation in the future.

[21]Similar approaches have been featured in recent allowance allocation proposals in the United States. See, for example, US EPA (2002) and NCEP (2004).

to buy forward contracts for all the allowances needed over the life time of the investment, since the seller of such contracts would charge a price for the risk he assumes in place of the operator. Member States may therefore feel there is reason to provide a capital subsidy to promote new investment.

Policies on closures and new entrants interact and have a bearing on retirement and investment decisions. The decision to withdraw emission allowances from an installation that closes or to allocate to a new installation, in each case if and only if the installation is in operation, creates an advantage to operating the installation. Since Member States adjust allocation to closures, incumbent emitters would be subsidized over new entrants if new entrants also did not receive an allocation, and vice versa. Free allocation to new entrants may be justified, and certainly is understandable, as an effort to offset the incentives provided by withdrawal of allowances from installations that close. But as a consequence, the suite of policies currently in place in the EU has large detrimental effects on efficiency.

To obtain level incentives affecting production decisions, there is a need for symmetry in the regulation on closures and new entrants. To move in this direction requires stronger guidance from the EU; otherwise, strategic behavior among Member States can be expected. A reasonable option in the short run would be fuel-neutral benchmarking, e.g., withdrawal from closed installations and allocation to new installations based on the same measure such as a set number of allowances per unit output, regardless of fuel or technology. This could remedy the asymmetry between installations that otherwise might close and new entrants, but it does not remedy the asymmetry between these units and other incumbent facilities. One could expect activity levels among facilities to change as firms seek to capture the incremental allocation for closures and new entrants. Hence, any adjustments on the basis of closure or entry will have an efficiency cost, and we seek to provide a transition rule to guide the ETS in a more efficient direction.

We propose a Ten-Year Rule to guide a transition from the current set of existing policies to a level playing field for new entrants and closures. The rule would replace an indefinite linkage to a frozen base year as the basis for allocation with a linkage that steadily modernizes, but which preserves a sufficient time lag from current decisions (10 years) that incentives affecting current behavior would be affected slightly or not at all.

The Ten-Year Rule aims to capture the efficiency gains from a stable system of property rights. Yet it also addresses the perception of fairness by providing a finite horizon for the potentially infinitely lived property rights that could be created under historic allocation and that have been created in previous systems. Moreover, the literature indicates that historic allocation typically over-compensates installations for sunk costs. The need for compensation varies according to the change in the economic value of installations, not according to activity levels in any given year. The EU needs to articulate a precise compensation criterion as the rationale for free allocation, and then design and limit free allocation to meet this purpose. Following the compensation rationale, the portion of allowances distributed for free could decrease over time, which provides an opportunity to transition away from free allocation.

Acknowledgments

References

Betz, R., Eichhammer, W., Schleich, J., 2004. Designing national allocation plans for the EU emissions trading—a first analysis of the outcome. Energy & Environment 15 (3), 375–425.

Bode, S., Hubl, L., Schaffner, J., Twelemann, S., 2005. Discrimination against newcomers: impacts of the German emission trading regime on the electricity sector. Hamburgisches Welt-Wirtschafts-Archiv Discussion Paper 316, June.

Boemare, C., Quiron, P., 2001. Implementing greenhouse gas trading in Europe: lessons from economic theory and international experience. Report for the Interact project, DG Research of the EU Commission, Centre International de Recherche sur l'Environnment et le Developpement. www.centre-cired.fr.

Böhringer, C., Lange, A., 2004. Mission impossible!? On the harmonization of national allocation plans under the EU Emissions Trading Directive. Discussion Paper No. 04-15, ZEW Centre for European Economic Research.

Bovenberg, A.L., Goulder, L.H., 2001. Neutralizing the adverse industry impacts of CO_2 abatement policies: what does it cost? In: Carraro, C., Metcalf, G.E. (Eds.), Behavioral and Distributional Effects of Environmental Policy. University of Chicago Press, Chicago.

Burtraw, D., Palmer, K., 2003. Economic efficiency and distributional consequences of different approaches to NOx and SO2 allowance allocation. Prepared for the US Environmental Protection Agency. http://www.epa.gov/air/clearskies/econ.html (accessed June 8, 2005).

Burtraw, D., Palmer, K., Bharvirkar, R., Paul, A., 2001. The effect of allowance allocation on the cost of carbon emission trading. Resources for the Future Discussion Paper 01-30, August.

Burtraw, D., Palmer, K., Bharvirkar, R., Paul, A., 2002. The effect on asset values of the allocation of carbon dioxide emission allowances. The Electricity Journal 15 (5), 51–62.

Burtraw, D., Krupnick, A., Palmer, K., Paul, A., Toman, M., Bloyd, C., 2003. Ancillary benefits of reduced air pollution in the United States from moderate greenhouse gas mitigation policies in the electricity sector. Journal of Environmental Economics and Management 45 (3), 650–673.

Burtraw, D., Palmer, K., Kahn, D., 2006. CO_2 allowance allocation in the regional greenhouse gas initiative and the effect on electricity investors. The Electricity Journal 19 (2), 79–90 (March).

Carlson, C., Burtraw, D., Cropper, M., Palmer, K., 2000. SO2 control by electric utilities: what are the gains from trade? Journal of Political Economy 108 (6), 1292–1326.

Cramton, P., Kerr, S., 2002. Tradable carbon permit auctions: how and why to auction not grandfather. Energy Policy 30, 333–345.

Dinan, T., Rogers, D.L., 2002. Distributional effects of carbon allowances trading: how government decisions determine winners and losers. National Tax Journal 55 (2).

Ellerman, A.D., 2006. US experience with emissions trading: lessons for CO_2 emissions trading. In: Hansjürgens, B. (Ed.), Emissions Trading for Climate Policy: US and European Perspectives. Cambridge University Press, Cambridge.

European Union, 1996. Council Directive 96/61/EC of 24 September 1996 concerning integrated pollution prevention and control. Official Journal of the European Union L 257, 0026–0040 (10/10/1996).

European Union, 2004. Directive 2004/101/EC of the European Parliament and of the Council of 27 October 2004 amending Directive 2003/87/EC establishing a scheme for greenhouse gas allowance trading within the Community, in respect of the Kyoto Protocol s project mechanisms. Official Journal of the European Union L 338, 18–23 (13/11/2004).

Fischer, C., 2001. Rebating environmental policy revenues: output-based allocations and tradable performance standards. Resources for the future discussion paper 01–22 July.

Gollop, F.M., Roberts, M.J., 1985. Cost-minimizing regulation of sulfur emissions: regional gains in electric power. Review of Economics and Statistics 67, 81–90.

Goulder, L., Parry, I., Williams, R., Burtraw, D., 1999. The cost effectiveness of alternative instruments for environmental effectiveness in a second best setting. Journal of Public Economics 72 (3), 329–360.

Haites, E., 2003. Output-based allocation as a form of protection for internationally competitive industries. Climate Policy 3 (Suppl. 2), S29–S41.

Harrison, D., Radov, D., 2002. Evaluation of alternative initial allocation mechanisms in a European Union greenhouse Gas emissions allowance trading scheme. National Economic Research Associates, prepared for DG Environment, European Commission.

House of Commons, 2005. The international challenge of climate change: UK leadership in the G8 and EU. Environmental Audit Committee, Fourth Report of Session 2004–2005, HC 105 (Published 27 March), The Stationery Office Limited, London.

Joint Research Center, 2005. BREF documents. Can be downloaded from http://eippcb.jrc.es/pages/Fmembers.htm.

Matthes, F., Graichen, V., Repenning, J., 2005. The environmental effectiveness and economic efficiency of the European Union emissions trading scheme: structural aspects of allocation. Report to World Wildlife Federation, Öko-Institut, Berlin.

National Commission on Energy Policy (NCEP), 2004. Ending the Energy Stalemate: a Bipartisan Strategy to address America's energy challenges. National Commission on Energy Policy (NCEP), Washington, December, 2004.

Parry, I., 2005. Fiscal interactions and the costs of controlling pollution from electricity. Rand Journal of Economics 36 (4), 850–870.

Schultze, C.L., 1977. The Public Use of Private Interest. Brookings, Washington, DC.

Smith, A.E., Ross, M.T., Montgomery, W.D., 2002. Implications of Trading Implementation Design for Equity-Efficiency Trade-Offs in Carbon Permit Allocations. Charles River Associates, Washington, DC (December).

Stavins, R.N., 1998. What can we learn from the grand policy experiment? Lessons from SO2 allowance trading. Journal of Economic Perspectives 12 (3), 69–88.

Stavins, R.N., 2005. The Effect of Vintage-Differentiated Environmental Regulations. Discussion Paper 05-12, Resources for the Future, Washington, DC (March).

Tietenberg, T., 2001. The tradable permits approach to protecting the commons: what have we learned? Nota di Lavoro 36.2002, Fondazione ENI Enrico Mattei (FEEM).

US EPA, 2002. The Clear Skies Act, Technical Support Package, Section A: Program Elements. September 2002, p. A9, http://www.epa.gov/air/clearskies/tech_sectiona.pdf.

Vis, P., 2005. European Commission DG Environment. Presentation given at workshop on the EU ETS review, organized by Centre for European Policy Studies, Brussels, April 13, 2005.

Paper IV

The Impact of the EU Emissions Trading System on CO_2 Intensity in Electricity Generation

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Abstract

Prior to the launch of the EU Emissions Trading System (EU ETS) in 2005, the electricity sector was widely proclaimed to have more low-cost emission abatement opportunities than other sectors. If this were true, effects of the EU ETS on carbon dioxide (CO_2) emissions would likely be visible in the electricity sector. Our study looks at the effect of the price of emission allowances (EUA) on CO_2 emissions from Swedish electricity generation, using an econometric time series analysis for the period 2004–2008. We control for effects of other input prices and hydropower reservoir levels. Our results do not indicate any link between the price of EUA and the CO_2 emissions of Swedish electricity production. A number of reasons may explain this result and we conclude that other determinants of fossil fuel use in Swedish electricity generation probably diminished the effects of the EU ETS.

Introduction

January 1, 2005, saw the launch of the European Union's Emissions Trading System (EU ETS)—the EU's flagship climate policy instrument and a centrepiece in its commitment to reach established greenhouse gas reduction targets. Its primary objective is to reduce emissions reductions for the least cost, over and above what would have occurred without the trading system. In this paper, we analyze to what extent the EU ETS has affected the CO_2 intensity[1] in the Swedish electricity sector with an econometric time series analysis of the period 2004–2008.

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[1] CO_2 intensity is defined as the emissions of CO_2 per generated unit of electricity.

The initial allocation of emissions allowances to participants is critical when designing an emissions trading system. In the EU ETS, this allocation—constituting significant monetary value—has largely been handed out to firms at no cost. In the first and second trading periods of the EU ETS (2005–2007 and 2008–2012, respectively), each EU member state had significant discretion in how they allocated their allowances to firms, which resulted in a plethora of different allocation methodologies. One recurring feature, however, was that many member states allocated fewer allowances to the electricity sector in relation to their past emissions, compared to other industry sectors. Two arguments seem to be the principal motivations for this decision. First, because price elasticity of electricity is low and the electricity sector is not exposed to direct competition from non-European countries, electricity companies could more easily pass on additional costs to consumers without loss of output or market share. Second, which is important for this study, several member states—including Sweden—identified the electricity sector as having better opportunities to implement low-cost abatement measures (Swedish Ministry of Enterprise, Energy, and Communications 2004a, 2004b; Kolshus and Torvanger 2005; Swedish Environmental Protection Agency 2006; Jansson, 2009). This was stated both explicitly by government officials and implicitly through the design of the so-called National Allocation Plans (NAPs).

In the aggregate, demand for emissions allowances in a cap and trade system will be constant, given the cap on total emissions. If demand for allowances in certain sectors of the economy increases, this will push the price of the allowances up.[2] Because marginal abatement costs vary across firms and sectors, their emissions elasticities, in regard to change in allowance price, will be different. If the Swedish electricity sector does have lower marginal abatement costs than other sectors, it is more likely to adjust its demand for EU emissions allowances (EUA) in response to price variations in the market than sectors with higher marginal costs for emissions reductions. Hence, the EU ETS would have a visible impact on the CO_2 intensity of electricity generation, even though total emissions in the economy are constant.

Our paper contributes to the scarce empirical literature on the influence of the carbon price on emissions reductions. We hope to shed light on whether the EU ETS has encouraged any short-term abatement of emissions in the electricity sector. If no evidence of this is present, either ex ante assumptions of low-cost measures in electricity generation were incorrect or one must find other explanations for firm behaviour. To our knowledge, this is the first study of its kind. Buchner and Ellerman (2006) make an effort to untangle the relationships between fluctuating carbon prices, over-allocation of emissions allowances, and potential abatement measures at the European level. They conclude that there likely has been abatement of emissions due to the EU ETS, but find it difficult to quantify.

The remainder of the paper is structured as follows. Section 1 provides background on the Swedish electricity market. Section 2 presents the data used for the analysis. In section 3, we develop our econometric specification, detail the variables, and discuss what results can be anticipated. In section 4, we estimate the model and show the results, while section 5 concludes.

1. Swedish Electricity Generation: Dynamics and Drivers of CO_2 Intensity

The Swedish electricity system is characterised by a high degree of liberalisation and a smaller capacity for fossil fuel-based generation, compared to other European electricity markets. The Swedish system is integrated with Norway, Finland, and Denmark. Together, they form a Nordic electricity market, which has been transformed from a regulated market into its current, more liberalised form through a gradual process that started in the early 1990s. The liberalisation of the market aimed to make its capacity more efficient, increase the choices for consumers, and

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[2] In the EU ETS, as discussed below, price variations also came from factors than changes in demand, notably political decisions, new information, and external developments that influence market expectations from the demand.

develop a more cost-effective energy supply. The dominant position of some utilities, especially in local markets, was an issue (and still is according to some observers), and a common Nordic electricity market would significantly reduce their dominance and guarantee stronger competition. Generation and trade of electricity are now open to competition, although the transmission networks are still regulated monopolies with national government control. Although it shows many characteristics of a competitive market, the integration, harmonisation, and expansion of this market is ongoing.

Table 1 shows the profile of electricity generation in the Nordic countries in 2007. In Sweden, coal is used in a small number of combined heat and power plants (CHP) and in some industrial boilers. Natural gas is also used in CHP and some peak-load units. Oil is mainly used in industrial boilers and in units which come on line during extreme cold spells or are reserve capacity when other plants are taken off line for maintenance (for example, when the Forsmark nuclear power plant was taken out of operation due to safety concerns in 2007).

Table 1   Electricity Production in the Nordic Electricity Market in 2007

	Denmark	Finland	Norway	Sweden
Total generation*	**37,2**	**77,8**	**137,4**	**145,1**
Total thermal power	**27,7**	**53,6**	**0,7**	**68,2**
Nuclear power	–	22,5	–	64,3
Other thermal power**	27,7	31,1	0,7	3,9
- Coal	20,3	13,6	–	0,9
- Oil	0,3	0,4	–	0,8
- Peat	0,0	7,0	–	0,1
- Natural gas	6,8	10,1	0,7	1,2
- Others***	0,3	–	–	0,9
Total renewable power	**9,6**	**24,2**	**136,7**	**76,9**
Hydro power	0,0	14,0	135	65,5
Other renewable power	9,5	10,2	1,7	11,4
- Wind power	7,2	0,2	0,9	1,4
- Biofuel	0,3	9,4	0,0	8,7
- Waste	1,6	0,6	0,8	1,3
- Geothermal power	–	–	–	0,0
Net imports	**-1,0**	**12,9**	**-10,0**	**1,3**

* In Norway, gross electricity production; ** fossil fuels; *** West Denmark includes refinery gas.
Source: Nordel (2007).

The dynamics of fossil fuel-based electricity in Sweden are closely tied to district heating because a large proportion of thermal power is generated by CHP units. The low fossil fuel volume also affects the dynamics of dispatch and the flexibility of the fuel mix. The demand for heat is a major determinant of CHP generation, so the impact of electricity generation on input price fluctuations may be somewhat lower, compared to simple power generation. The same can be true for electricity generation by industrial boilers, which primarily support production of other goods (such as steel or paper pulp). Thus, the price of fuels may be less important for

these units than for a regular power plant. Finally, some of the most CO_2-intensive plants exist as back-up capacity for unexpected events, which may decrease the elasticity regarding input prices.

We expect our model to capture opportunities to reduce emissions that are available to firms in the short run. These include fuel switching, technical means of improving efficiency, and dispatch planning, such as modifying the merit order. Large utilities, such as Vattenfall, EON, and Fortum, have portfolios of capacity units and can thus change the internal merit order in response to market fluctuations. Smaller firms have less flexibility and altered output is sometimes the only option for dispatch planning. Furthermore, the large district heating networks in Stockholm, Göteborg, and Malmö can respond more quickly to market price changes because they have more options for altering the merit order of units than do smaller networks. New investments offer the best possibilities for switching fuels in the long term. In the short term, some plants which co-fire fossil fuels with biofuels have some flexibility.[3] In sum, fossil fuel-based generation often constitutes the marginal capacity and thereby sets the electricity price in the Nordic market. (Figure 1 shows the principal merit order of the Nordic market.) It is clear that opportunities exist for abating CO_2 emissions in the electricity sector, but the structure of the Swedish electricity sector and the existing mix of fuels and plant types restrict how quickly firms can respond to changes in input prices.

A related question is how electricity prices are affected by the price of EUA. Research on this issue has been done for the Nordic market, as well as other European electricity markets (e.g., Sijm et al. 2006, 2008; Fell 2008; Bunn and Fezzi 2007; Alberola et al. 2008; Åhman et al. 2008, and Wråke et al. 2008). Fell (2008) uses a co-integrated vector autoregressive (CVAR) analysis and reports a near full pass through of carbon costs in Nordic electricity prices. Bunn and Fezzi (2007) use similar methodology and find comparable results for the U.K. market. This supports the view that electricity firms internalise the cost of carbon into their product prices. Alberola et al. (2008) apply a single equation specification, primarily to identify structural breaks in the allowance market itself.

Figure 1   Principal Merit Order Curve in the Nordic Electricity Market
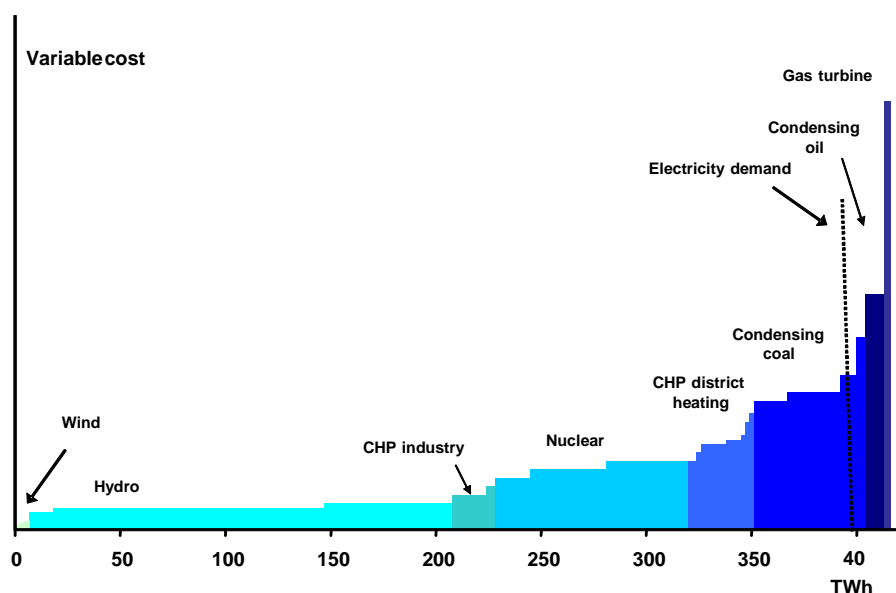
Source: Swedenergy

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[3] (For a detailed bottom-up inventory of CO_2 abatement opportunities in the Swedish energy sector, see Särnholm 2005).

2. Data

Getting access to accurate and detailed data has been one of the greatest challenges for quantitative assessments of the EU ETS. We are interested in the link between EUA prices and CO_2 intensity in the Swedish electricity sector. For this purpose, we combine two unique data series to calculate weekly CO_2 emissions: weekly output of different kinds of generation capacity and monthly data on fuel consumption for each type of plant. By dividing total emissions by total output, we can calculate the CO_2 intensity for each week. Although this approach is not ideal, it still permits a relatively detailed analysis of short-term responses in firm behaviour to variations in the price of allowances.

An implicit assumption in the construction of the data set is that the proportion of fossil fuels used in each plant type is constant within a month. This puts certain restrictions on what types of measures our analysis can capture and in what resolution. We cannot detect how much fuel switching occurs weekly by specific plant type, only their monthly levels. However, weekly variations in emissions for each plant type reflect variations in output, which means that we can capture variations in how the portfolio of plants is used on a weekly basis.

Our data covers the period January 2, 2004–August 29, 2008, i.e., from one year before the launch of the first trading period through three-quarters of the first year of the second trading period. Inputs relevant to Swedish electricity generation include prices of EUA, natural gas, coal, oil, electricity, and biofuels. (The time series for these variables are presented in figure 2.) As relative prices are most important for fuel choice, we normalise all prices against the price of electricity. We also include a proxy for the value of water in the hydropower reservoirs. Because nuclear power plants have limited flexibility to respond to short-term changes, we feel it unnecessary to include the price of uranium in the analysis.

CO_2 intensity has a clear seasonal pattern. Total demand for electricity increases during the colder months, and more fossil fuels are used. Also noteworthy is the spike in electricity prices during the second half of 2006. This was primarily driven by the dry conditions that year, which reduced the volume of water available as hydropower, as seen in figure 2 in the panel showing reservoir level.

EUA price (€/ton emitted) is the weekly average of European prices.[4] The natural gas price[5] (€/Btu) is the weekly average of day-ahead prices from the Zeebrugge hub.[6] The coal price[7] (€/ton) used is the weekly average of spot prices for coal delivered to the Amsterdam/Rotterdam/Antwerp region. The oil price (€/barrel)[8] used is the weekly average of the daily prices of Brent North Sea oil. The biofuel prices[9] (€/MWh) are based on quarterly data for Sweden, interpolated to weekly resolution. As a proxy for the value of the water available for hydropower generation, we use the deviation from average levels in the Nordic hydropower reservoirs for each week to measure the relative scarcity of water:[10]

$$(level)_t = (percent\_of\_capacity)_t - \overline{(percent\_of\_capacity)_t}$$

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[4] We use the weighted spot/over the counter (OTC) price as reported by Point Carbon.

[5] Primary source, Reuters

[6] Prices for natural gas, oil, and biofuels were converted from British pounds, American dollars, and Swedish kronor to euros, using daily exchange rates.

[7] Primary source, Reuters

[8] Ibid.

[9] Primary source, the Swedish Energy Agency

[10] Primary source, Nordpool.

where $(percent\_of\_capacity)_t$ is the percent of the Nordic region's reservoir capacity that is filled for week t and $\overline{(percent\_of\_capacity)}_t$ is the historical median of percent of capacity for week t. Electricity prices (€/MWh) used are the average day-ahead Elspot hourly system prices for weekdays.[11] Figure 2 displays plots of all variables, and table 2 gives descriptive statistics of the variables.

Table 2   Descriptive Statistics of the Variables

	Obs.	Mean	Std. dev.	Min.	Max.
CO_2 intensity	244	10,53	4,13	4,03	22,17
Gas price	244	37,67	17,31	16,21	109,24
Coal price	244	61,73	19,64	43,59	135,54
Oil price	244	49,30	14,88	23,32	102,68
Biofuel price	244	15,96	1,18	13,96	18,47
EUA price	244	11,38	10,58	0,00	30,14
Reservoir level	244	-2,24	8,10	-23,00	11,48

Figure 2   Plots of the Data

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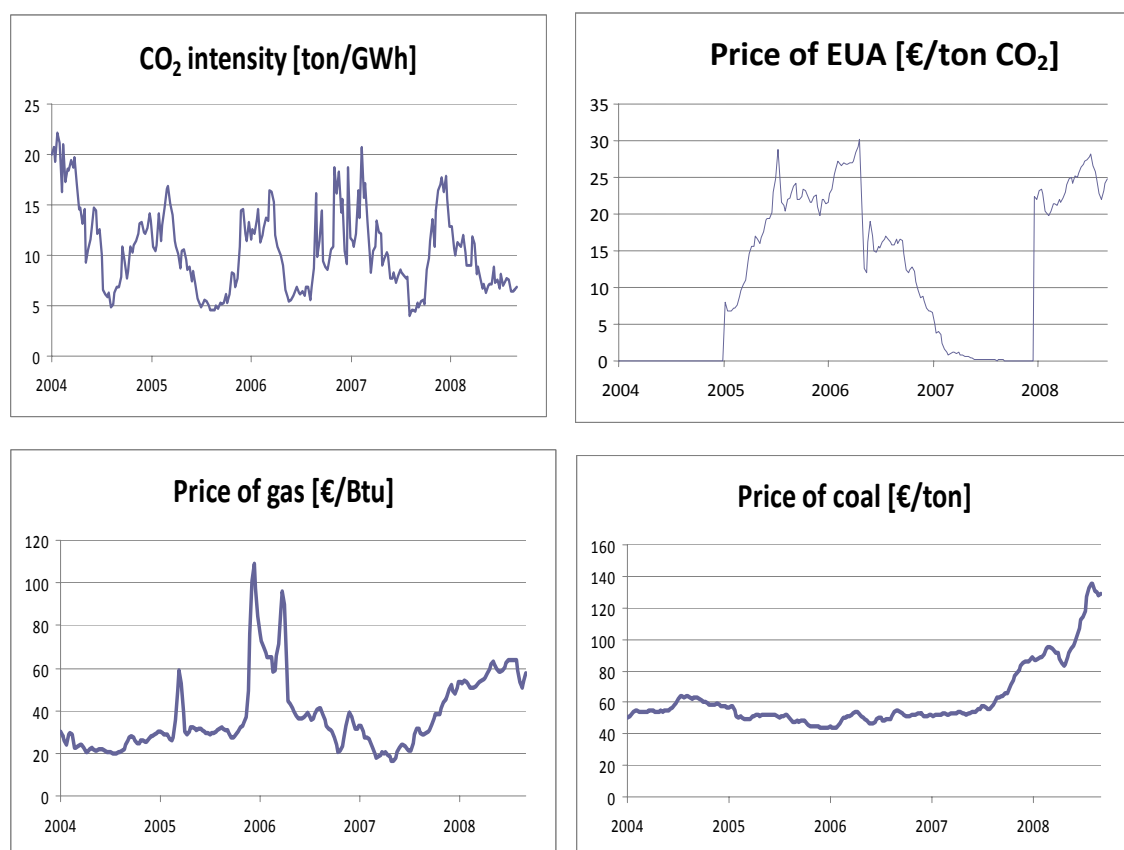
[11] We use only weekdays since demand patterns shift during weekends when households are more important. Our focus here is on the industry actors in the market.

Price of oil [€/barrel]

Price of biofuel [€/MWh]

Reservoir level (% deviation from median)

Price of electricity [€/MWh]
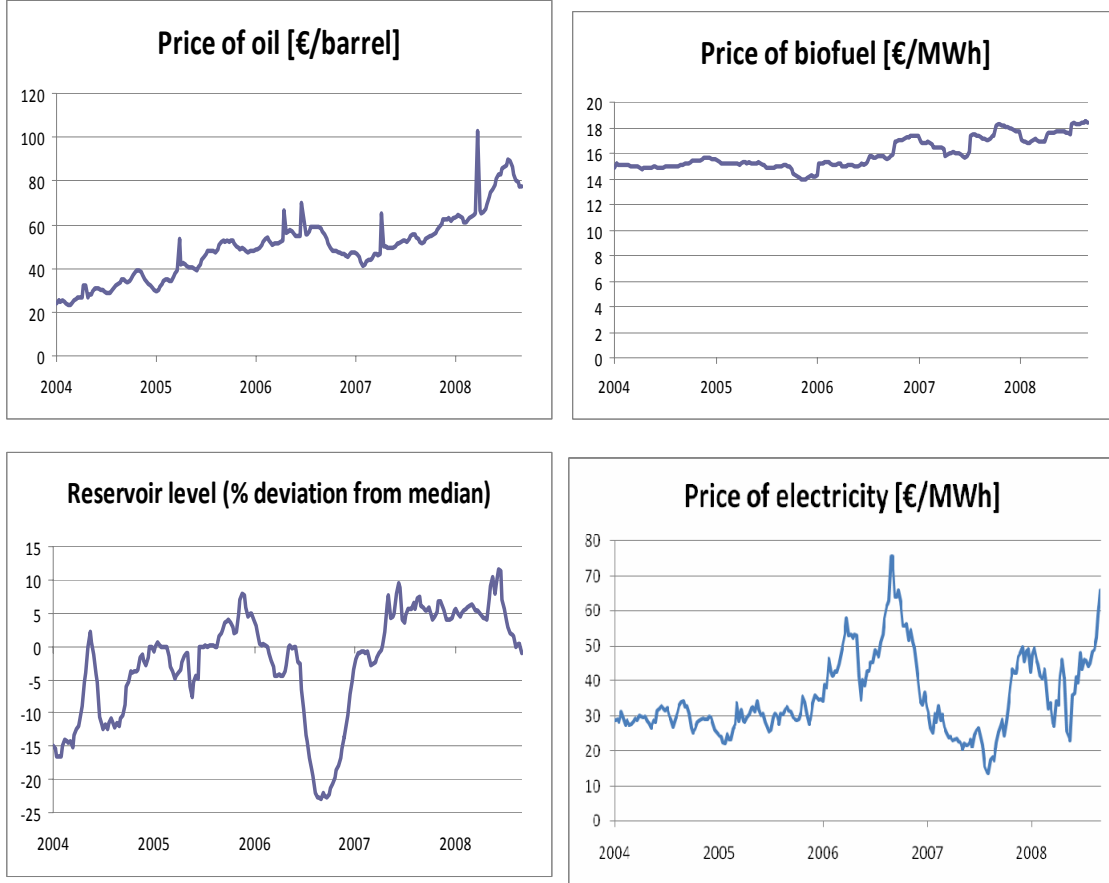
3. Econometric Specification and Anticipated Results

We apply an autoregressive distributed lag (ADL) model with the general specification:

$$Y_t = \alpha + \sum_{i=1}^{k-1}\left[\beta_i Y_{t-i}\right] + \sum_{i=0}^{k-1}\sum_{j=1}^{m}\left[\gamma_{j,i} Z_{j,(t-i)}\right] + \varepsilon_t \ ,$$

where α is the intercept; Y_t is the CO_2-intensity in week t; Z_t are the m exogenous variables in week t; k is the number of time lags chosen for each variable; β_i, $\gamma_{i,j}$ are estimated coefficients; and ε_t is the error term.

The robustness of the model and the quality of subsequent results were verified through preliminary and diagnostic tests as described below and in appendix A. In order to obtain estimates that are easy to interpret, we use the natural logarithm of the relative input prices. This also has the advantage that the variables are stationary, which simplifies the analysis. A common alternative, when variables are non-stationary in levels, is to use the first differences. We applied such a specification, but opted against it because the estimated coefficients are harder to interpret and did not make sense economically. Results are available on request.

We chose CO_2 intensity as the dependent variable in the model. Another possible approach would be to analyse CO_2 emissions and control for electricity generation. This yields very

similar results[12] to those presented below, but we detected some heteroskedasticity in this model specification.

The dramatic variations in the allowance prices have featured prominently in discussions about the EU ETS. In particular, the April 2006 price crash, the October 2006 price slide, and the sharp increase in prices in 2008 (the start of the second trading period) attracted significant attention both in the public debate and the academic literature.[13] Our dependent variable does not display any structural breaks, so we have no concerns about our approach in this regard. However, in order to ensure that the breaks in the EUA price do not influence the behaviour of Swedish electricity firms, we conducted analyses where we considered this possibility without finding any evidence. (See appendix B for a discussion and results.)

The seasonality of CO_2 intensity reflects the variable Swedish climate, not surprisingly, and needs to be included in the analysis. One option is to include seasonal dummies in the model specification, but (as discussed below) the results of our regression strongly indicate that seasonality is captured with the model specification we apply without them. Based on previous knowledge of the characteristics of the electricity system, we anticipate certain results:

- **Past CO_2 intensity.** Since we expect the system to display some degree of inertia, it is reasonable to believe that past CO_2 intensity will have a positive but decreasing influence on present intensity. That is, we anticipate a positive sign of the estimated parameter.

- **Price of natural gas.** The effect of a change in gas price depends on the substitute for natural gas in the system. As the merit order curve in figure 1 indicates oil or coal-fired plants are likely substitutes in the short run and, hence, we anticipate that an increase in the price of natural gas will cause an increase in CO_2 intensity (positive sign).

- **Price of coal.** Coal is the most CO_2-intensive fuel used in the electricity system (barring some process gases produced by the steel industry), so we expect that an increase in coal prices will prompt a fall in CO_2 intensity (negative sign).

- **Price of oil.** The effect of oil price change is more ambiguous than for natural gas and coal since it is less clear what the substituted fuel would be. If it is coal, an increase in oil price would spur an increase in emissions, but if the substitute is gas or biofuels, emissions would fall. Consequently, it is difficult to anticipate the sign.

- **Price of EUA.** EUA prices add a cost that is directly linked to emissions of CO_2, so we anticipate any effect on CO_2-intensity will be negative.

- **Price of biofuel.** As biofuels are regarded as having zero emissions, any shift away from biofuels would have a neutral or positive effect on emissions. Thus, we expect a positive estimated coefficient for the price of biofuel.

- **Reservoir level.** This variable was constructed to measure the value of water in the hydropower reservoirs. If the reservoir levels exceed the median for a particular week, we take that as a proxy for a decrease in the value of the water. Thus, we expect a negative sign on the estimated parameter for the level variable, indicating that as reservoir levels increase, more hydropower is used in the system, which prompts a fall in CO_2 intensity.

- **Variables controlling for institutional changes.** If electricity companies had realised that there was a surplus of allowances as early as April 2006 and consequently changed their behaviour (even though the market as a whole did not), we would expect a

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[12] Results available on request

[13] See, for instance, Alberola et al. (2008) for a thorough discussion of structural breaks in the EUA price.

negative sign on the April 2006 dummy estimate and a positive sign on the December 2007 dummy estimate.

4. Results and Discussion

The significant results of the regression are shown in table 3. In the regression, we include three lags of each variable. The number of lags was chosen through a step-by-step reduction from six lags until all lags were significant for at least one variable.

Oil price, EUA price, and reservoir level, all with three lags, are included in the regression, but the estimates are not significant. (The full table of results can be found in appendix A.) We find CO_2 intensity to be significant in the first lag with a positive sign. The following lags are not significant, but show drastically decreasing coefficients, as anticipated.

The price of gas is significant, in both the unlagged price and all lags. For gas and coal, the estimated lags change between positive and negative signs. This is not surprising, as it shows that a spike in input prices at time t should affect the CO_2 intensity in that period, and then fade away. The CO_2 intensity returns to its average level, hence the opposite sign of t-1 estimates. In order to understand the total effect, long-term estimates[14] of the variables were calculated in table 4. Here, all significant estimates have the anticipated signs.

Table 3   Reduced Results from the Regression Analysis and Diagnostic Tests

CO_2 intensity	Coeff.		Std. err.
CO_2 intensity lag 1	0,796	***	0,068
Gas price (relative)	0,677	***	0,184
Gas price (relative) lag1	-1,244	***	0,380
Gas price (relative) lag2	0,928	**	0,381
Gas price (relative) lag3	-0,338	*	0,183
Coal price (relative)	-1,382	***	0,257
Coal price (relative) lag1	2,185	***	0,453
Coal price (relative) lag2	-1,446	***	0,481
Coal price (relative) lag3	0,634	**	0,273
Constant	0,308	**	0,096
Diagnostic Tests			
R-square		0.86	
Adjusted R-square		0.84	
F-stat, p-value		0.00	
Durbin Watson, h-value		0.33	

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[14] The long-term solution is calculated as the sum of the coefficients for the unlagged and lagged independent variable divided by (1 minus the coefficients of the lagged dependent variable).

Breusch-Godfrey, p-value	0.31
Breusch-Pagan, p-value	0.88

*** significant at 1% level, ** significant at 5% level, and
  * significant at 10% level. Other included variables are
  oil price, EUA price, and reservoir level.

Table 4   Long-Term Estimates of the Variables

	Long-term estimate
Gas price (relative)	0,20
Coal price (relative)	-0,081
EUA price (relative)	0,14

Even though the estimates of the EUA price were not significant, we chose to present the long-term estimate since this is the most important variable in the analysis. We found no other significant estimates.

To our knowledge, there is no data on biofuel prices with better resolution than ours. Still, because prices are stable (see figure 2), we have doubts about how much information the data series contains; thus, in the final model the variable is excluded from the regression.[15]

The robustness of our model we verified through diagnostic statistics (see table 3). We calculated the Durbin-Watson h-value and the Breusch-Godfrey test statistic to detect autocorrelation, and the Breusch-Pagan tests for Heteroskedasticity, and test statistics showed no indication of either.[16] To ensure that the insignificant variables do not jointly influence the CO_2 intensity, we performed F-tests on the sum of the coefficients for these variables, without finding significance.

Because we began the analysis one year before the launch of the EU ETS, the first year of the study had no prices for allowances. In order to keep this year in the analysis, we set the logarithm of the relative EUA price to zero for this period. We also ran regressions with two different low prices for allowances, 0.01 €/ton and 0.0001 €/ton, for this period. To ensure robustness, we also ran regressions with 2004 omitted from the analysis altogether. In all cases, results were similar to those presented here.[17] As mentioned previously, we further tested model specifications with the variables in first differences and with CO_2 emissions as our dependent variable in lieu of CO_2 intensity.

Figure 3 shows observed values of CO_2 intensity along with model predictions for CO_2 intensity. The fit of the model indicates that the specification is able to capture most variations, including the seasonality in CO_2 intensity.

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[15] When the price of biofuel is included, the estimated coefficient is insignificant.

[16] Additional plots of the data, such as the residuals versus the fitted value of the CO2-intensity, pp-plot, and qq-plot were studied. All showed the same, with no evidence of autocorrelation or heteroskedasticity.

[17] Results available on request.

Figure 3   Observed Values of the Dependent Variable (blue line) and Predicted Values from Our Model Specification (red line)
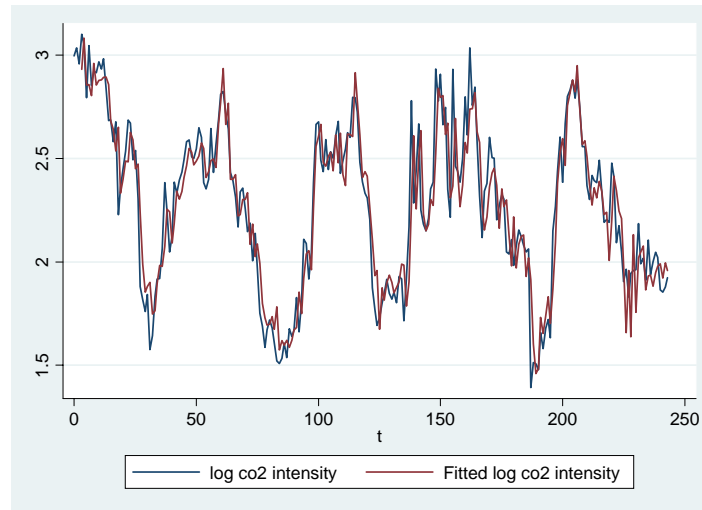
Our results do not indicate any link between the price of EUA and the CO$_2$ intensity of Swedish electricity production in the period 2004–2008. We see a number of potential explanations for our findings:

- **Other drivers for CO$_2$ emissions, stronger than the price of carbon, are hiding or diminishing the effect of the EU ETS.** The generation in the fossil-fuel intense units, such as CHP and industrial boilers, can also be driven by special circumstances (accidents in other plants, unplanned maintenance), but it can also include heat demand.

- **The price of carbon, so far, with the EU ETS has been too low to induce any significant emission reductions.** This argument carries some weight, particularly because the price of EUA approached zero toward the end of the first trading period. However, at any point in time, a positive price of EUA creates incentive to abate emissions.

- **Sectors other than electricity have implemented emission reduction measures.** This is certainly possible. However, it means that the Swedish government was wrong in its assumption before the launch of the EU ETS that easy, low-cost opportunities for emissions reductions were more prevalent in the electricity sector than in other sectors.

- **Emission reductions were made in other member states.** This is also possible, but if it were the only explanation, it would mean that firms in Sweden were the only buyers in the EU ETS.[18]

- **The response time of abatement measures is longer than what our model can capture.** New, innovative abatement measures may require lead times of several years to become accepted, active, or built. However, a number of existing abatement measures could be introduced more quickly, such as fuel switching, efficiency improvements, and dispatch planning, until new measures replace old plants with new and more efficient generation capacity.

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[18] Although each member state, as well as the EU Commission, collects data on market transactions, this data is not public and a deeper analysis of this issue has not been possible.

- **Firms are still learning to incorporate the cost of carbon emission into their decisions and thus did not respond fully.** This could help explain why it is difficult to link a relatively high price of EUA to abatement measures in a single sector. However, as the electricity sector is perhaps the best informed of all sectors participating in the EU ETS this argument seems unlikely.

- **Firms were expecting the price of EUA to reach zero at an early stage and thus had no incentive to implement abatement measures.** This reasoning does not convey why the price was positive for most of the trading period. Without speculating about an inefficient market for EUA—in which some agents may have supported the price using market power to gain economic benefits—this is difficult to explain. Ironically, those who put forward this argument often point to the electricity sector, claiming that many firms reaped substantial windfall profits from the higher electricity prices resulting from the EU ETS. We would also expect our model to capture this effect through the variables allowing for institutional changes to affect the results.

- **The response of CO_2 emissions to prices in EUA is asymmetric.** This argument is relevant for abatement measures where a reversal does not decrease operating costs. An example would be efficiency improvements; it would not make sense for a firm to reduce efficiency even if the price of EUA dropped below the level which triggered the improvement in the first place. However, for other measures, such as fuel switching, this explanation seems less likely to hold.

5. Conclusions

Given that the electricity sector was generally thought to hold many abatement opportunities and given that the objective of the EU ETS is to lower CO_2 intensity in the economy in general, the findings may be disturbing. However, even though our results do not indicate a significant impact of the EU ETS on emissions from Swedish electricity generation in the short run, it is difficult to see how a positive price of EU, in general and over time, would *not* lead to any abatement of carbon emissions over and above those in a scenario without a price on carbon. If, as previous research indicates, firms incorporate the opportunity cost of carbon emissions into their operating and investment decisions, we would expect to see emission reductions measures—which would not have been implemented if there was no cost of emitting carbon. Hence, we believe that the absence of a significant impact of EUA prices on CO_2 intensity primarily hinges on the structure and characteristics of Swedish electricity generation.

We draw two main conclusions. First, it seems unlikely that the EU ETS has generated any significant reductions of CO_2 emissions in Swedish electricity generation. Second, it seems unlikely that there are significant volumes of low-cost CO_2 abatement measures with short response times in the Swedish electricity sector. In order to better understand the long-term impacts of the EU ETS on CO_2 intensity, one needs to complement the analysis with studies that have stronger emphasis on investment planning.

References

Åhman, M., D. Burtraw, and K. Palmer. 2008. Interactions between the EU ETS and Nordic Electricity: Setting the scene. Elforsk Report 08:29. Stockholm, Sweden: Elforsk Market Design.

Alberola, E., J. Chevallier, and B. Cheze. 2008. Price drivers and structural breaks in European carbon prices 2005–2007. *Energy Policy* 36: 787–97.

Bai, J., and P. Perron. 1998. Estimating and testing linear models with multiple structural changes. _Econometrica_ 66(1): 47–78.

Bunn, D.W., and C. Fezzi. 2007. Interaction of European carbon trading and energy prices. FEEM Working Paper, no. 63.2007. Milan, Italy: Fondazione Eni Enrico Mattei.

Ellerman, D., and B. Buchner. 2006. Over-allocation or abatement? A preliminary analysis of the EU ETS based on the 2005 emissions data. FEEM Working Paper, no. 139.06. Milan, Italy: Fondazione Eni Enrico Mattei.

Environment Protection Agency. 2006. Determination of the distribution of allowances under the National Allocation Plan 2008-2012: A Report for the EPA. Prepared by Indecon (International Economic Consultants) for EPA. County Wexford, Ireland: EPA. http://www.epa.ie/downloads/pubs/air/etu/name,12257,en.html.

Fell, H. 2008. EU-ETS and Nordic Electricity: A CVAR analysis. Elforsk Report 08:57. Stockholm, Sweden: Elforsk Market Design.

Gilbert, A., J.-W. Bode, and D. Phylipsen. 2004. Analysis of the National Allocation Plans for the EU Emissions Trading Scheme. Ecofys Report YUES4003. London: Ecofys.

Greene, W.H. 2003. *Econometric Analysis*. 5th ed. Upper Saddle River, N.J., USA: Prentice-Hall, Inc.

Jansson, K. 2009. Personal communication to authors. [Authors' note: Mr. Jansson was the chairman of two parliamentary commissions, "FlexMex" and "FlexMex II" which made many of the early preparations for the Swedish NAP I. He is currently chairman of Swedenergy.]

Kolshus, H., and A. Torvanger. 2005. Analysis of the EU member states 'National Allocation Plans. CICERO Working Paper 2005:02. Oslo, Norway: CICERO.

Nordel. 2007. Annual Statistics 2007. Online document. Available at www.nordel.org.

Särnholm, E. 2005. Åtgärdskostnader för minskning av koldioxidutsläpp vid svenska kraftvärme- och värmeanläggningar [Abatement costs for emissions of carbon dioxide in Swedish combined heat and power and heat generation plants]. IVL Report B1650. Stockholm, Sweden: IVL Swedish Environmental Research Institute Ltd.

Sijm, J.P.M., K. Neuhoff, and Y. Chen. 2006. CO$_2$ cost pass through and windfall profits in the power sector. *Climate Policy* 6: 49–72.

Sijm, J.P.M., S.J. Hers, W. Lise, and B.J.H.W. Wetzelaer. 2008. The impact of the EU ETS on electricity prices: Final report to Directorate General of Environment of the European Commission. ECN Report E-08-007. Petten, The Netherlands: Energieonderzoek Centrum Nederland (ECN).

Swedish Ministry of Enterprise, Energy and Communications. 2004a. Sweden's National Allocation Plan. Memorandum 2004-04-22. Online document. Available at http://www.regeringen.se/sb/d/108/a/29896.

———. 2004b Recent development in emissions trading in Sweden." Memorandum 2004-09-27. Online document. Available at http://www.iea.org/textbase/work/2004/ghg/country_reports/sweden.pdf

Wråke, M., E. Myers, S. Mandell, C. Holt, and D. Burtraw. 2008. Pricing strategies under emissions trading: An experimental analysis. RFF Discussion Paper DP 08-49. Washington, DC: Resources for the Future.

Zetterberg, L., K. Nilsson, M. Åhman, A.-S. Kumlin, and L. Birgersdotter. 2004. Analysis of National Allocation Plans for the EU ETS. IVL Report B1591. Stockholm, Sweden: IVL Swedish Environmental Research Institute Ltd.

Appendix A Full Regression and Preliminary Tests

Results of the Regression Analysis

Table A1 Full Results from the Regression Analysis and Diagnostic Tests

CO_2 intensity	Coefficient		Std. error
CO_2 intensity lag 1	0,796	***	0,068
CO_2 intensity lag 2	0,036		0,084
CO_2 intensity lag 3	0,051		0,067
Gas price (relative)	0,677	***	0,184
Gas price (relative) lag1	-1,244	***	0,380
Gas price (relative) lag2	0,928	**	0,381
Gas price (relative) lag3	-0,338	*	0,183
Coal price (relative)	-1,382	***	0,257
Coal price (relative) lag1	2,185	***	0,453
Coal price (relative) lag2	-1,446	***	0,481
Coal price (relative) lag3	0,634	**	0,273
Oil price (relative)	0,030		0,160
Oil price (relative) lag1	-0,119		0,191
Oil price (relative) lag2	0,071		0,191
Oil price (relative) lag3	-0,160		0,157
EUA price (relative)	-0,027		0,024
EUA price (relative) lag1	0,002		0,033
EUA price (relative) lag2	-0,003		0,033
EUA price (relative) lag3	0,012		0,023
Reservoir level	0,003		0,009
Reservoir level lag1	0,001		0,014
Reservoir level lag2	-0,001		0,014
Reservoir level lag3	-0,001		0,009
Constant	0,308	**	0,096
Diagnostic tests			
R-square		0.86	
Adjusted R-square		0.84	
F-stat, p-value		0.00	
Durbin Watson, h-value		0.33	
Breusch-Godfrey, p-value		0.31	
Breusch-Pagan, p-value		0.88	

*** significant at 1% level, ** significant at 5% level, and * significant at 10% level.

Stationarity

A visual inspection of the data in figure 2 in the text indicates potential non-stationarity in some the variables. To formally test this, we performed the Augmented Dickey-Fuller (ADF) test on all variables (table A2). We cannot reject the null of a unit root (and thus non-stationarity) for any variables except CO$_2$ intensity and reservoir level. In order to obtain stationary variables, we transformed the price variables into relative prices with the price of electricity as base, and then took the natural logarithm of the relative prices. Relative prices to some extent also capture the magnitude and importance of each input price in relation to the price of the output (electricity). The series are presented in figure A1 and the test statistics from the ADF test for the transformed variables are presented in table A2.

Table A2   Test Statistics for Augmented Dickey-Fuller Test with Drift, 3 Lags

CO$_2$ intensity	Gas price	Coal price	Oil price	Biofuel price	EUA price	Reservoir level
3,99***	-2.23	1.11	-1.03	-0.78	-1.49	-2.62[*]

Log CO$_2$ intensity	Log gas price (rel)	Log coal price (rel)	Log oil price (rel)	Log biofuel price (rel)	Log EUA price (rel)	Reservoir level
-3,58***	-2,96***	-1,91*	-2,79***	-2,27***	-1,87*	-2,65***

*** significant at 1% level, ** significant at 5% level, and * significant at 10% level. Critical values applied are -1,29 for 10%, -2,65 for 5% and -2,34 for 1%.

Figure A1   Plots of the Transformed Variables
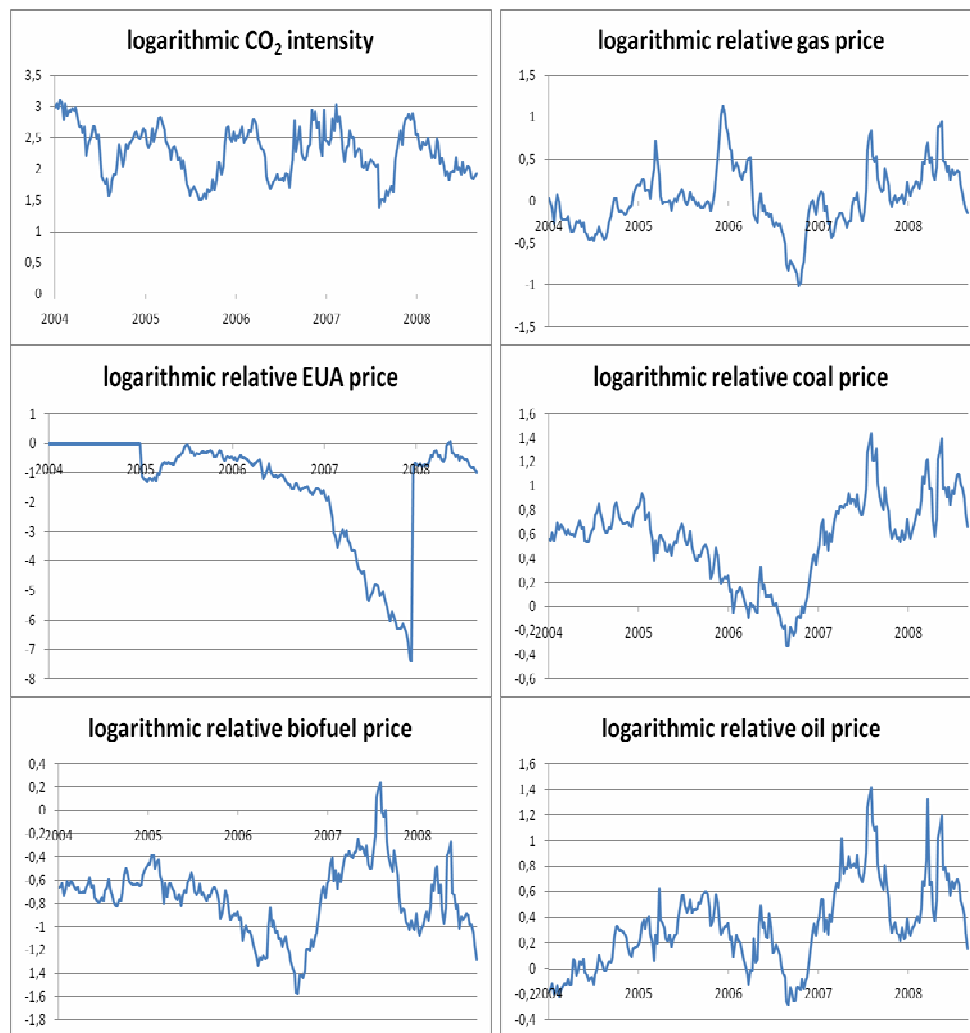
Multicollinearity

Multicollinearity is a common cause of concern in regression modelling. A simple first step to assess the risk of multicollinearity is to check the cross correlations between the variables. These are shown in table A3.

Table A3   Correlation Coefficients

	Log CO_2 intensity	Log gas price (rel)	Log coal price (rel)	Log oil price (rel)	Log biofuel price (rel)	Log EUA price (rel)	Reservoir level
Log CO_2 intensity	1,00						
Log gas price (rel)	-0,08	1,00					
Log coal price (rel)	-0,19*	0,40*	1,00				
Log oil price (rel)	-0,49*	**0,56***	**0,67***	1,00			
Log biofuel price (rel)	-0,15*	0,31*	**0,81***	**0,64***	1,00		
Log EUA price (rel)	0,03	-0,01	-0,21*	-0,41*	-0,29*	1,00	
Reservoir level	-0,25*	**0,69***	**0,56***	**0,77***	0,44*	-0,31*	1,00

Moderate to high correlations exist between some variables (in bold in table A3). The high correlation between the biofuel price and coal and oil prices is unexpected, but could come from the construction of the variable. Quarterly prices of biofuels are fairly stable, and a large proportion of the fluctuation observed in our variable is in fact related to the SEK-Euro exchange rate. The fluctuation in the series may, therefore, be related to the general state of the economy, which in turn may be correlated with the price of coal and oil. The reservoir level shows high correlation with the prices of gas, coal and oil. We see no apparent theoretical underpinning for this correlation.

Table A4   Results of Variance Inflation Test

	Log gas price (rel)	Log coal price (rel)	Log oil price (rel)	Log biofuel price (rel)	Log EUA price (rel)	Reservoir level	Degree day	Log Nuclear gener.
VIF	2,78	3,68	5,76	3,93	1,64	4,06	2,67	2,43

To further explore whether the presence of multicollinearity could be problematic, we performed a Variance Inflation Factor (VIF) test. The results are presented in table A4. As a rule of thumb, if VIF exceeds 10, a variable can be suspected of high collinearity with some other variable.[19]

The conclusion from these procedures is that multicollinearity does not appear to be a problem for the analysis.

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[19] For a discussion, see Greene (2003).

Appendix B. Structural and Institutional Changes

Assessments of the NAPs by Zetterberg et al. (2004) and Gilbert, Bode, and Phylipsen (2004), before the EU ETS was started, indicated that installations covered by the EU ETS were given more allowances than what their emissions had been historically. They also received more allowances than warranted, if each sector of the economy were to carry an equal burden in relation to the EU Kyoto target. This led many to criticise the system for not being stringent enough even before it was launched.

Nevertheless, the first year of trading saw prices of EUAs, which were higher than many observers had expected, peaking at over 30 €/ton early in 2006 (figure 2). However, the price variations were significant and during 2007 prices fell to near zero levels. Most observers now agree that there was, in fact, an over-allocation of emissions allowances which contributed to a decline in prices. This led to speculations whether the EU ETS has reached its primary goal of reducing carbon emissions. When the second trading period was launched in January 2008, prices increased again, indicating expectations of a shorter market for allowances.

When studying the plot in figure 2 in the text, four sudden changes in the EUA price series are apparent: January 2005, April 2006, mid-autumn 2006, and December 2007. The first price increase marks the launch of the EU ETS, before which there was no price on CO_2 emissions. The sudden drop in prices in April 2006 can be directly related to the release of data of verified emissions for 2005, which indicated an over-allocation of emissions allowances. The October 2006 price slide can be linked to statements by the EU Commission, which pointed to a more stringent allocation in the second trading period starting in 2008. This may have been interpreted as another indication that there was a surplus of allowances in the first trading period, prompting a further decline in prices. The final dramatic price change, in December 2007, indicates the start of the second trading period.[20]

Some observers were surprised that the price of EUA did not immediately drop to zero after the verified 2005 emissions became available in April 2006. Instead, the prices were relatively stable for a period, before they gradually fell towards zero in 2007. This seems to indicate that the market as a whole did not realise that there was a surplus of allowances until mid-2007. However, it has been suggested that electricity firms—given their long experience from trading in markets similar to the EU ETS, the importance of the EUA price to their operations, and their active role in the debate on the EU ETS—may have been in a better position to analyze the EUA market than other industry sectors. Furthermore, they did have an incentive to keep allowance prices positive, as this earned them profits on the large volumes of non-emitting power generation, such as nuclear and hydro.

This could suggest controlling for a change in behaviour of Swedish electricity firms at the times of the breaks in EUA prices, even though our dependent variable, CO_2 intensity, does not show corresponding structural breaks. For example, it is possible that seasonal variations in electricity generation are masking effects of institutional changes. Therefore, we also ran regressions with dummies aimed at controlling for these changes in the model specification.

In order to formally identify the break points in the EUA price series, we ran the test developed by Bai and Perron (1998), allowing for four potential breakpoints. The test indicates a first break in the first week of March 2005, a second break in the third week of April 2006, a third break in the second week of January 2007 and a fourth break in the last week of 2007.

If we instead allow for three structural breaks in the Bai and Perron test, the break in October 2006 appears instead of the January 2007 break. Due to the uncertainty in the October

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[20] As our price series are for contracts with December delivery, the increase in prices occurred in December 2007, rather than January 2008.

2006 break, we limited our model to include the April 2006 price drop and the December 2007 price increase.

A comparison with previous research shows that Fell (2008), Bunn and Fezzi (2007), and Alberola et al. (2008) included the April 2006 break. Fell also included dummies for the start of the second trading period, while Alberola et al. included the break in October 2006 in their analysis. Alberola et al. used a slightly different approach and first identified a "compliance break period" between April 26 and June 23, 2006, and excluded this period from the data. They then identified a second break in October 2006 and explained it in a similar way as we do.

However, regression results do not show any significance in either the April 06 dummy or the December 2007 dummy.[21] Thus, we find no support for the suggestion that electricity firms altered their way of incorporating the carbon price after these events.

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[21] Results available on request.

Paper V

New entrant allocation in the Nordic energy sectors: incentives and options in the EU ETS

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Abstract

In the EU emission trading scheme (EU ETS), the treatment of new entrants has proved to be one of the most contentious issues. This article analyses the impact of allocation to new entrants in the energy sector, and identifies options for improved regulation in this field. The point of departure for the discussion is a comparative analysis of the allocation in phase I and phase II of the EU ETS to two hypothetical energy installations located in different EU Member States. The study focuses on the Nordic countries due to their integrated energy market. The quantitative analysis was complemented by interviews with policy-makers and industry representatives. The results suggest that current allocation rules can significantly distort competition. The annual value of the allocation is comparable to the fixed investment costs for a new installation and is not insignificant compared to expected revenues from sales of electricity from the installation. The study finds that the preferred option would be that Nordic countries should not allocate free allowances to new entrants in the energy sector. This should be combined with adjusted rules on allocation to existing installations and closures in order to avoid putting new installations at a disadvantage. A second, less-preferred choice would involve harmonized benchmarks across the Nordic countries.

Keywords: Emissions trading; Allocation; New entrants; Competitiveness

1. Introduction

In the EU emission trading scheme (EU ETS), the treatment of new entrants to the scheme has proved to be one of the most contentious issues. In the first set of national allocation plans (NAPs) this is also one of the areas where policies among Member States differ most. Clearly, given that all 25 Member States have set up provisions to give allowances to new installations free of charge, allocation to new entrants is a political priority. Decisions about allocations to new entrants involve considerations about investment incentives, perceived fairness, economic efficiency and competitiveness.

From a political point of view, it is difficult to introduce policies that would make new investments less attractive and closures of installations more favourable in one's own country than it is in

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neighbouring Member States. The government of a Member State may be tempted to introduce incentives that compromise the efficiency of the trading programme as a whole. The possibility that Member States could obtain a better outcome by individual action that undermines the outcome for the broader ETS constitutes the well-known 'prisoner's dilemma'. Thus, the extent to which national competitiveness may be affected by the allocation is a highly relevant question.

Although it is still too early to pass final judgement on the effects and efficiency of the EU ETS, it is clear that the design of some allocation methodologies has created distortions in competitiveness, and that the effectiveness of the system in guiding investments to low-carbon technology can be improved. However, there are good reasons to structure the discussion on competitiveness, as, in simplified terms, the EU ETS affects at least three distinct aspects of the issue:

– the competitiveness of Member States
– the competitiveness of new compared with existing installations
– the competitiveness of European companies active in a global market.

This article aims to examine the incentives created by the treatment of new entrants in the energy sectors of some northern European countries in the first two phases of the EU ETS. The article analyses potential effects of this regulation and identifies options for harmonization and improvements in future phases of the EU ETS. At the time of writing (December 2006), not all of the Member States had notified their NAPs for 2008–2012. Furthermore, some NAPs contained too little information on the treatment of new entrants to allow a complete assessment; however, for the purposes of this article, we believe that the available material is sufficient.

The analysis primarily covers the first two of the three aspects of competitiveness mentioned above. The first aspect, the competitiveness of Member States, is affected by differences in regulation between Member States. If the objective is only to create a level playing field within the EU, harmonization of the rules across Member States in such a way that the incentives provided to the operators by the allocation are equal is more important than the actual details of the regulation. Given that the treatment of new entrants can affect the efficiency of the entire trading scheme (Åhman et al., 2006), one could argue that the best solution would be to regulate at the EU level, although this would require a change in the EU Directive governing the EU ETS (European Union, 2003). Harmonizing the rules in the Nordic counties would be an important step in this direction. Given the structure of the Nordic energy market, with limited transmission capacity to the rest of the EU, a Nordic harmonization may suffice to avoid the most serious distortions of competition for companies within those countries. In the longer term, in particular considering the EU objective to reach an integrated European energy market, harmonization across the EU would carry further advantages.

The second aspect – competitiveness of new versus existing installations – should be analysed in the light of the close link between treatment of new entrants and rules on closures. As discussed in this article and shown previously (Åhman et al., 2006; Bode et al., 2005; Schleich and Betz, 2005) the policy on closures that dominates in the NAPs in the first two phases in the EU ETS, i.e. to withdraw the allocation to installations that close, constitutes an implicit subsidy of existing installations, as it provides incentives to not close down old plants. This puts new entrants at a disadvantage if the allocation to new installations is not generous enough to compensate for the subsidy of incumbents. Hence, the rules on new entrants in combination with those on closures

may have important consequences for the incentives for new investments, to what technologies those investments are directed, and the competitiveness of new versus existing installations.

The third aspect, competitiveness of European companies active in a global market, is mainly driven by the differences in climate policy between EU and the rest of the world. In the context of emissions trading, the most important factor is the difference in the price of carbon between the EU and the rest of the world. Although the general competitiveness of European industry is important and could indeed be affected by European climate policy, it is not as relevant when discussing allocation methodologies. The first priority in order for the EU to be able to pursue a progressive climate policy without risking the competitiveness of industry would be to continue the efforts to achieve a broader, preferably a global, climate regime.

The starting point for the analysis is a comparison of the hypothetical allocation to two new standard energy installations to be localized in Denmark, Finland, Sweden, Germany, Poland, Estonia, Latvia or Lithuania, in phase I and phase II of the EU ETS, respectively. In order to understand the importance of the allocation, and the extent to which differences in allocation methodology can affect where investments in new capacity are made, the value of the allocation is compared to the fixed costs and annual revenues of the standard installations.

The discussion is focused on the Nordic countries and Germany, Poland and the Baltic States. The Nordic electricity market is almost completely integrated, with increasing transmission capacity to the other countries studied. Thus, all of the chosen countries affect each other to various degrees and the chosen region is well suited to study how differences in allocation methodology can affect an integrated or semi-integrated energy system.

2. Are new entrants discriminated against compared with existing installations?

There is an ongoing debate on how allocation to new entrants should be made, and to what extent allocation affects company behaviour and decisions (e.g. Åhman and Holmgren, 2006; Sterner and Muller, 2006). A basic question is whether new entrants should receive free allowances at all. A second question concerns the extent to which free allocation actually affects the investment decisions of operators.

Some observers claim that the denial of free allowances would discriminate against new entrants compared with existing installations, thus inhibiting new investments. Opponents of this view (e.g. Åhman and Zetterberg, 2005; Haites and Hornung, 1999; Harrison and Radov, 2002) point out that the main argument for free allocation to existing installations is to compensate them for sunk costs, i.e. costs for investments that were made before the ETS was constructed and that are now less profitable due to the carbon price. Since new entrants have no such sunk costs and operate with full knowledge of the ETS, this justification for free allocation to new entrants is not valid.

In our view there are two significant arguments that could justify free allocation to new entrants.

First, capital markets discriminate in the price they charge a firm for acquiring new capital in response to observable accounting measures such as debt, liquidity, and cash flow, and also due to uncertainties such as exposure to price volatility in factor inputs, including emission allowances. Since the firm is capital constrained, and the cost of capital varies with the amount of capital needed, free allocation reduces the need of the firm to borrow money. The lower requirement to obtain capital may reduce the firm's cost of capital and convey economic advantage to owners of incumbent installations that receive allowances for free relative to investors in new installations, in cases where they have to buy all the allowances they need.[1]

Second, most Member States implicitly subsidize existing installations by withdrawing the allocation to existing installations that decide not to operate (e.g. Neuhoff et al., 2006). Under this policy, the operator of an installation will not only maximize profits with respect to the cost of production and market price of the products, but also has to take into account the value of the allowances that will be lost should the installation be closed. This puts new entrants, which could potentially replace existing installations, at a disadvantage.

It can be shown that if the value of the allowances that are lost equal the allocation to the new investment, this effect is diminished (Åhman et al., 2006). One approach that approximates this prescription is illustrated by the transfer rules used by, for instance, Germany and Austria: the allowances from an installation that closes can be transferred to a new installation. However, Bode et al. (2005) have argued that the German transfer rule still discriminates against new entrants and causes large profits for incumbent generators. Furthermore, the Austrian rules explicitly state that the new installation has to have the same owner as the old one; thus the rule provides little help to new investors wanting to enter the market.

Thus, it is our view that there may be some justification for allocating free allowances to new entrants, particularly considering the effects of the current rules on closures.

3. Comparison of new entrant allocation methodologies

Although in theory there are an infinite number of options to compare allocation methodologies, as well as many different terminologies, the allocation methods can be structured in a few different approaches:

– *Input- or output-based.* Input-based allocation is calculated by multiplying input or production factors such as fuel use or installed capacity with a benchmark (e.g. 1,710 EAU/MW). Output-based allocation is calculated by multiplying e.g. emissions or generated energy with a benchmark. The major advantage of choosing output-based allocation over input-based is that it rewards high-efficiency technologies.
– *Fuel-neutral or fuel-specific.* Some countries use different benchmarks for different fuels, or groups of fuels. The major advantage with fuel neutral benchmarks is that they provide incentives to use low-carbon fuels. Fuel-specific benchmarking provides incentives that are similar to allocation based on emissions only, thus does not encourage investments in low-carbon fuels.[2]
– *Technology-neutral or technology-specific.* This means using different methodologies for electricity generated in condensing plants and in combined heat and power (CHP) systems. This can be used in order to promote one specific technology or to accommodate for the different conditions in which different technologies are used. If the objective is to create incentives for least-cost emissions reductions, however, there is little justification for technology-specific benchmarks.[3]
– *Product-specific or product-neutral* ('product' in this context being electricity or heat). In the context of competition for investments between countries, heat and power have very different characteristics. The advantage of using different benchmarks for heat and electricity is that it would allow this difference to be taken into account. Harmonization is a higher priority for electricity than it is for heat.

The NAPs of the Member States in this study contain examples of all the approaches listed above, in various combinations. In addition, even when the same basic approach is used, for instance

output-based, fuel-specific benchmarking, the actual number of allocated allowances differs significantly between countries. The Nordic countries all apply different benchmarks for electricity and heat.

Depending on which approach is used, different incentives are created for investments. This will affect the competitiveness of different fuels, products and technologies. Even if two different approaches can result in identical allocation, the investment incentives can still be different.

For the competitiveness of a country, the total volume of allocated allowances for a given installation may be as important as what incentive structure is created by the allocation methodology. That is, an operator who has already chosen which fuel or technology to use only cares about how many allowances he will receive upon entering the market.

For a more comprehensive analysis of phase I NAPs, see, e.g., Åhman and Holmgren (2006), Kolshus and Torvanger (2005), DEHSt (2005), Matthes et al. (2005), Zetterberg et al. (2004) and Ecofys (2004).

3.1. Quantitative examples

We have calculated the allocation that would be awarded to two hypothetical installations if they were to be started (Table 1). The first installation type is taken from an ongoing Elforsk project (Ekström et al., 2006). The second is modelled closely on the CHP currently planned by Göteborg Energi (Göteborg Energi, 2005).

Figures 1 and 2 show what allocation the installations would receive if they were built in the respective countries, in relation to expected annual emissions.

There are few signs of increased harmonization of the allocation methodologies to new entrants in phase II, and it is striking how much the allocation differs between Member States. For a natural-gas-fired condensing plant the percentage of emissions covered range from zero in Sweden in both phases, to 119 in Lithuania (phase I) and 131 in Germany (phase II). For a CHP, the allocation also differs widely across countries: in phase I ranging from 62% of emissions covered (Sweden) to 131% in Germany, and in phase II from 70% in Finland to 157% in Germany (Figure 2).

Germany has changed the allocation methodology to new entrants in phase II, resulting in significantly increasing volumes allocated on installation level. The allocation is based on so-called standard utilization factors, specifying the number of operational hours for different installation types, installed capacity and BAT (best available technology) factors. In phase I, the allocation was not based on standard utilization factors but on projected emissions. Furthermore the allocation was proposed to be subject to *ex-post* adjustments, although this proposal was rejected by the EU Commission. In phase II, the explicit *ex-post* adjustment rules have been discarded. However, the NAP states that the allocation in coming periods will be based on the actual number of hours of operation in previous periods, thus providing an updating component of the allocation methodology.

Table 1. Hypothetical standard new installation

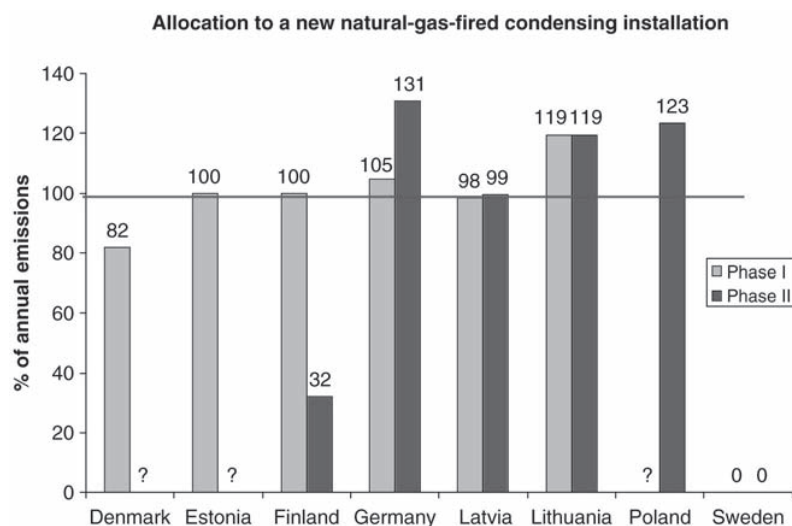| Fuel	Technology	Power efficiency	Total efficiency	Production capacity	Operational hours
Natural gas	CC condensing	58%	58%	400 MW$_e$	6000 h/a
Natural gas	CHP	50%	92.5%	261 MW$_e$	5000 h/a
				294 MW$_{heat}$	

Figure 1. The allocation to a new natural-gas combined-cycle electricity production unit (no heat) in different Member States. Results presented as percentage of expected annual emissions, i.e. the number of allowances required for compliance, covered by the allocation, with the horizontal line representing 100% of allowances needed for compliance. NAP I of Poland and NAP II of Estonia did not contain enough information to allow calculation of the allocation, while Denmark had not notified its NAP II to the European Commission at the time of writing (December 2006).

It is interesting that while both Poland and Germany claim they use benchmarks based on BAT, the benchmarks used are very different. For a natural-gas-fired power plant the Polish benchmark is 430 kg CO_2/MWh electricity whereas the corresponding German value is 365 kg CO_2/MWh electricity, a difference of almost 18%. For heat the benchmarks used are more similar, being 60 kg/GJ for a Polish natural-gas-fired CHP, corresponding to 216 kg CO_2/MWh, and 215 kg CO_2/MWh heat for a corresponding German installation.

3.2. The role of assumptions and forecasts

Forecasts and assumptions are used frequently in the NAPs. Parameters such as assumed annual operating hours and efficiency of the installations differ significantly across Member States, which has a significant impact on the allocation.

For instance, the Danish allocation to a new natural-gas combined-cycle condensing plant (NGCC) in phase I cover approximately 82% of the estimated annual emissions. The main reason for the 20% shortfall is that the estimated number of operational hours is 20% higher in our example than the default value used for the calculation of the benchmark in the Danish NAP.

In Finland we find an opposite example. The 20% surplus of allowances allocated to the CHP can be explained by the fact that Finland in its NAP assumes 6,000 hours of operation per annum, while our standard assumption is 5,000 hours.

An interesting feature of the German NAPs is that they specifically state that the benchmarks used in the allocation will not be changed until 14 years after the installation has started its operation.
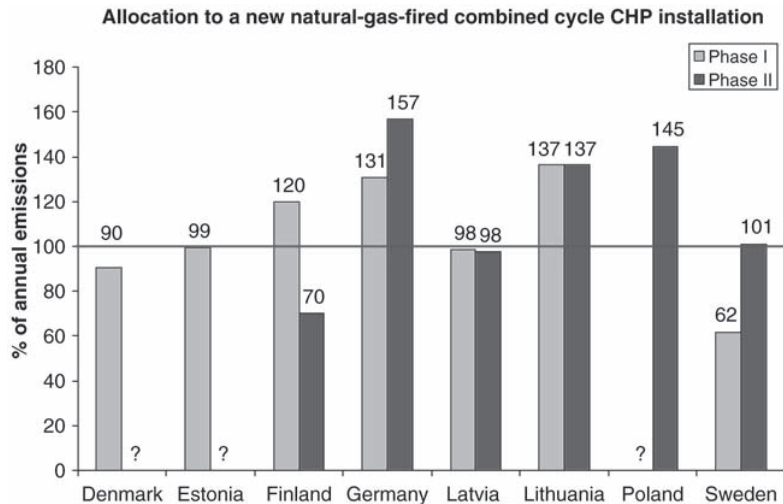
Figure 2. Allocation to a new natural-gas-based combined-cycle CHP plant assuming location in different countries. Results presented as percentage of expected annual emissions, i.e. the number of allowances required for compliance, covered by the allocation, with the horizontal line representing 100% of allowances needed for compliance. NAP I of Poland and NAP II of Estonia did not contain enough information to allow calculation of the allocation, while Denmark had not notified its NAP II to the European Commission at the time of writing (December 2006).

In the Commission decision for the first phase this was not commented on, whereas the second phase Commission decision includes language that can be interpreted as disallowing this rule.

In some Member States the regulator produces forecasts, while other Member States rely on operators to provide forecasts on which the allocation is based. This creates potentially large differences in the allocation even if the principles on which it is based are the same.

We have also found some differences between our calculated results and a previous study on allocation to energy installations (BALTREL, 2004). When analysing these differences, we found that BALTREL had applied different assumptions regarding, for instance, operating hours, efficiencies and emission factors. Some of these parameters are stated in the individual NAPs, some are to be submitted by the operator of the firm that applies for allocation. Furthermore, the BALTREL study used draft NAPs for some countries, which also may explain some of the discrepancies.

4. Does the allocation matter?

The value of the allocated allowances compared to other costs and sources of revenue of the firm is a key factor in determining whether the allocation actually has an impact on investment decisions and to what extent the observed differences between countries can distort competition.

This section illustrates the relative importance of the allocation by comparing the monetary value of the allocation to the fixed costs associated with the installations, and to the expected annual revenue from sales of electricity on the Nordic electricity market.

Table 2 shows the absolute value of the allocation to the two standard installations discussed above in different Member States. In Figures 3 and 4 these values are compared to the annualized fixed costs of the installations that are shown in Table 3, i.e. the annualized investment costs plus fixed operation and maintenance costs. Finally, in Figure 5, the value of the allocation to the condensing plant is compared to estimated annual revenues from sales of electricity from the installation.

Table 2. Value of annual allocation for the two standard installations (€million, EAU price €10)

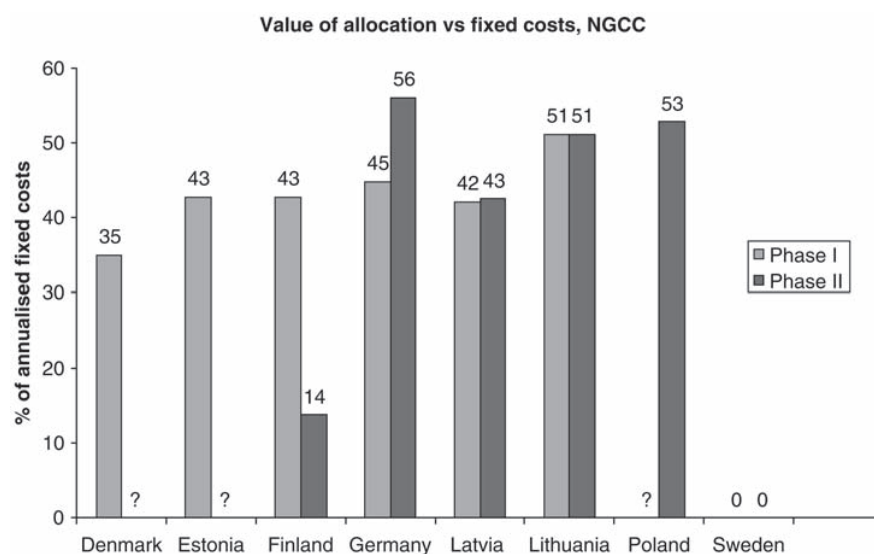|	Value of annual allocation (€million/year)			
	NGCC		CHP	
	Phase I	Phase II	Phase I	Phase II
Denmark	6.8	n.a.	5.5	n.a.
Estonia	8.4	?	6.0	?
Finland	8.4	2.7	7.3	4.3
Germany	8.8	11.0	7.9	9.5
Latvia	8.2	8.3	6.0	5.9
Lithuania	10.0	10.0	8.3	8.3
Poland	?	10.3	?	8.8
Sweden	0.0	0.0	3.7	6.1

Figure 3. The value of the annual allocation shown as percentage of estimated annualized fixed costs of a natural-gas-fired condensing installation. Assumed real interest rate is 6%, depreciation time 20 years. EAU price €10. Data on investment costs and fixed operation and maintenance taken from Elforsk (2003).
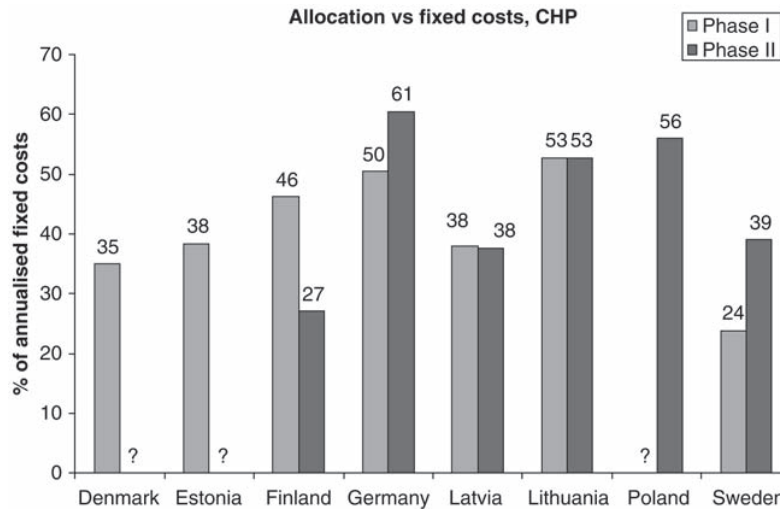
Figure 4. The value of the annual allocation shown as percentage of estimated annualized fixed costs of a natural-gas-fired combined-cycle CHP installation. Assumed real interest rate is 6%, depreciation time 20 years. EAU price €10. Data on investment costs and fixed operation and maintenance taken from Elforsk (2003).

Table 3. Estimated selected costs and revenues. The calculations on investment costs are based on data from Elforsk (2003)

Fixed annual costs NGCC	€19.5 million
Fixed annual costs gas-fired CHP	€15.7 million
Annual sales revenues NGCC	€74.4 million
Underlying assumptions	
Depreciation rate:	20 years
Real interest rate	6%
Investment costs NGCC	€560,000/MWe
Investment costs gas-fired CHP	€690,000/MWe
Fixed operation and maintenance costs	2% of investment cost
Power price	€31/MWh
Annual power generation NGCC	2.4 TWh

5. Does the allocation to new entrants affect investment decisions?

At the core of this study lies the hypothesis that the allocation does have an impact on investment decisions and competitiveness. However, this hypothesis can be challenged. The EU Commission has stated (Zapfel, 2005) that it is the price of allowances, not the allocation, which should drive new investment in CO_2-efficient technologies and changes in behaviour. According to this position, the allocation is a way to compensate for sunk costs and to facilitate the introduction of the trading scheme, not an instrument to drive technological change or strengthen the competitiveness of industry.
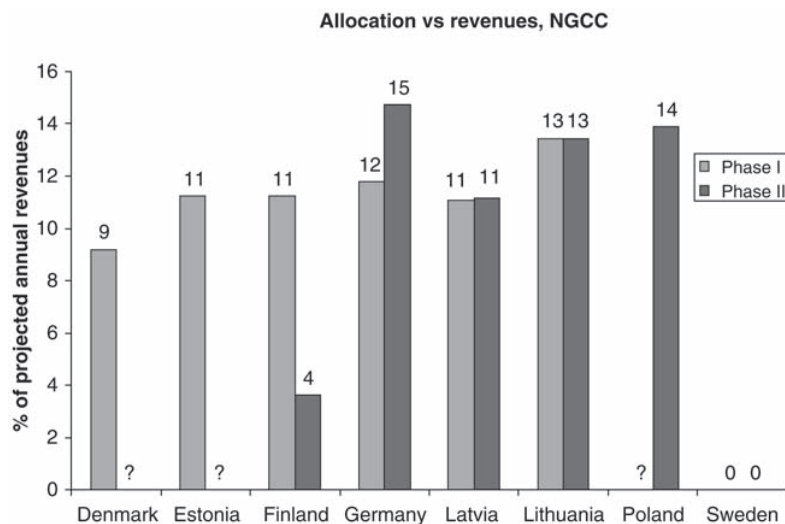
Figure 5. The value of the annual allocation shown as percentage of estimated annual sales revenue from electricity for the standard NGCC installation. Assuming 2,400 GWh annual electricity generation, allowance price €10, electricity price €31/MWh.

Behind this lies the view that since the allocation does not affect the variable costs for an installation, it has no significant impact on the competitiveness. However, this reasoning only holds as long as the allocation is not conditioned on the choices of an investor (for instance the choice to start operations or the choice of fuel or technology) and does not significantly affect the cost of capital for an operator.

The first assumption is negated by the fact that an operator can indeed affect the allocation to a new entrant: first through the decision to start operations at all, and then, depending on what allocation methodology is used, the allocation could be affected through choice of fuel, technology etc. In this respect, the allocation to a new entrant is different from allocation to existing installations based on historical measures, since the allocation can be directly affected by the operator's investment decisions.

The second assumption, that the capital costs are not a barrier to investment, is also unlikely to hold true. As discussed above, capital is a scarce resource for most firms, although the significance of this scarcity may vary between firms and settings.

Furthermore, the value of the allocated allowances in relation to other costs must be understood in order to determine its importance for investment decisions. Simply put; if the allocation represents a significant source of revenue for an installation, and the value of the allocation depends on the investment decisions, the allocation is likely to affect the decisions taken by the investor.

For a more formal presentation of incentive distortion caused by different allocation methodologies see, e.g., Sterner and Muller (2006). A general conclusion is similar to the one suggested above: when the firm can influence the allocation, the allocation does affect the firm's behaviour.

In Table 2, the annual values of the allocated allowances are presented. As shown in Figure 4, the annual value of the allocation is comparable to the estimated annualized investment cost for

the installation. This implies that the allocation is an important source of revenue for the operator of an installation. It is also worth pointing out that the results are very sensitive to the EAU price. If the price increases from €10 to levels around €20 (which is close to the average price during the first two years of the trading scheme) the value of the allocation will in fact be higher than the fixed investment costs in several Member States. In addition, we see significant differences between countries in the outcome of the allocation, thus distorting competition between countries.

However, in order to get a more complete assessment of the importance of the allocation, several other factors have to be taken into account. While a full analysis is beyond the scope of this article, a few issues that are likely to be important can be pointed out:

– Uncertainty in the allocation. The operator can only be certain of receiving a 5-year allocation under current rules.
– Variable costs, such as fuel prices and salaries.
– Market factors such as energy prices and access to customers vary over time and between countries, which affects the role of the allocation in determining the investment decision. The large differences in power prices that currently exist between Germany and the Nordic countries are likely to have a significant impact on investment decisions.
– Uncertainty in other energy policies. The European energy system is affected by a wide range of policy instruments that differ between Member States, for instance subsidies for renewables, guaranteed feed-in tariffs, green certificates and restrictions on nuclear power. Many of these policies are important for ensuring which fuels or technologies are profitable, and they are all subject to more or less predictable changes.

Therefore it is difficult to precisely estimate to what extent the allocation will impact the decisions on where to invest and in what type of installations. But all this said, we conclude that the sheer magnitude and value of the allocation, in combination with the incentive structure the current methodologies provide, makes it very likely that the allocation to new entrants does affect investment decisions and competitiveness of countries, firms and technologies in northern Europe.

6. Transition from new entrants to existing installations

The treatment of new entrants is closely related to the general issue of updating allowance allocations over time. Although this does not directly affect the competitiveness between Member States, it is an important issue for the efficiency of the trading scheme as a whole. As such, it should be addressed when discussing how to harmonize the allocation rules between the Nordic countries.

The questions are how long should a new installation receive allocation according to some special allocation methodology before it is regarded as an existing installation, and how can the transition between different allocation methodologies be made without providing incentives for firms to act strategically in order to increase their future allocations. According to the EU Directive on Emissions Trading (European Union, 2003) a 'new entrant' is an installation that starts its operations after the NAP has been submitted to the EU Commission.[4] This would suggest that a new installation could only be regarded as a new entrant for one trading period.

In general, the economics literature indicates that changing or updating allowance allocations over time may have a distorting effect on company decisions. For example, Burtraw (2001) and

Fischer (2001) found that updating output-based allocation methodologies serves as an economically inefficient subsidy for production that lowers product prices for consumers. Similarly, in an analysis of a potential emissions trading programme in Alberta, Canada, Haites (2003) found that an output-based updated allocation provides an incentive for production.

These considerations have clearly guided the Commission's prohibition on updating within each phase of the EU ETS. Nevertheless, it is not clear how this should be applied to the treatment of new entrants. One option suggested by Åhman et al. (2006) would be to introduce a 10-year time delay in the allocation. Under such a scheme, a new entrant would first be allocated based on some projected measures, but then, after 10 years, the allocation would be updated. For instance, an installation starting in operation in 2006 would receive free allocation based on forecasts until 2015. From 2016 onwards the allocation would be based on actual activity 10 years previously. A 10-year time delay would significantly weaken the tendency of updating to produce perverse incentives for operators and thus reduce some of the distortions of free allowance allocation. A similar approach, but with a 4-year time delay, is used in the US NO_xSIP Call, a programme that requires summertime reductions in NO_x in the eastern half of the USA.[5]

7. What about auctioning?

According to the Emissions Trading Directive (European Union, 2004), a Member State only has to explain how new entrants can gain access to allowances; it does not need to provide special allocation to them. Furthermore, up to 10% of the total volume of allowances may be sold, for instance in an auction, in the second trading period. This means that there would be room to use full auctioning of allowances to new entrants. The Directive also allows for forcing new entrants to buy allowances on the open market. The discussion regarding free allocation or not to new entrants is analogous to the one on whether new entrants are discriminated against compared with existing installations, presented above. For a full discussion on the advantages of auctioning versus grandfathering, see, for instance, Cramton and Kerr (2002) and Hepburn et al. (2006). The major advantage of auctioning is that it provides efficient incentives for investments in efficient technologies and low-carbon fuels. It also eliminates the creation of windfall profits in the energy sector. Moreover, it avoids the problems of obtaining accurate data that are associated with benchmarking methodologies. However, if allocation is not harmonized across Member States with a common energy market, competition for investments may be distorted. Thus, if other northern European countries continue to apply free allocation, this adds weight to the arguments favouring free allocation to new electricity producers in the Nordic countries.

8. Framing the discussion: input from authorities and industry

In order to gain a better understanding of the rationale behind the current allocation methodologies and the potential changes that could be possible and politically realistic in future phases, we sought input from policy-makers in Denmark (Sigurd Lauge Pedersen), Finland (Timo Ritonummi) and Sweden (Truls Borgström). We also interviewed representatives from the energy associations from the respective countries: Danish Energy Companies (Charlotte Söndergren), Finnish Energy Industries (Jukka Leskelä) and Swedenergy (Maria Sunér Fleming).

All of the people we interviewed have extensive experience of working with the design of phase I NAPs, and several were also involved in phase II NAPs. The respondents acted in their personal capacity and were not asked to give the official position of their respective countries or industry associations. For reasons of confidentiality we have summarized the respondents' comments in the sections below, only disclosing individual views in a few cases. At the time of the interviews, no Member States had notified their phase II NAPs to the European Commission. Thus the views expressed primarily refer to phase I allocation.

8.1. Views from policy-makers

There seems to be agreement that, should the current allocation methodologies remain unaltered, it will distort competition between the Nordic countries. However, since the allocation is only determined for a short period of time in relation to the life-span of a power plant, it is difficult to judge the importance of the allocation compared with other factors determining investment decisions. Furthermore, both Sigurd Lauge Pedersen and Truls Borgström expect that the allocation will be decreased in future trading periods, and thus its importance and impact on competitiveness will also decrease.

There seem to be no fundamental or principal reasons that would prohibit the countries from adjusting their allocation principles in order to harmonize them with each other. The main barriers are probably political. What other Member States, in particular Germany, do is very important for what is feasible in the Nordic countries.

None of the respondents ruled out the option to force new entrants to pay for their allowances, although this will probably be politically difficult to pursue if other neighbouring countries continue with free allocation.

8.2. Views from industry

All respondents believed that the current allocation methodologies distort competition and that harmonizing them is of high priority. However, allocation is expected to be decreased in coming trading periods, and thus this effect is likely to decrease also. All respondents also pointed out that there are differences in other energy policies between the Nordic countries that strongly affect investment decisions, including taxation and application process. There is a need to harmonize other policies as well in order to achieve a level playing field.

There seem to be no fundamental barriers to adjusting the allocation principles in future phases. However, a critical condition for almost any changes is that the Nordic countries would in fact implement harmonized allocation methodologies. Further, since all allocation methodologies create winners and losers, it may be difficult to get general support for any one system. Auctioning would probably meet great resistance unless at least Germany, Poland and Estonia also radically decreased the allocation to new entrants.

The first priority, according to all respondents, is to get a harmonized system, preferably across the entire EU or at least on the northern European energy market. A harmonized Nordic system would be a step forward, but the approach used in particular in Germany should be considered. The exact design of the allocation is important, but is a secondary priority. It would probably be easiest to get wide support for a common benchmarking methodology that would take technology and fuel into account, thus avoiding the creation of major winners and losers, although not all respondents favoured this methodology.

9. Conclusions

Given the current allocation methodologies, and the discussion above, several options for a harmonized allocation methodology exist. But first, a few general conclusions can be drawn:

– Current allocation rules do have an impact on investment decisions, and can significantly distort competition if they remain unchanged.
– Under current allocation rules, the annual value of the allocation is comparable to the fixed investment costs for a new installation. Furthermore, it is not insignificant compared with expected revenues from sales of energy from the installation.
– There seem to be no fundamental obstacles in any Nordic country to changing the allocation system to new entrants as part of a harmonizing process.
– Although it would be an important accomplishment if Denmark, Finland and Sweden could harmonize their allocation methodologies, the Nordic countries must also consider policies in other neighbouring countries when deciding allocation methodology. This is also stressed by both policy-makers and industry representatives. Although the transmission capacities between the Nordic countries are significantly higher than they are to other countries, the Nordic energy sector is already part of the larger northern European energy market. Germany, Poland and Estonia are of particular importance, since the transmission capacities to those countries are relatively large.
– Harmonizing allocation is a higher priority for electricity generation than for heat, due to the higher sensitivity of electricity generators to competition.
– Since the energy sector can pass on the majority of the cost for emission allowances to clients, a stringent allocation is easier to justify in this sector than in others.

The primary reasons for allocating free allowances to new entrants in the energy sector are:

– *A level playing field vis-à-vis existing installations.* Under current regulations on closures, Sweden excepted, incumbents are favoured over new entrants. If rules on closures are changed so that a plant that closes does not lose its allocation, this argument fails.
– *A level playing field vis-à-vis neighbouring countries.* As long as neighbouring countries (in particular Germany, Poland and Estonia) allocate free allowances to new entrants, there may be a reason to allocate free allowances to electricity producers in order to avoid discouraging investments in the Nordic electricity sector. For heat generation, the argument is less relevant since the market is local, although, in the choice between investing in heat generation in two different countries, the allocation may be a factor if capital is a constraining factor.
– *Stimulating investments in new capacity.* Since capital is a scarce resource, allocating free allowances may have a positive impact on the rate of new investments. However, setting a 'correct' level of subsidy is difficult, and subsidizing investments through the allocation risks distorting the market in other ways.

Although we include both electricity and heat generation in the 'energy sector', the need for a harmonized allocation methodology is greatest for electricity generation. However, as heat and power are often co-generated, and to some extent can be substituted for one another, a harmonized allocation methodology for heat generation would also carry advantages.

A few observations can also be made regarding the changes in allocation between phase I and phase II. First, no harmonization of allocation methodologies between Member States can be detected.[6] Rather, there are still striking differences in how Member States allocate allowances to new entrants. This includes differences in general principles, but also in underlying assumptions such as emission factors and activity rates for energy installations.

Second, the use of fuel- and technology-specific benchmarks is still widespread. No Member State applies a uniform benchmark regardless of fuel or technology used.

Third, Finland is the only Member State that significantly reduces the allocation to new entrants. Instead, we see important increases in the allocation to new entrants in Germany and Sweden.

These findings are particularly disturbing when considering the growing economics literature showing that the efficiency of EU ETS would benefit from a much more stringent allocation to new entrants, as well as from more harmonized allocation methodologies across Member States, fuels and technologies (e.g. Åhman et al., 2006; Gagelmann, 2006; Grubb and Neuhoff, 2006; Hepburn et al., 2006; Betz et al., 2006, this issue).

We conclude that the preferred and most cost-effective solution would be that the Nordic countries do not allocate free allowances to new entrants in the energy sector. It is crucial to avoid updating, i.e. allocations must not depend on some variable that the firms can influence. Instead operators would have to buy allowances, either from the government or on the open market. Combined with adjusted rules on allocation to existing installations and to installations that close, this would give the most efficient incentives for new investment. However, this solution may not be the most appropriate, or even efficient, if Germany and Poland[7] do not follow or radically decrease their allocation to new entrants.

A full discussion on allocation rules to existing installations is beyond the scope of this article. However, a restrictive allocation to the entire energy sector can be justified considering the possibility of energy producers to pass on costs to clients. Furthermore, a level playing field between existing and new installations and between technologies would be achieved if auctioning to incumbents was applied, if the rules on closures were changed or if identical, fuel- and technology-independent, benchmarks were used for both existing and new installations.

This solution would eliminate the distortion of competition between the Nordic countries and decrease windfall profits created by the allocation to the energy sector.[8] It would also provide incentives to invest in efficient technologies and low-carbon fuels.

Free allocation to electricity producers in the Nordic countries may be justified in order to avoid distorting the competition in relation to Germany, Estonia and Poland if those countries continue with a generous allocation to new electricity producers, and if the rules on allocation to incumbent emitters and closures remain unaltered. In such a scenario, a second option would be to use harmonized fuel- and technology-independent benchmarks based on output. The allocation should be kept as restrictive as possible, in particular for heat producers. There should also be harmonized assumptions and guidelines on how to forecast production and, if compliance factors are used, of course these should be harmonized as well. This solution, while definitely in the realm of second best, would eliminate the distortion of competition between the Nordic countries created by the allocation. It would also give incentives to invest in low-carbon fuels and efficient technology. However, there would still be windfall profits created by the allocation to energy producers. Moreover, if other neighbouring northern European countries continue to use fuel- and/or technology-specific benchmarks, there will still be distortion of competition between those countries and the Nordic countries. However, the benefits of preserving correct incentives for investments

could well be greater than the potentially negative effects created by some distortion in competition. A 'race to the bottom', where the Nordic countries apply allocation methodologies that create perverse incentives for fear of losing investments to neighbouring countries, would be regrettable. In the long term, this would risk shifting the structure of the energy system in the wrong direction. Furthermore, the magnitude of the distortion in competition, and the impact this will have on investments, depends not only on the allocation principle but also on the actual level of allocation.

A third option would be to use harmonized, fuel- and/or technology-dependent benchmarks, keeping the allocation as restrictive as possible. This would probably meet less resistance from industry and some policy-makers than the first or second options. It would also fulfil the objective of removing distortion of competition from the allocation. However, the incentives to invest in low-carbon fuels and efficient technologies would be reduced, and windfall profits would still be created by the allocation. In this case, the benefits of having harmonized allocation methodologies have to be weighed against the negative effects of not having incentives to invest in low-carbon energy generation.

9.1. Further research

Although the allocation as shown in this study has a large monetary value, it is only determined for 5 years into the future. Considering the long investment cycles in the energy sector, increased certainty over the allocation would carry significant benefits. If auctioning is used for the allocation to existing installations as well as to new entrants in the energy sector, this issue is dealt with. An area identified for future research is therefore to explore the options to provide higher certainty, for instance by extending the allocation periods.

This also relates to the issue of how the transition from status as a new entrant to existing installation is to be done. If free allocation to existing installations is kept, we find that a solution where a new installation is treated as a new entrant in the allocation for two successive trading periods would be preferable to the current regulations. It would weaken the perverse incentives to increase production and/or emissions created by updating the allocation at the beginning of each trading period. For further discussion of this topic, see Åhman et al. (2006). An argument against an approach with longer allocation periods would be if there is an intention to move away from free allocation in future trading periods. Such a transition may be more difficult to make if allocation to certain installations is determined many years into the future. As we find that auctioning carries significant advantages over free allocation, there is a need for research that could facilitate a transition to such a scheme. The power of policy and political path dependency should not be underestimated, which adds to the urgency of changing the allocation system.

Finally, a question that is beyond the scope of this article but is important to understand, is *why* it is important to have the same incentives for new investments across Member States and to what extent this objective should be given priority over others. In an integrated electricity market such as that in the Nordic countries, harmonizing incentives makes economic sense and is intuitively appealing. But when markets are only semi-integrated, like the northern European electricity market, or even completely separated like the market for district heating, there may be other considerations that are more important when designing the allocation, for instance security of supply, volatility in energy prices, and the structure of the energy system. Thus the interaction between the EU ETS and other policy instruments, and the potential trade-off between the objectives of the trading programme and other priorities, are areas where further research is needed.

Acknowledgements

Notes

1 One reviewer suggested a countervailing effect: the volatility of sales revenue minus allowance costs and thus investment risk could be lower if electricity prices are positively correlated with allowance prices and allowances are auctioned frequently.

2 If a common benchmark is used for all fossil-based energy generation, but no allocation is given to energy based on biofuels, as is the dominating methodology in the EU ETS, this is an example of fuel-specific allocation, since it discriminates between different categories of fuels.

3 In the case of CHP, it has been argued that two reasons can justify generous allocation: first, generation of energy is more efficient in CHPs than in condensing plants and, second, heat generated in CHPs competes with small-scale heating installations that are not covered by the EU ETS. The first argument is weak as long as each technology bears the full costs of production; a higher efficiency is then rewarded in the market. The second argument is relevant, but it can be questioned whether one distortion should be fixed by creating another.

4 The technical definition is given in Article 3(h) of the Directive: 'Any installation carrying out one or more of the activities indicated in Annex I, which has obtained a greenhouse gas emissions permit or an update of its greenhouse gas emissions permit because of change in the nature or function or an extension of the installations, subsequent to the notification to the Commission of the national allocation plan.'

5 Of course, there are risks associated with longer allocation periods; committing to policies that turn out to be bad, for a long time into the future, can be worse than updating. Since the EU ETS is still in its infancy, there may be reasons to conduct more frequent reviews, although it should be a priority to keep correct incentives in place.

6 This said, one can speculate whether individual Member States may have adjusted their allocation methodology in phase II in response to how other Member States set up their allocation in phase I. Two possible examples of this are the significant increase in allocation to new CHPs in Sweden and the introduction of compliance factors in Finland.

7 We base our conclusion on the figures given in the Polish NAP submitted to the Commission in July 2004. We have not been able to obtain any official confirmation of changes to these in response to the Commission decision.

8 However, it would not address 'secondary' windfall profits created by electricity producers being able to sell *all* electricity, not just fossil-based, at higher price due to the price on carbon. This secondary effect is probably at least as important as direct windfalls from the allocation.

References

Åhman, M., Holmgren, K., 2006. Harmonising New Entrant Allocation in the Nordic Energy Sectors: Current Principles and Options for Phase II. TemaNord 2006:515, Report for the Nordic Council of Ministers, Copenhagen.

Åhman, M., Zetterberg, L., 2005. Options for allowance allocation under the EU emissions trading directive. Mitigation and Adaptation Strategies for Global Change 10, 597–645.

Åhman, M., Burtraw, D., Kruger, J., Zetterberg, L., 2006. A Ten-Year Rule to guide the allocation of EU emission allowances. Energy Policy 35(3), 1718–1730.

BALTREL, 2004. Emission Trading Task Force of BALTREL: Final Report. Presented at BALTREL Committee meeting, Riga, 4 November 2004.

Betz, R., Rogge, K., Schleich, J., 2006. EU emissions trading: an early analysis of national allocation plans for 2008–2012. Climate Policy 6(4), 361–394.

Bode, S., Hubl, L., Schaffner, J., Twelemann, S., 2005. Discrimination Against Newcomers: Impacts of the German Emission Trading Regime on the Electricity Sector, Hamburgisches Welt-Wirtschafts-Archiv Discussion Paper 316.

Burtraw, D., 2001. The Effect of Allowance Allocation on the Cost of Carbon Emission Trading. Resources for the Future Discussion Paper 01-30. RFF, Washington, DC.

Cramton, P., Kerr, S., 2002. Tradable carbon permit auctions: how and why to auction not grandfather. Energy Policy 30, 333–345.

DEHSt, 2005. Implementation of Emissions Trading in the EU: National Allocation Plans of All EU States. UBA, Berlin.

Ecofys, 2004. Analysis of the National Allocation Plans for the EU Emissions Trading Scheme. Ecofys Report, London.

Elforsk, 2003. El från nya anläggningar, Elforsk, Stockholm.

Ekström, C., Bröms, G., Eidensten, L., Hammarberg, A., Herbert, P., Kapper, R., Krohn, P., Larsson, S., Rydberg, S., Nyström. O., Olsson, F., 2006. Kostnader och potential för att minska CO2-utsläppen i Sverige. Elforsk-report 05:47, Stockholm.

European Union, 2003. Directive 2003/87/EC of the European Parliament and the Council of 13 October 2003 establishing a Scheme for Greenhouse Gas Emission Allowance Trading within the Community and amending the Council Directive 96/61/EC.

European Union, 2004. COM (08/III/2005). Commission Decision concerning the National Allocation Plan for the Allocation of Greenhouse Gas Emission Allowances notified by Poland in accordance with Directive 2003/87/EC of the European Parliament and of the Council.

Fischer, C., 2001. Rebating Environmental Policy Revenues: Output-Based Allocations and Tradable Performance Standards, Resources for the Future Discussion Paper 01-22. RFF, Washington, DC.

Gagelmann, F., 2006. Innovation effects of tradable emission allowance schemes: the treatment of new entrants and shutdowns. UFZ (Centre for Environmental Research), Department of Economics Discussion Papers (4/2006).

Göteborg Energi, 2005. Definition of New Power Plant under Construction, November 2005. [Available at http://www.goteborgenergi.se/Om_Goteborg_Energi/Var_verksamhet/Rya_Kraftvarmeverk_DXNI-5363_.aspx].

Grubb, M., Neuhoff, K., 2006. Allocation and competitiveness in the EU Emissions Trading Scheme: policy overview. Climate Policy 6(1), 7–30.

Haites, E., 2003. Output-based allocation as a form of protection for internationally competitive industries, Climate Policy 3(Supplement 2), S29–S41.

Haites, E., Hornung, R., 1999. Analysis of Options for Gratis Distribution of Allowances, National Roundtable on the Environment and the Economy, Canada.

Harrison, D., Radov, D., 2002. Evaluation of Alternative Initial Allocation Mechanisms in a European Union Greenhouse Gas Emissions Allowance Trading Scheme, National Economic Research Associates, prepared for DG Environment, European Commission.

Hepburn, C., Grubb, M., Neuhoff, K., Matthes, F., Tse, M., 2006. Auctioning of EU ETS phase II allowances: how and why? Climate Policy 6(1), 137–160.

Kolshus, H.H., Torvanger, A., 2005. Analysis of EU Member States' National Allocation Plans, Working Paper No. 2, CICERO, Oslo.

Matthes, F., Graichen, V., Repenning, J., Doble, C., Macadam, J., Taylor, S., Zanoni, D., Chodor, M., 2005. The Environmental Effectiveness and Economic Efficiency of the European Union Emissions Trading Scheme: Structural Aspects of Allocation. WWF Report.

Neuhoff, K., Keats Martinez, K., Sato, M., 2006. Allocation, incentives and distortions: the impact of EU ETS emissions allowance allocations to the electricity sector. Climate Policy 6(1), 73–91.

Schleich, J., Betz, R., 2005. Incentives for energy efficiency and innovation in the European emission trading system. In: European Council for an Energy-Efficient Economy (Paris): Proceedings of the 2005 ECEEE Summer Study. Energy Savings: What Works & Who Delivers? ECEEE, Mandelieu, Côte d'Azur, France.

Sterner, T., Muller, A., 2006. Output and abatement effects of allocation readjustment in permit trade. RFF Discussion Paper 06-49. Resources for the Future, Washington, DC.

Zapfel, P., 2005. Personal communication 24 October. DG Environment, EU Commission.

Paper VI

Implications of announced phase II national allocation plans for the EU ETS

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Abstract

We quantified the volume of free allowances that different national allocation plans proposed to allocate to existing and new installations, with specific reference to the power sector. Most countries continue to allocate based on historic emissions, contrary to hopes for improved allocation methods, with allocations to installations frequently based on 2005 emission data; this may strengthen the belief in the private sector that emissions in the coming years will influence their subsequent allowance allocation. Allocations to new installations provide high and frequently fuel-differentiated subsidies, risking significant distortions to investment choices. Thus, in addition to supplying a long market in aggregate, proposed allocation plans reveal continuing diverse problems, including perverse incentives. To ensure the effectiveness of the EU ETS in the future, the private sector will need to be shown credible evidence that free allowance allocation will be drastically reduced post-2012, or that these problems will be addressed in some other way.

Keywords: Emissions trading; National allocation plans; Comparison; European Member States

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1. Introduction

The EU Emissions Trading Scheme is designed to cap emissions of energy-intensive industry in Europe. Under the European Directive on Emissions Trading, each Member State is required to state within the proposed national allocation plan (NAP), both the allocation volume of emissions allowances to the covered sectors and the allocation methodology.

Sensible decisions on the allocation volume or 'cap' level by Member States are crucial. Stringent caps create scarcity, which holds the key to both the environmental efficacy of the scheme and good functioning of the CO_2 market. Yet, Neuhoff et al. (2006a, this issue) argue that the volume of allowances allocated under the currently proposed NAPs for phase II is too high, by comparing the NAPs with CO_2 emissions projection scenarios and the historic trend of emissions extrapolated forward. The analysis by Betz et al. (2006, this issue) also shows that, in many Member States, allocation for phase II is excessive relative to 2005 emissions, historic trends and country-level projections.

The national allocation plans also have to specify how the allowances are distributed among existing installations, new installations and auctions. Betz et al. (2006, this issue) analyse how the different approaches selected by Member States increase complexity and reduce transparency of the overall system. Much of this complexity arises from industry interests and aims to address distributional concerns. However, the complexity not only complicates participation by industry, but also complicates the role of NGOs and less-informed industrial sectors in controlling the outcome of the political process. Thus the need for harmonization in the methodology used across the Member States has been widely emphasized (e.g. del Rio González, 2006, this issue).

Economic theory and anecdotal evidence also suggest that the methodology chosen for the allowance allocation can directly influence decisions on investment, retrofitting and plant operation.

Fischer (2001) looks at the effects of output-based allocation, Harrison and Radov (2002) provide a policy-oriented description of allocation options, Palmer and Burtraw (2004) focus on distribution effects, Böhringer and Lange (2005) assess macro implications of increased allowance imports from inefficient emission-based allocation, Burtraw et al. (2005) assess the impacts of partial coverage by emissions trading schemes, Entec and NERA (2005) discuss benchmarks, and Matthes et al. (2005) assess distortions in phase I allocation plans, Åhman et al. (2007) argue for longer allocation periods and Bartels and Müsgens (2006) simulate the impact of the German allocation plans on investment decisions. A theoretical framework for the distortions from allocation procedures is provided by Neuhoff et al. (2005) and Sterner and Muller (2006).

This article compares the allocation methodologies envisaged in the different national allocation plans for the period 2008–2012 that have been submitted to the Commission or alternatively which were available in draft format at the end of December 2006.

Taking the power sector as an example, in this study we quantify the differences in free allowance allocation across Member States and across generation technologies.

– For new installations, investment incentives are distorted towards fossil-fuel generation and in many Member States even more towards CO_2-intensive fuel types. This reduces the effectiveness of EU ETS in reducing CO_2 emissions and compliance costs. Free allocation also represents output subsidies and might thus undermine substitution effects to less CO_2-intensive products.

The free allocation of allowances to new installations, with all its negative implications, is unique to the EU ETS, except for a few 'set asides' in the US NO_X programme.

– For existing installations, the heterogeneous allocation can distort the merit order choice, incentives for energy efficiency improvements and closure decisions, again detrimental for the cost-efficiency of the EU ETS in reducing CO_2 emissions. This effect is even stronger if *ex-post* adjustment is allowed. This is under legal dispute between Germany and the European Commission and is still demanded by some Member States. The large differences across countries again illustrate that significant reductions in free allowance allocation are not only economically, but also politically, possible.

A reduction in free allowance allocation in the period 2008–2012 could be used both to reduce the overall level of free allocation to the covered sector and also to increase the share of auctions. In its approval of the first 10 national allocation plans, the EU Commission made auctioning the one area of discretionary flexibility within its decision and where the deadline of 31 December 2006 does not apply. 'Without prior acceptance by the Commission', Member States can increase the share of auctioning up to 10%.

Distortions from the free allocation to existing facilities mainly occur because without credible government commitments about future allocation, owners and operators expect that the future allocation will be similar to the current approach. For example, allocation in many Member States factors installation-level emissions data from 2005 into allocation for 2008–2012. Thus, market participants expect future allocations to be similarly 'updated' over time – large emissions today will be rewarded by large volumes of free allocation in the future. This will distort their investment and operational choices (early action problem) and lead to a less cost-efficient outcome with higher costs to society.

A movement towards less distorting allocation methods increases confidence in non-distorting future allocation methods. The use of auctions and the initial use of benchmarks in some sectors and countries represent a promising start. A strong commitment to rapid phasing-out of free allowance allocation post-2012 could avoid most distortions. A thorough assessment of the free allocation under EC Law State Aid criteria could conclude that the continued allocation post-2012 would offer a disproportionate benefit relative to the cost of the environmental regulation (Johnston, 2006). This could provide a credible commitment towards phasing-out free allocation and thus address the early action problem.

This article does not address closure conditions. The expectation of receiving future allowances within the commitment period or in the next commitment period only with the continued availability or operation of a power station creates an incentive to postpone the retirement of power stations or to invest in the retrofit of power plants rather than closing the power station (Spulber, 1985; Neuhoff et al., 2006b). This distortion is only partially compensated for by transfer provisions (Gagelmann, 2006). Åhman and Holmgren (2006) and Betz et al. (2006, this issue) compare such closure and transfer provisions across Member States.

We summarize the information contained in the currently proposed (28 December 2006) national allocation plans for the period 2008–2012 and present it for general scrutiny in an Excel database.[1] Our first findings regarding the economic effects that might follow from these plans are summarized below. Please note that some national allocation plans still require approval by the Commission, while most of the national allocation plans assessed required some adjustments which are discussed within the Member States.
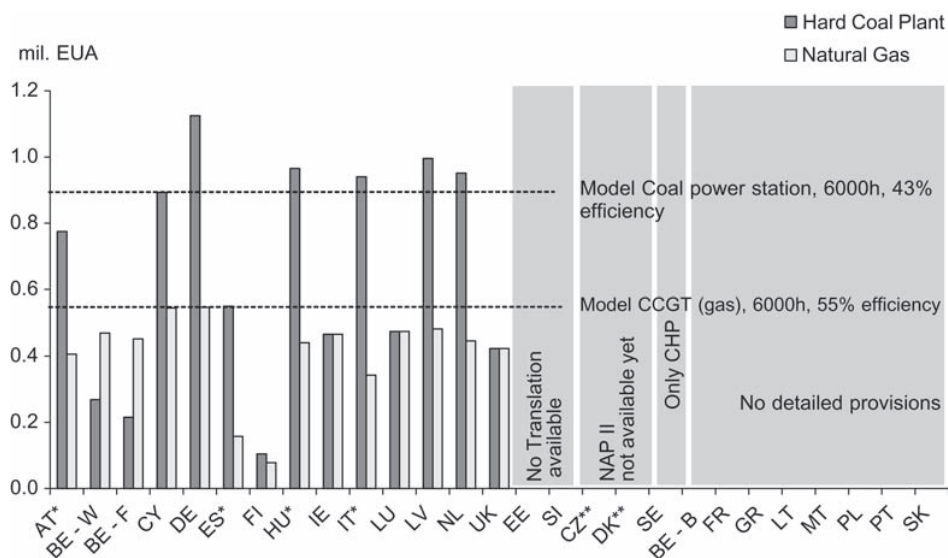
Figure 1. Comparison of new entrant allocation. * Draft NAP, ** NAP not available. In all Figures and Tables the regions of Belgium are identified as BE-F – Flemish, BE-W – Walloon, BE-B – Brussels. A different load factor is used for the UK (5,812 h for both coal and gas), as indicated in NAP II. In Germany (DE) a standardized load factor of 7,500 h is applied to all power plants.

2. Quantity of free allocation to installations – using the power sector as an example

To illustrate the distortions of free allocation to new investment, we calculate the subsidy that it constitutes for new-build coal and gas power stations[2] across the Member States.[3] To facilitate the comparison we assumed that a model power station runs for 6,000 h per year. Figure 1 illustrates that in all Member States, fossil-fuel generators receive high subsidies in terms of free new entrant allocation. In many countries, a new entrant allocation covers the emissions of CCGT gas plants, and in some countries it even covers all the emissions that a coal power station is expected to produce. While it is sometimes argued that new power stations should receive the allowances that they require for covering their emissions, this is not in accordance with economic principles. In liberalized electricity markets, power generators pass the opportunity costs of CO_2 allowances into the electricity price (Burtraw et al., 2002; Neuhoff and Keats-Martinez, 2005) and thus do not require any free allocation. This is desirable in order to achieve substitution effects, and is only avoided where electricity price regulation covers just real costs and not opportunity costs (Burtraw et al., 2005).

Any free allocation represents a subsidy – and where only fossil-fuel generation is subsidized, this distorts investment choices in favour of fossil-fuel generation. Where coal receives a higher allocation than gas, the investment choice is, in addition, distorted towards coal. The level of such subsidies under proposed second-phase NAP is so high that the construction of coal power stations is more profitable under the ETS with such distorting allocation decisions than in the absence of the ETS (Åhman and Holmgren, 2006; Matthes et al., 2006; Neuhoff et al., 2006a, this issue). The long-run consequences of these distortions can be significant since, once built, plants will stay on

the system for many decades, significantly increasing the cost of shifting towards a low-carbon economy in the future (Bartels and Müsgens, 2006; Neuhoff et al., 2006b).

The German National Allocation Plan notified to the Commission not only provided the highest allocation for new coal generation in general, but the draft Allocation Law also contained a provision allowing an even higher free allocation for new lignite-fired installations. In addition, the current NAP guarantees the continuation of free fuel-specific allocation for 14 years. This would undermine investments in low-carbon technologies and has not been accepted by the EU Commission in its decision on the German NAP. Fixing the free allocation beyond the commitment period 2008–2012 would also reduce the flexibility to evolve climate policy in a national, European and global context in the coming decade and might pre-empt negotiations about future burden sharing between sectors or among European Member States. Since Germany has announced its intention to put climate change on the agenda of its Presidencies of the EU and the G8 in 2007, accepting changes on these long-term provisions would strengthen the German government's credibility for requesting more stringent emission targets from other countries.

Figure 2 illustrates the allocations per Member State to two standard types of existing power stations of 200 MW, assuming that they operate on average for 6,000 h/year. Once again, the large discrepancies between different EU Member States are striking. Also striking is how some countries can still justify large free allocation when others manage to negotiate with their industry a significantly lower level of free allowance allocation.

The high degree of free allocation to the power sector could easily be reduced without reducing power sector profits below pre-ETS levels (Pál and Bartek-Lesi, 2006). As these large demands by individual installations inflate the aggregate national and therefore the EU allowance budgets, such reductions are required in order to achieve a reduction of the EU ETS cap and thus to achieve the Kyoto emission target with sufficient scarcity for a strong price signal.
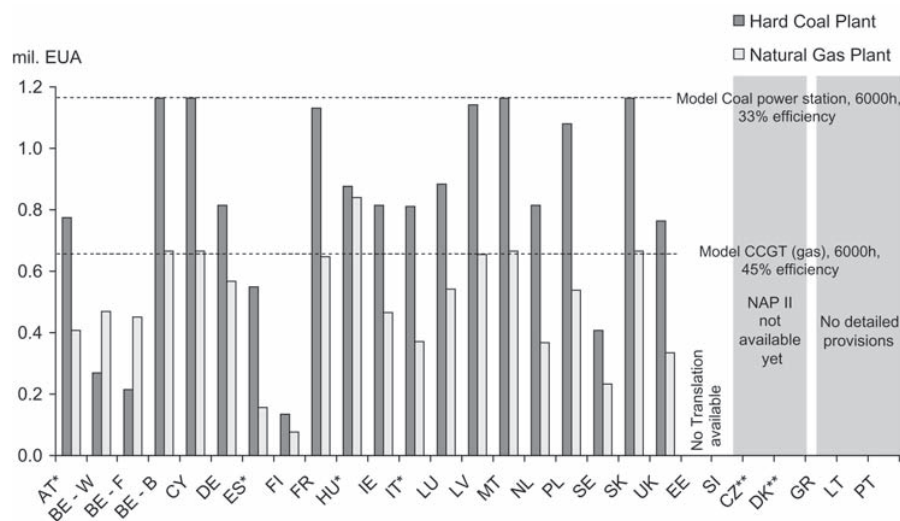

Figure 2. Comparison of allocations to existing facilities. * Draft NAP, ** NAP not available. Different load factors are used for BE-F (3,000 h for coal, 6,300 h for gas), ES (4,167 h for coal). PL – low SO_2-emitting installations. SE – adjustment factor assumed to be equal to 0.35.
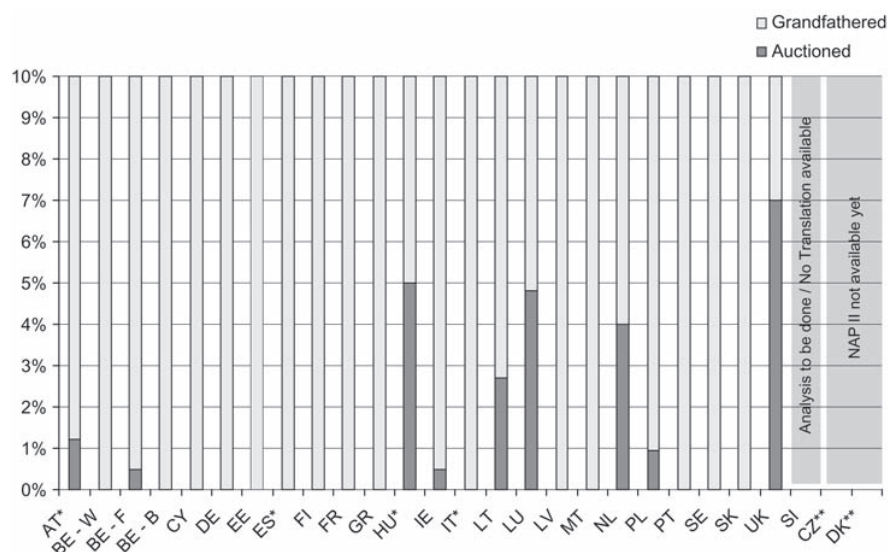
Figure 3. Comparison of the potential to extend the use of auctions between countries. * Draft NAP, ** NAP not available.

3. The use of auctioning

The EU Directive allows Member States to auction up to 10% of the allowances available. Figure 3 illustrates that all Member States can still make more use of this option. Making use of the 10% auction allowance in this phase would not only reduce distortions from the free allocation, but would also allow all parties involved to become comfortable with allowance auctions. Additionally, a minimum price auction could, by ensuring a price floor, further facilitate investment in low-carbon technologies (Hepburn et al., 2006). Auction revenues could be recycled to support industrial competitiveness and development and the initial deployment of suitable technologies. Furthermore, auctioning of significant amounts of allowances could support the transparency of the allowances market, especially in the settlement period, and avoid price volatilities resulting from asymmetric risk-hedging strategies between sectors which are short and sectors which hold long positions.

4. The use of CDM and JI credits

In the context of the overall Kyoto Protocol implementation framework, the linkage with the international trading scheme is another important dimension. With uncertainty over future demand for JI and CDM credits from Canada, Japan, other Annex I countries, and governments of the EU Member States themselves, some market participants anticipate that the European market could be flooded by these allowances to such an extent that the EU allowance price would plummet. Such uncertainty undermines investment certainty for low-carbon options and also poses obstacles to implementing a price floor using auctions.

Article 30(3) of the EU Directive on Emissions Trading requires that the use of JI and CDM credits is supplementary to domestic action. Figure 4 compares the maximum fraction of total allocation that can be covered by individual installations using JI and CDM credits under the currently proposed NAPs. As all installations can freely trade allowances, the only binding limit is the resulting
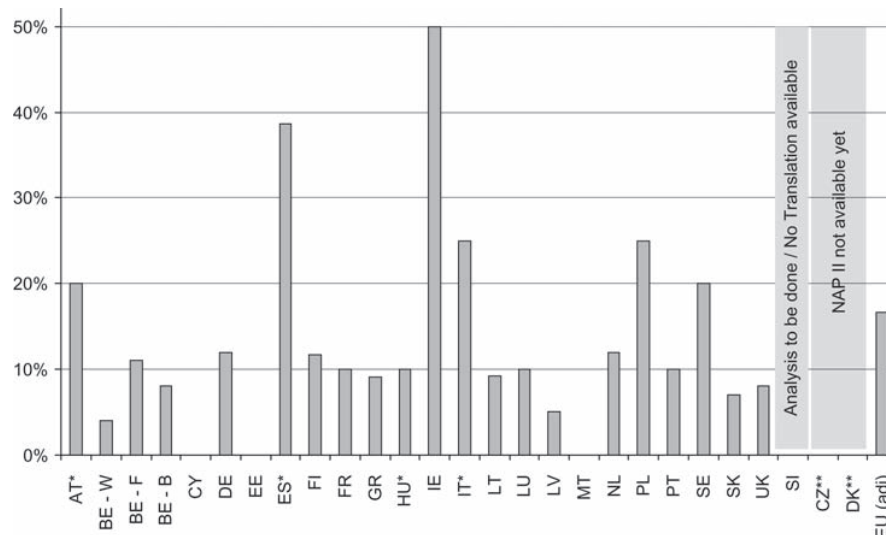
Figure 4. National limits of using JI/CDM credits for EU ETS compliance as a percentage of emissions allocations at installation level ('EU adj' excludes CZ, DK, SI; weighted average for ES). * Draft NAP, ** NAP not available. Limits on the use of Kyoto Credits of 70% of the overall phase-II allocation for the sector 'Electricity production for public service' and 20% for the others, at installation level.

overall import volume from JI and CDM credits. Extrapolating from the currently available NAPs for phase II, a maximum of 16.6% of the emissions of the eligible installations in the EU, or 315.1 MtCO$_2$, may be covered by JI and CDM credits. Whether this upper limit will be reached will depend on prices for EUAs and ERUs or CERs, which in turn depend on demand and supply. The upper limit could for example be reached if Japanese demand, which is estimated to account for about half of total demand for JI and CDM credits (Grubb and Neuhoff, 2006), were to fall.

Article 30(3) requires that the eligible installations across the EU also directly implement measures to reduce emissions by at least the same volume. However, current projections (Neuhoff et al., 2006a, this issue) do not support this hypothesis. Compliance with the Directive would thus require the reduction of the overall budget allocated and/or the volume of JI and CDM credits that can be imported into the EU ETS.

The EU Commission addresses this issue in its decisions on the NAPs. For example, in the Irish NAP the maximum amount of JI and CDM credits that can be used per installation to cover its emissions was declared to be 'inconsistent with Ireland's supplementarity obligations under the Kyoto Protocol and decisions adopted pursuant to the UNFCCC or the Kyoto Protocol, to the extent that it exceeds 21.914%'(EU Commission, 2006).

5. The basis for free allocation

The successful cap-and-trade programmes for SO$_2$ and NO$_X$ in the USA allocate emission allowances to existing facilities typically based on emissions in a fixed historic base period, and then auction the remaining allowances. Only few US states have set aside allowances of their NO$_X$ programmes for new sources. Thus, in most cases the free allowance allocation to existing installations constitutes a lump-sum transfer which does not create distortions for the effectiveness of the scheme.
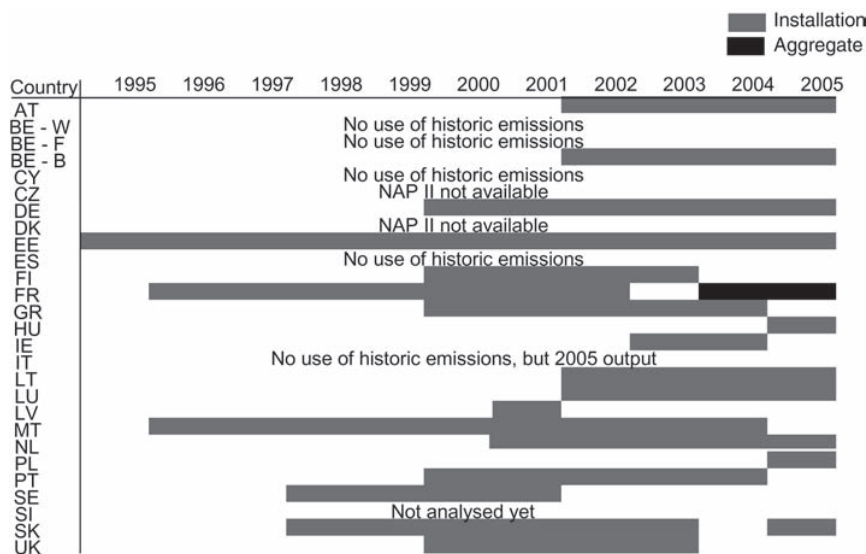
Figure 5. Base period for allowance allocation for the power sector.

In the European context, the limited availability of data, unknown mid- and long-term emissions reduction targets, and distributional considerations (regarding the allocation of allowances valued at around €30 billion) have prevented such a one-off allocation using one historic base period. Figure 5 shows that the 'historic' base period for allowance allocation for the period 2008–2012 has shifted to take account of the most recent data (including the year 2005) in many Member States.

If allowances are not allocated using a fixed historic base line or an auction, then Table 1 illustrates the different categories that can describe alternative allocation approaches (Grubb and Neuhoff, 2006). In Table 1, X indicates the types of distortions that result from the allocation process. The pyramid shape illustrates the increasing number of distortions that occur as we move down the Table from allocation based on auctions or a one-off allocation using a historic base line, to uniform benchmarks, then to fuel-specific benchmarks, and finally to emission-based allocation. Within each of these categories, in principle, allocation based on installed capacity is preferable to allocation based on projected output, which in turn is preferable to allocation based on historic output. The distortions only apply directly to existing facilities. If, however, investors in new installations expect that they will in the future be covered by similar provisions, then the provisions also result in distortions of investment decisions for new installations.

Following these classifications, we have assessed the performance of the allocation plans of different Member States. In Figure 6 we depict the methodology used to determine the allocation to existing facilities in the power (P) and other (O) sectors.

Distortions from allocation today are largely due to expectations about allocation in the future. For private-sector decision-makers, estimates of future allocation are inevitably based on allocation under the status quo. If emission levels in 2000–2005 are made the basis for the allocations in the period 2008–2012, then plant operators may expect that emissions in the period 2005–2010 will be the basis for the allocation post-2012. This creates a typical early action issue: that is to say,

Table 1. Effect of allocation methods to existing installations in the power sector

Allowance allocation method	**Impacts** More expenditure on extending plant life (and potential minimum-run) relative to new build **Distortions** Discourage plant closure	Discourage closure of higher emitting plants	Increase operation of (higher) emitting plants CO$_2$-inefficient fuel choice and plant operation	Less efficiency improvements Reduce incentives for efficiency improvements
Auction				
Uniform benchmark Installed capacity	X			
Output projection	X	X		
Historic output	X	X	(X)[a]	
Technology/fuel-specific benchmark Installed capacity	X	X		
Output projection	X	X		
Historic output	X	X	X	
Emissions-based Emissions projection	X	X		X
Historic emissions	X	X	X	X

[a] To avoid distortions between generation technologies, non-fossil-fuel stations would also have to receive free allowances. This would avoid internalization of CO$_2$ costs in the electricity price, and thus distort choices of input factors and consumption for electricity consumers.

allocation undermines the incentives to invest in emission reductions because such investment may be 'punished' during future allowance allocation. As allocation plans for phase II continue to allocate most allowances to existing facilities based on historic emissions, the early action problem remains to be addressed. Some countries experiment with benchmark approaches – and thus could possibly increase the confidence of private-sector investors that future allowance allocation methodology will improve in terms of economic efficiency and environmental effectiveness.

Figure 7 provides the same analysis for the allocation methodology to new entrants, again separately for the power sector (P) and other sectors (O). It illustrates the variety of approaches selected by different Member States. The big challenge, again, comes from the distortions that follow from private-sector expectations regarding the allocation methodologies in subsequent periods. Thus the assessment of the allocation for the existing installations also carries significance for investment decisions for new facilities.
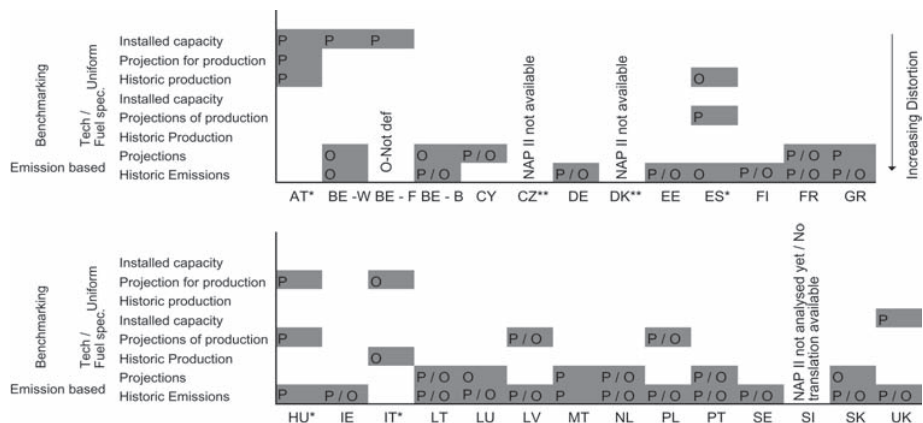
Figure 6. Pyramid of distortions applied to existing installations, power (P) and others (O).

6. Conclusions

We have quantified the volume of free allowances that different national allocation plans envisage to allocate to the power sector. This varies widely across Member States and technologies and can create strong distortions of investment decisions. The level of free allocation seems rather high, given that in most EU countries the electricity market is liberalized and electricity generators are therefore in a position to pass through the opportunity costs of CO_2 allowances. Thus, a significant reduction in free allowance allocation to the power sector seems viable for phase II of the NAPs. As the aggregate demand of individual installations inflates national, and therefore EU, allowance budgets, any such reduction could facilitate a reduction in the EU ETS cap as proposed by the EU Commission in its decision on the first 10 NAPs announced on 29 November 2006. This in turn can ensure sufficient scarcity of CO_2 allowances and a viable emissions market that drives low-carbon investment decisions.
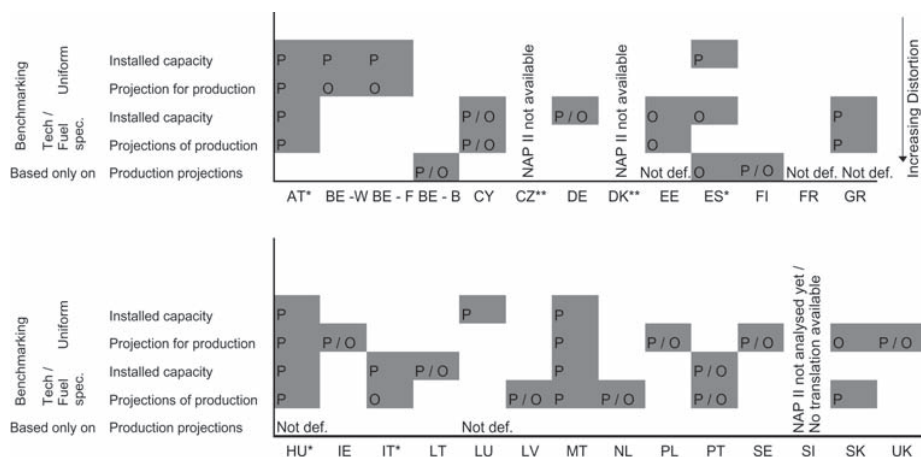

Figure 7. Pyramid of distortions applied to new installations, power (P), others (O).

A reduction in free allowance allocation, mainly to the power sector, could in addition allow for an increased use of the auctioning of CO_2 allowances. Auction volumes vary significantly across Member States. In all Member States, the potential for an increase to 10% envisaged by the Directive remains. The Commission made auctioning the one area of discretionary flexibility within its decision on the first 10 NAPs, 'without prior acceptance by the Commission', and even after the deadline of 31 December 2006. If a tighter cap, stringent limits on CDM and JI inflows, and 10% auctions were implemented, then a price floor in the auction – agreed between EU Member States – could also establish a price floor for EU allowances and thus facilitate low-carbon investments.

A comparison of the volume of CDM and JI credits that individual installations are allowed to use to cover their CO_2 emissions shows large discrepancies between Member States. A more stringent approach seems required in order to satisfy the supplementarity criteria of the Directive and also to avoid too much exposure of the EU ETS market to the uncertainties regarding Japanese and Canadian demands for JI and CDM credits. The Commission decided to disallow the higher limits envisaged in the Irish NAP.

Most allowances are still allocated relative to historic emissions. If the private sector takes this as an indicator for future allowance allocation, then we may face a serious early action problem. Some Member States have started to explore different benchmarking approaches, mainly for the power sector. This has the potential to reduce, but not eliminate, the economic distortions from free allowance allocation. Thus, to ensure the effectiveness of EU ETS in the coming years, it is important to provide credible evidence to the private sector that free allowance allocation will be drastically reduced post-2012. By disallowing the German provision to commit to free allowance allocation post-2012, the Commission has ensured that we retain the flexibility for such changes.

Finally, the EU Directive on Emissions Trading requires that Member States notify their national allocation plans to the Commission to be assessed in relation to State Aid criteria. There are some concerns that the excessive allocation to sectors that both pass on opportunity costs and receive free allowance allocation cannot be aligned with EC Law State Aid criteria (Johnston, 2006). One solution might be to treat the resulting benefits as a transitional payment to compensate for the transition costs of the environmental regulation. This would, however, require a strong commitment to phasing-out free allocation post-2012 – and would thus also address the early action problem.

Acknowledgements

Notes

1   The database, which can be found at http://www.econ.cam.ac.uk/research/tsec/euets/, covers volume of the allocation, verified and projected emissions, allocation methodologies for power and non-power sectors, auctioning, general features, and evaluation of the allocation that a standard power plant would receive in each Member State according to the proposed rules.
2   We assume a 200 MW coal power station and combined cycle gas turbine with efficiencies of 33% (existing coal), 45% (existing CCGT gas), 43% (new coal) and 55% (existing CCGT gas).
3   BE-W Belgium Walloon, CY Cyprus, DE Germany, ES Spain, FI Finland, HU Hungary, IE Ireland, IT Italy, LV Latvia, NL Netherlands, UK, BE-F Belgium Flemish, EE Estonia, LU Luxembourg, SI Slovenia, AT Austria, CZ Czech Republic, DK Denmark, SE Sweden, BE-B Belgium Brussels, FR France, GR Greece, LT Lithuania, MT Malta, PL Poland, PT Portugal, SK Slovakia.

References

Åhman, M., Holmgren, K., 2006. Harmonising New Entrant Allocation in the Nordic Energy Sectors: Current Principles and Options for EU ETS Phase II. IVL Swedish Environmental Research Institute, Stockholm, Sweden.

Åhman, M., Burtraw, D., Kruger, J., Zetterberg, L., 2007. A Ten-Year Rule to guide the allocation of EU emission allowances. Energy Policy 35(3), 1718–1730.

Bartels, M., Müsgens, F., 2006. Do technology specific CO_2 allocations distort investments? In: Securing Energy in Insecure Times. International Association of Energy Economics, Potsdam, Germany.

Betz, R., Rogge, K., Schleich, J. 2006. EU emissions trading: an early analysis of national allocation plans for 2008–2012. Climate Policy 6(4), 361–394.

Böhringer, C., Lange, A., 2005. Mission impossible!? On the harmonization of national allocation plans under the EU Emissions Trading Directive. Downloadable Appendix: CGE model description. Journal of Regulatory Economics 27(1). 81–94.

Burtraw, D., Palmer, K., Bharvirkar, R., Paul, A., 2002. The effect on asset values of the allocation of carbon dioxide emissions allowance. Electricity Journal 15(5), 51–62.

Burtraw, D., Palmer, K., Kahn, D., 2005. Allocation of CO_2 Emissions Allowances in the Regional Greenhouse Gas Cap-and-Trade Program. RFF Discussion Paper 05-25. Resources for the Future, Washington, DC.

del Rio González, P., 2006. Harmonization versus decentralization in the EU ETS: an economic analysis. Climate Policy 6(4), 457–475.

Entec and NERA (2005). EU Emissions Trading Scheme Benchmark Research for Phase II. Final Report. DTI.

EU Commission, 2006. Decision on Irish NAP, 29 November [available at http://ec.europa.eu/environment/climat/2nd_phase_ep.htm].

Fischer, C., 2001. Rebating Environmental Policy Revenues: Output-Based Allocations and Tradable Performance Standards. RFF Discussion Paper 01-22. Resources for the Future, Washington, DC.

Gagelmann, F., 2006. Innovation Effects of Tradable Emission Allowance Schemes: The Treatment of New Entrants and Shutdowns. UFZ (Centre For Environmental Research), Department of Economics, Discussion Paper 4/2006.

Grubb, M., Neuhoff, K., 2006. Allocation and competitiveness in the EU emissions trading scheme: policy overview. Climate Policy 6(1), 7–30.

Harrison, D.J., Radov, D.B., 2002. Evaluation of Alternative Initial Allocation Mechanisms in a European Union Greenhouse Gas Emissions Allowance Trading Schemes. NERA Report to DG Environment, European Commission, NERA.

Hepburn, C., Grubb, M., Neuhoff, K., Matthes, F., Tse, M., 2006. Auctioning of EU ETS phase II allowances: how and why? Climate Policy 6(1), 137–160.

Johnston, A., 2006. Free allocation of allowances under the EU emissions trading scheme: legal issues. Climate Policy 6(1), 115–136.

Matthes, F., Graichen, V., Harthan, R.O., Repenning, J., 2006. Auswirkung verschiedener Allokationsregeln auf Investitionen im Strommarkt, www.oeko.de.

Matthes, F., Graichen, V., Repenning, J., Doble, C., Macadam, J., Taylor, S., Zanoni, D., Chodor, M., 2005. The Environmental Effectiveness and Economic Efficiency of the European Union Emissions Trading Scheme: Structural Aspects of Allocation. WWF, Öko Institute.

Neuhoff, K., Keats-Martinez, K., 2005. Allocation of carbon emission certificates in the power sector: how generators profit from grandfathered rights. Climate Policy 5(1), 61–78.

Neuhoff, K., Grubb, M., Keats, K., 2005. Impact of the Allowance Allocation on Prices and Efficiency. Electricity Policy Research Group, Judge Institute of Management, University of Cambridge, Cambridge, UK.

Neuhoff, K., Ferrario, F., Grubb, M., Gabel, E., Keats, K., 2006a. Emission projections 2008–2012 versus national allocation plans II. Climate Policy 6(4), 395–410.

Neuhoff, K., Keats Martinez, K., Sato, M., 2006b. Allocation, incentives and distortions: the impact of EU ETS emissions allowance allocations to the electricity sector. Climate Policy 6(1), 73–91.

Pál, G., Bartek-Lesi, M., 2006. The Polluter Profits Principle: A Note on the Grandfathering of CO_2 Emissions Rights in Lax-Cap Trading Regimes. REKK (Regional Centre for Energy Policy Research) Working Paper 2006-1.

Palmer, K., Burtraw, D., 2004. Distribution and Efficiency Consequences of Different Approaches to Allocating Tradable Emission Allowances for Sulfur Dioxide, Nitrogen Oxides and Mercury. RFF Discussion Paper. Resources for the Future, Washington, DC.

Spulber, D.F., 1985. Effluent regulation and long-run optimality. Journal of Environmental Economics and Management 12(2), 103–116.

Sterner, T., Muller, A., 2006. Output and Abatement Effects of Allocation Readjustment in Permit Trade. RFF Discussion Paper 06-49. Resources for the Future, Washington, DC.

Paper VII

OPTIONS FOR EMISSION ALLOWANCE ALLOCATION UNDER THE EU EMISSIONS TRADING DIRECTIVE

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Abstract. The article investigates four alternative allocation schemes for emission allowances. The investigated schemes are emission-based allocation, production-based allocation with actor-specific emission factors, production-based allocation with benchmarking and production-based allocation based on data on best available technology (BAT). All the examined schemes apply free allocation based on historical activities. The allocation schemes are evaluated against the criteria for a National Allocation Plan, listed in the Annex III of the EU ETS Directive, and regarding their conformity with the criteria put forward by the Swedish Parliamentary Delegation on Flexible Mechanisms, The FlexMex 2 Commission. No allocation scheme unambiguously meets all criteria. Each has its advantages and disadvantages. Emission-based allocation schemes are most straightforward, transparent and are the easiest to implement. Production-based allocation schemes meet more of the criteria, but are more costly to implement and require more data. Data on BAT will not be available to the extent necessary in order to base an allocation scheme implemented for the trading starting 2005 on BAT. It is unlikely that any given allocation scheme will be perceived as fair by all concerned parties, no matter how sophisticated it is. The overall characteristics of the studied allocation schemes are summarised in the paper. Due to the lack of abatement cost curves, it is not possible to accurately model capital flows between the trading sectors. Data availability will most probably limit the options available to the authorities designing the allocation schemes.

1. Introduction

It is likely that trading of carbon dioxide emissions will start in the EU from January 1, 2005. In July 2003, the European Parliament and the European Commission agreed upon a directive defining the basic rules and regulating the emissions trading scheme(ETS). The final text of the directive, referred to in this article as the ETS Directive or just the Directive, will be formally adopted in September 2003. However, the work with this article started in September 2002, and was based upon the proposal for the ETS Directive available at that time. Since then the wording of the Directive has been changed many times, and even though most of the fundamentals remain the same, some of the assumptions that this article is based upon are no longer completely accurate. However, this does not affect the conclusions of the article, and the authors have also adjusted the text in order to accommodate for the most important changes in the ETS Directive.

2. Objectives

The objectives of the study were to:

(i) Explore four different allocation schemes based on historical activities, with regard to their conformity with the requirements for an allocation scheme, as defined by the EU ETS Directive and the criteria put forward by the Swedish Parliamentary Delegation on Flexible Mechanisms, the FlexMex 2 Commission. The investigated allocation schemes were
- Emission-based allocation
- Production-based allocation with actor-specific emission factors
- Production-based allocation with benchmarking
- Production-based allocation with BAT levels

(ii) Identify what information is necessary in order to implement each of the studied allocation schemes.

(iii) Study the availability and quality of the necessary information.

(iv) Through case studies, describe the consequences of the various allocation schemes in the five sectors included in the trading scheme, with respect to the following aspects:
- How large emissions will the various industries have?
- In which industries is it expected that emissions-reducing actions will take place?
- Approximately how large costs/incomes will the industries have?

3. Delimitation

The study was focussed on possible allocation schemes for the EU emissions trading scheme. The selection of the investigated allocation schemes was done pragmatically – the schemes that seemed most likely to be seriously considered by the authorities were chosen.

According to the EU ETS Directive (2003/.../EC), all Member States have to observe common allocation criteria. In this study, Sweden has been used as the case Member State. Only schemes with free allocation based on historical activities have been analysed in full. The aim of this work has not been to provide new information that can be necessary for an allocation scheme, but rather focuses on examining if such information already exists, or if it can be provided for reasonable costs when an allocation scheme is implemented. How many allowances that will be allocated to the trading sectors in total, and how this should be determined, has only been examined briefly in this report. Allocation between countries, the size of total reductions and the issue of banking have not been analysed. The basic starting point for the study has been that the allocation scheme shall be as homogenous as possible, i.e., the same calculation principles shall be applicable to as many installations as possible.

There is a rich flora of terms and definitions for identical or similar phenomena in the literature that deals with emissions trading. We have tried to be as consistent and clear in our use of terms as possible. However, we recognise that there may be alternative definitions of the terms used in the article, and the reader should be aware of this. When costs for a given allocation scheme are referred to, we include administrative and transaction costs and, when discussing costs for industry, abatement costs or other costs associated with achieving a reduction target. Unless explicitly stated, we do not consider overall costs to society.

4. Characteristics of an Allocation Scheme

In ETS it is the right to emit a given pollutant that is being traded. In the EU trading scheme, the pollutant is carbon dioxide (CO_2). The unit of trade is called an *allowance,* and 1 allowance equals the right to emit 1 tonne of CO_2. *Allocation* is the distribution of allowances to the participants in the trading scheme.

There are two fundamentally different approaches to allocation; allocation that is based on historical activities and allocation based on current or future activities. The former is often referred to as *grandfathering*, often also implying free allocation, and the latter as *updating.* Further, one can distinguish between allocation schemes were the allowances are distributed at no cost, and schemes where the allowances are sold or auctioned to the participants in the trading scheme.

This article is focussed on free allocation based on historical activities. Auctioning and updating is only discussed briefly in section 10. But the reader could bear in mind that many of the choices and problems that have to be dealt with in an allocation scheme are the same, regardless if the allocation is based on historical activities or updating.

In order to do an allocation, it is necessary to decide which installations should be included in the trading scheme, how many allowances should be distributed in total, and how the distribution should be calculated. This article is focussed on the last of these issues. The EU ETS Directive indicates which sectors will be included, and it is these five sectors that will be examined in this study. Not all CO_2 emissions will be included in the trading. Exactly which installations will be included, and to which sector they will be counted, has not yet been decided. While this does not affect the principal reasoning in this report, it can have an effect on the actual result when implementing the different allocation schemes (Figure 1).

4.1. WHAT IS A 'FAIR' ALLOCATION SCHEME?

To freely allocate allowances is to distribute financial capital to installations, and as long as the available fortune is not infinitely large it is probably safe to say that some
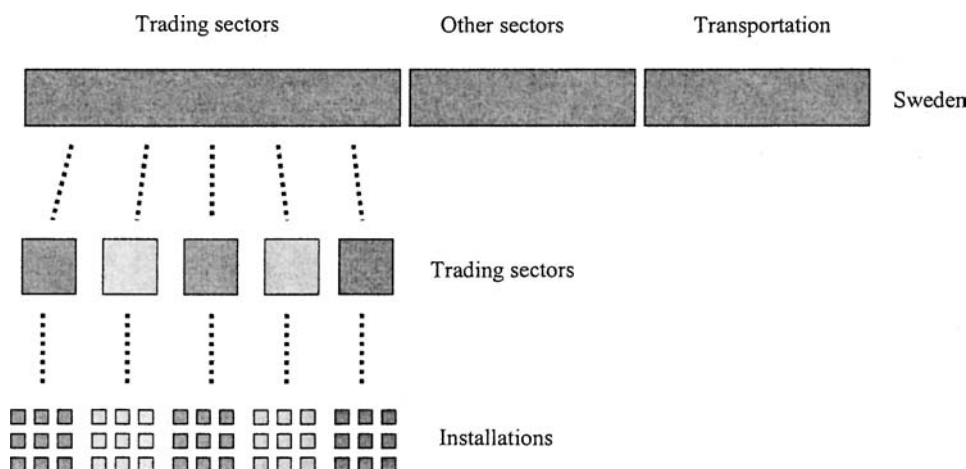
Figure 1. Principle outline for allocation of emissions allowances. Each figure represents CO_2^- emissions from an installation or a sector, or allocated allowances to an installation or a sector.

stakeholders will receive less than they feel they have right to. No allocation scheme will be considered to be "fair" by all stakeholders, regardless of how sophisticated it is. The ideal is of course to construct a scheme that meets the objectives of the authorities, is effective and that is accepted and considered to be reasonable by as many stakeholders as possible. As will be shown in this article, the quality of an allocation scheme is in part limited by costs and availability of data. But to find the perfect scheme, one that is accepted by all stakeholders as it is and that handles all problems that will come up, is in all likelihood impossible regardless of what resources or data are available.

According to the ETS Directive, each Member State has to draw up an ex-ante national allocation plan. The plan has to be made available for the public to comment on, and can be also rejected by the European Commission if it does not meet the criteria stipulated in the ETS Directive.

These criteria, listed in the Annex III of the ETS Directive, are:
(a) The total quantity of allowances to be allocated for the relevant period shall be consistent with the Member State's obligation to limit its emissions pursuant to Decision 2002/358/EC and the Kyoto Protocol, taking into account, on the one hand, the proportion of overall emissions that these represent in comparison with emissions from sources not covered by this Directive and, on the other hand, should be consistent with the national climate change programme. The total quantity of allowances to be allocated shall not be more than what is likely to be needed for the strict application of the criteria in this Annex. Prior to 2008, the quantity shall be consistent with a path towards achieving or over-achieving each Member State's target under Decision 2002/358/EC.

(b) The total quantity of allowances to be allocated shall be consistent with assessments of actual and projected progress towards fulfilling the Member States' contributions to the Community's commitments made pursuant to Decision 93/389/EEC.

(c) Quantities of allowances to be allocated shall be consistent with the potential, including the technological potential, of activities covered by this scheme to reduce emissions. Member States may base their distribution of allowances on average emissions of greenhouse gases by product in each activity and achievable progress in each activity.

(d) The plan shall be consistent with other EC legislative and policy instruments. Account should be taken of unavoidable increases in emissions resulting from new legislative requirements.

(e) The plan shall not discriminate between companies or sectors in such a way as to unduly favour certain undertakings or activities in accordance with the requirements of the Treaty, in particular Articles 87 and 88 thereof.

(f) The plan shall contain information on the manner in which new entrants will be able to begin participating in the Community scheme in the Member State concerned.

(g) The plan may accommodate early action and shall contain information on the manner in which early action is taken into account. Benchmarks derived from reference documents concerning the best available technologies may be employed by Member States in developing their National Allocation Plans, and these benchmarks can incorporate an element of accommodating early action.

(h) The plan shall contain information on the manner in which clean technology, including energy efficient technologies, are taken into account.

(i) The plan shall include provisions for comments to be expressed by the public, and contain information on the arrangements by which due account will be taken of these comments before a decision on the allocation of allowances is taken.

(j) The plan shall contain a list of the installations covered by this Directive with the quantities of allowances intended to be allocated to each**.

(k) The plan may contain information on the manner in which the existence of competition from countries/entities outside the EU will be taken into account.

In addition to this, the Swedish parliamentary delegation working on the subject of flexible mechanisms, including emissions trading, in Sweden, the FlexMex 2 Commission, has listed certain criteria for the Swedish national allocation scheme. These criteria are that the allocation scheme should be:

(i) in accordance with the political greenhouse gas goals

(ii) acceptable for the stakeholders

(iii) transparent

(iv) non-bureaucratic

  (v) predictable
 (vi) in accordance with the EU rules for State aid
(vii) in accordance with the EU rules for competition
(viii) possible to implement by the end of 2003

Both EU's Annex III criteria and the requirements stipulated by the Flex Mex 2 Commission can be seen as attempts to define a fair allocation scheme of high quality. In this study, we have evaluated the different allocation schemes against these two lists. However, Annex III criteria 1, 2, 3, 9 and 10 are only briefly discussed, as are the FlexMex 2 criteria 1, 2, 6 and 7. Either do all studied allocation schemes have the same possibilty to meet those criteria, or do they fall beyond the scope of this article.

The reader should also bear in mind that the interpretation of neither EU's Annex III criteria nor the Flex Mex 2 delegation's list are clear. For example, can several of the Annex III criteria be seen as being contradictory. It would therefore be unwise to say for sure if any given allocation scheme meets the criteria of the EU directive or not.

4.2. DETERMINING THE TOTAL NUMBER OF ALLOCATED ALLOWANCES

The consequences of a given allocation scheme will be affected by the total number of allocated allowance in Sweden and in other countries.

The EU directive provides no explicit rules for how many allowances should be allocated. It is, however, stated that consideration should be given to EU's distribution of commitments under the Kyoto Protocol, which applies from 2008 to 2012. Each Member State will have to make it plausible that it will be able to fulfill its national obligation to limit emissions pursuant to the EU Decision, under the proposed naitonal allocation plan. This means that if the trading sectors are given many allowances, Sweden will have to show that larger reductions will be possible in non-trading sectors.

There are several ways to determine the total amount of allowances that will be allocated to the trading sectors. Above all else, it will depend on what reduction goals Sweden will have as a country and how large a proportion of the Swedish emissions the trading sectors will be allowed to have. Principally, one can differentiate between the bottom-up approach and the top-down approach. Bottom-up implies that one starts the allocation of allowances according to a given allocation scheme at the installation level, sum up the allowances for the respective industries and then sum up the industries to get the total for all trading sectors. The calculated amount of allowances can then either be allocated out to the trading installations, or adjusted based on other factors.

A top-down approach implies that the starting point is determining how many tonnes of emissions the trading sectors will be allowed to have under the trading scheme. Once that is determined, allowances can be allocated according to a given allocation scheme.

Even if Sweden's national reduction target under EU's joint commitment is +4%, and Riksdagen's (the Swedish Parliament) national reduction goal is –4%, calculated from 1990 levels, it is not clear if the same reduction targets can be used for the trading sectors as for Sweden as a whole. Preliminary data indicate that Sweden's total greenhouse gas emissions have been stable since 1990, but the proportions between different sectors can have changed. This means that emissions that came from the non-trading sectors in the past may now be produced from the trading sectors, and vice versa. One example is heating. During the 1990's, district heating replaced some furnace heating in the residential sector. This has led to increased emissions from district heating, while emissions from households may have decreased by the same amount, or more. These issues are, however, mostly political considerations and will not be dealt with further in this study.

5. Allocation Based on Historical Activities

5.1. DEFINITION

Allocation based on historical activities, or upon current activities that are beyond the control of an individual actor.

5.2. IMPORTANT ISSUES

Two important decisions must be taken before an allocation scheme based on historical activities can be introduced:
  (a) Allocation base, i.e. upon which activities the allocations will be based
  (b) Which base year(s) will be used for allocation

5.2.1. *Allocation base*
There are three main alternatives:
   (i) Emissions
  (ii) Output
 (iii) Input

Emissions.   The principle behind emission-based allocation is that the actors that have previously generated emissions are allocated allowances based upon these. Emission-based allocation is in many ways the most simple form of allocation. Emissions are also easy to measure and monitor, at least as long as only carbon dioxide is included in the trading scheme.

Output.   The principle behind output-based allocation is to multiply the production at an installation, or group of installations, by some form of emission factor, and to use this as the base for allocation. The emission factor can be determined in a number of ways. They can be actor-specific (see section 8.1), specific for a group of installations (see section 8.2), or technology-specific (see section 8.3).

Output-based allocation, regardless of which form it comes in, will be problematic if the products covered by the scheme are not similar. The more heterogeneous the product group, the greater the problems will be. It is obvious that it is difficult to compare a tonne of paper with a tonne of steel, but there are even large differences between different types of paper quality. In addition, the differences in a given installation's products can change over time, as the installation's product portfolio changes, or as the quality of the product changes.

An example of production-based allocation is the California's RECLAIM programme for trade of SO_2 emissions.

Input. When allocation is based upon the input of some resource, a lot of the comparison problems connected with output-based allocation are avoided. The most common resource named in this context is used energy, often measured as supplied fuel. It is often a relevant resource and it is also relatively easy to monitor.

There is a risk that input-based allocation can be disadvantageous for process efficiency and high efficiency in energy conversion processes. Take for example a case with two electricity-producing installations that both use natural gas as their energy resource. One installation uses 100 MJ of gas to produce 40 MJ of electricity, the other uses 100 MJ of gas to produce 45 MJ of electricity. With a supply-based allocation scheme, both installations would be allocated the same amount of allowances despite the fact that the second installation has a higher efficiency and produces more electricity. We feel that it is more effective to have an allocation scheme based upon the amount of produced goods and have therefore not analysed input-based allocation schemes further.

5.2.2. *Selection of Base Year*

We study allocation schemes based upon the activities conducted at an installation during a given (historical) time period. The time period can be a year, several consecutive years, or a number of years selected in some way. This year is referred to as the *base year*. There are two important factors when choosing the base year:
- is/are the year(s) representative for the actor's activities?
- how are installations that have/have not taken action before the base year affected?
  (See section 6 "Emission-based allocation" for further discussions around the issue of choosing the base year.)

6. Emission-Based Allocation

6.1. DEFINITION

Emission-based allocation to an installation within a given sector means that the allocation is calculated as the installation's portion of the total emissions from all
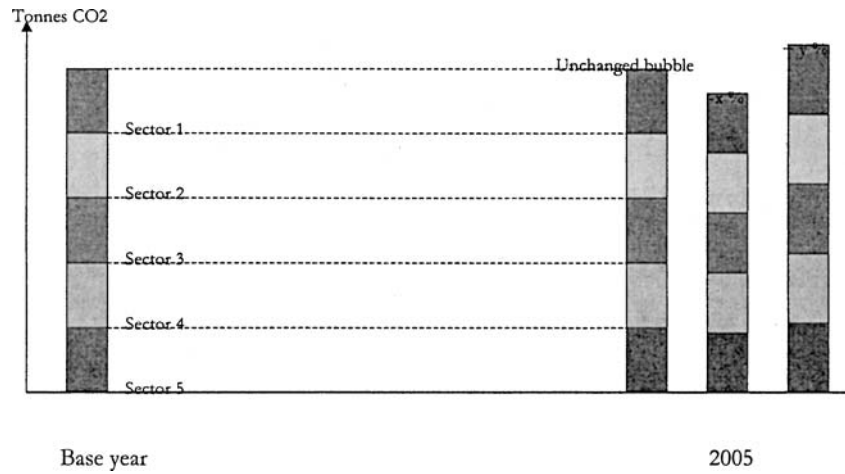
Figure 2. Principle for emission-based allocation with base year 1990. Each installation or sector keeps the same portion of the total bubble, calculated from the base year, regardless of emission levels when trading starts. Through adjustments against a scale factor, the total number of allowances can be greater than, or less than, the historical emissions.

installations in the sector, multiplied by the total allocations to all installations within the sector.

Mathematically, this can be expressed as:

$$A_{\text{installation}_T} = \frac{E_{\text{installation}_{\text{baseyear}}}}{E_{\text{sector}_{\text{baseyear}}}} \times A_{\text{sector}_T} \tag{1}$$

A = allocated allowances
A_{sector} = total allocations for all installations in a given sector
T = start year for trade
E = emissions

Allocation on sector level can be conducted in analogue with allocation to installations (Figure 2).

6.1.1. *Important Issues*

The selection of the base year is the most important issue with emission-based allocation. One or several base years can be chosen. The base year, or years, is used to determine the historical emissions for an installation or sector. Each actor's portion of the total number of allocated allowances is also determined based upon proportions during the base year(s). On the other hand, the size of the emissions during the base year(s) is not a factor when calculating the total number of allowances to be allocated. Each actor is allocated the same portion of the total allocated allowances as it had under the base year(s). The possible choice of a scale factor depends upon whether or not the total number of allowances to be allocated
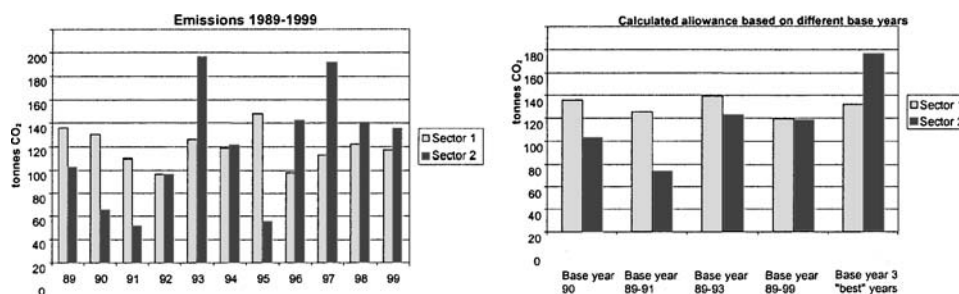
Figure 3. Effects of the choice of base year on allowance allocation. Despite the fact that sector 1 and sector 2 have similar total emissions during the time period, their calculated allowances differ greatly. Of particular note is that sector 2, which has greater variations in emissions than sector 1, benefits if the sectors get to choose the base year themselves.

is exceeded during allocation. For further information regarding the determination of the total number of allowances, see section 4.2 "Determining the total number of allocated allowances".

It is possible to see the general effects of the choice of the base year, but even differences between consecutive years can have significant effects for certain sectors or companies (see Figure 3). One problem can be the difficulties in logically and clearly motivating the choice of the base year, to the stakeholders, regardless of whether an early or late base year is chosen (Figure 3).

Some general effects of different choices can be identified:

Early base year:
- actors that have done early actions benefit, compared with if a late base year is chosen
- larger problems with new entrants and production changes that have occurred since the base year, compared with if a late base year is chosen
- problems in obtaining reliable data are greater the earlier the base year is.

Late base year:
- less problems with new entrants and changes in production that have occurred after the base year
- reduces risks for stranded costs
- less beneficial for actors that have done early actions

Sectors/companies get to choose their own base year, upon which allocation is determined in proportion to the distributions given by these choices, adjusted so that the bubble's size is not exceeded:
- less problems with individual sectors or companies being treated unfairly because of, for example, the choice of a base year that is non-representative for the actor
- benefits actors whose activities vary greatly between years

If one single year is used as the base year, there is a risk that the representativeness is compromised, such as if there has been a break in activities due to extreme events

or variations in external conditions. This risk can be decreased by choosing several base years, such as three, four or five.

6.2. RESULTS OF EMISSION-BASED ALLOCATION

Five sectors, which together represent approximately 30% of annual carbon dioxide emissions in Sweden, will be included in the trading scheme starting in 2005. These sectors are pulp and paper, refineries, mineral industries, iron & steel industries and coke plants, electricity and heating. In addition to this, industrial boilers with an installed capacity exceeding 20 MW will be included in the trading scheme.

TABLE I

Summary of carbon dioxide emissions from the trading sectors, 1990–2000, which have been used for the allocation

Year	Pulp & paper, printing	Refineries	Mineral	Iron & Steel, Coke plants	Electricity, heating	Industrial boilers	Total
SNI:	21	23	26	27–28, 34	40		
1990	2.0[a]	1.6[b]	2.3[b,d]	4.1[a]	6.9[a]		17.8[f]
1991	2.2[a]	1.9[b]	2.8[b,d]	4.3[a]	7.9[a]		19.8[f]
1992	1.9[a]	0.9[b]	2.5[b,d]	4.5[a]	8.4[a]		19.0[f]
1993	2.5[a]	0.6[b]	2.5[b,d]	4.6[a]	8.2[a]		19.3[f]
1994	3.1[a]	2.2[b]	2.6[b,d]	4.8[a]	8.9[a]		22.5[f]
1995	3.1[a]	1.9[c]	3.0[b,d]	5.3[a]	7.9[a]		22.0[f]
1996	2.8[a]	2.2[c]	3.0[b,d]	5.0[a]	12.0[a]		25.8[f]
1997	3.4[a]	2.3[c]		4.9[a]	7.7[a]		21.9[f]
1998	3.0[a]	2.1[c]		4.9[a]	8.7[a]		22.3[f]
1999	2.6[a]	2.3[c]		4.9[a]	6.8[a]		20.1[f]
2000	2.0[a]	2.3[c]	2.7[b,d]	5.4[a]	5.3[a]	0.9[e]	18.6[f]
(ÅF)	1.9[e]	2.3[e]	3.0[e]	5.0[e]	6.0[e]	0.9[e]	19.0[e]

NB. Emissions data has since been updated and shown to be vary considerably on sector level, throughout the time period (SOU 2003)For reference, data from the report by the ÅF group (Remes et al. 2002) are given at the bottom of the table. Data are given in Mton CO_2.

[a]The Swedish Environmental Protection Agency's Report to UNFCCC, 2002. Data for pulp and paper includes printing, whose share of total emissions from the sector was less than 4% in 1998.

[b]Rudander, Palm, 2002. Results from Statistics Sweden (SCB) information searches at the installation level. The basic data includes stationary sources from pulp and paper, refineries, the mineral industry (15 out of 20 installations, all major ones included), steel industry, coke plants and industrial boilers.

[c]Diczfalusy, 2002. Base data on refineries from the Confederation of Swedish Enterprise.

[d]Swedish Environmental Protection Agency, 2002. Process emissions for the Mineral industry and Steel industry from the Swedish Environmental Protection Agency's report to UNFCCC.

[e]Remes et al. (2002). ÅF Group report to the FlexMex 2 commission.

[f]When calculating total emissions, data for 2000 was used for the years for which nodata was available. (Mineral industries 1997–1999 and Industrial boilers 1990–1999).

TABLE II

Results of emission-based allocation to sectors using different base years

Base year	Pulp and paper, printing	Refineries	Mineral	Steel, Coke plants	Electricity and heating	Industrial boilers
SNI	21	23	26	27–28, 34	40	
1990–2000	2.3 (12%)	1.6 (9%)	2.4 (13%)	4.3 (23%)	7.2 (39%)	0.8 (4%)
1990	2.1 (11%)	1.6 (9%)	2.4 (13%)	4.3 (23%)	7.2 (39%)	0.9 (5%)
1990–1993	2.1 (11%)	1.2 (6%)	2.5 (13%)	4.3 (23%)	7.7 (41%)	0.9 (5%)
2000	2.0 (10%)	2.3 (12%)	2.7 (15%)	5.4 (29%)	5.3 (29%)	0.9 (5%)
1997–2000	2.5 (13%)	2.0 (11%)	2.4 (13%)	4.5 (24%)	6.4 (34%)	0.8 (4%)
Maximum 3 yr	2.4 (13%)	1.8 (9%)	2.2 (12%)	4.0 (21%)	7.5 (40%)	0.7 (4%)

The total number of allowances (the bubble) is assumed to be the same as emissions in the year 2000, i.e., 18.6 million tonnes. The results are given in million tonnes, and in percentage total number of allowances. Data for 2000 was used for the years for which nodata was available. (Mineral industries 1997–1999 and Industrial boilers 1990–1999). Please note that there are uncertainties in the data. It is likely that this data will be revised in the future, which will affect the allocation.

TABLE III

Result for the Swedish iron and steel sector after benchmarking

	Production (tonnes)	Allocation CO_2 (tonnes)
Company	**5 760 000**	**4 610 000**
Avesta Polarit Avesta	433 000	346 550
Avesta Polarit Degerfors	172 000	137 660
Erasteel	23 000	18 408
Fundia	600 000	480 208
Höganäs Höganäs	100 000	80 035
Höganäs Halmstad	100 000	80 035
Ovako Steel	500 000	400 174
Sandvik Steel	210 000	168 073
Scana Steel	70 000	56 024
SSAB Luleå	1 980 000	1 584 688
SSAB Oxelösund	1 500 000	1 200 521
Uddeholm tooling	72 000	57 625

Table II shows carbon dioxide emissions for the years 1990–2000 in the trading sectors. Table III shows the outcome within the various sectors with emission-based allocations, given different base year alternatives.

The table shows that installations that have reduced their emissions during the 1990's benefit from an early base year (electricity and heating), while sectors that have increased their emissions benefit from a late base year (pulp and

paper, refineries, iron & steel). One can also notice that sectors that show large variations in emissions benefit from a scheme where the three maximum years in terms of emissions for each sector are chosen (electricity and heating).

This type of allocation could result in certain installations receiving more allowances than they need. This is particularly true if one adjusts for production changes at the same time as a sector or installation, to a larger extent than other industries or installations, has improved its efficiency and increased its production. This can conflict with one of the EU criteria for an allocation scheme that states that no actor shall be unduly favoured (the Annex III criteria are listed in section 3).

6.3. SUMMARY ASSESSMENT OF EMISSION-BASED ALLOCATION

(a) Easy to understand.
(b) The smallest information requirements of the allocation schemes studied.
(c) The mechanism that makes the situation for companies the most similar to a scenario without trade, which minimises the risk for companies having stranded costs.
(d) Does not benefit early action, if a late base year is used.
(e) Does not handle changes in production or installations, if an early base year is used.
(f) Can mean that installations receive more allowances than they need.

7. Production-Based Allocation with Actor-Specific Emission Factors

7.1. DEFINITION

A pure emission-based allocation does not take into consideration possible changes in production or emissions, which have taken place after the base year used for allocation. One can, however, make adjustments for such changes. The allocation is then calculated based on an installation's efficiency (emissions per produced unit) during the base year, but proportional to the production that the installation has in the year used for calculation (i.e., a later year than the base year). The total number of allocated allowances can also be adjusted with a scale factor.

Production-based allocation with actor-specific emission factors means that the allocation is calculated from specific emissions for an actor during the base year, multiplied by the production during the year used for the calculation of the allocation, possibly adjusted with a scale factor.

Mathematically, this can be expressed as:

$$A_{\text{installation}_\text{T}} = P_{\text{installation}_\text{prod.year}} \times \frac{E_{\text{installation}_\text{baseyear}}}{P_{\text{installation}_\text{baseyear}}} \times f_{\text{installation}}$$

A = allocation
P = production

T = start year for trade

E = emissions

baseyear = the year chosen to define the sector's specific emissions

prod.year = the year chosen to define the production levels of an installation upon which allocation is based f = scale factor to adjust an installation's allocation so that the total amount of allocated allowances for all installations in a sector concur with the sector's allocation. The scale factor is set at the time of allocation, and is thus known to the installation before trading starts.

7.2. Summary assessment for production-based allocation with actor-specific emission factors

(a) Takes into consideration production changes that have occurred after the base year that is used for calculating the general allocation.

(b) Requires more information than emission-based allocation schemes do.

(c) Difficult to unquestionably motivate the choice of base year for emission factors and calculating year for production.

(d) Difficult to handle differences in products between actors, both between sectors and within a given sector.

(e) Difficult to handle changes in individual installations' products.

8. Production-Based Allocation with Sector-Specific Emission Factors - Benchmarking

8.1. Definition

Benchmarking implies that one compares an installation, in this case an installation's CO_2 emissions with a reference value, a benchmark. In order to be able to compare installations with different sizes, emissions must be given in relation to something, for example production. For a paper mill, this could be tonnes CO_2/tonnes produced pulp. This measure is sometimes called *specific emission* and is an efficiency measure for the installation. The installation's specific emission can then be compared with other equivalent measures within the sector. In order for the comparison to be relevant, the installation's activities must be as similar to the sector's as possible.

In this report, we have defined allocation based on benchmarking as follows:

"Benchmarking means that the allocation to an installation is calculated as the installation's production multiplied by the specific emission for the sector, in tonnes of CO_2 per produced unit, possibly adjusted with a scale factor."

Mathematically, this can be expressed as:

$$A_{\text{installation}} = P_{\text{installation,prod.year}} \times e_{\text{sector,baseyear}} \times f$$

where

$$e_{\text{sector,baseyear}} = \frac{E_{\text{sector,baseyear}}}{P_{\text{sector,baseyear}}}$$

A = allocated allowances
P = production
E = carbon dioxide emissions
e = specific emission, i.e., emissions per production unit
baseyear = the year chosen to define the sector's specific emissions
prod.year = the year chosen to define the production levels of an installation upon which allocation is based
f = scale factor to adjust an installation's allocation so that the total amount of allocated allowances for all installations in a sector concur with the sector's allocation. The scale factor is set at the time of allocation, and is thus known to the installation before trading starts.

For most applications, the base year is the same as the production year. In this case, f can be calculated as:

$$f = \frac{A_{\text{sector}}}{E_{\text{sector,baseyear}}}$$

and the expression for an installation's allowance can be simplified to:

$$A_{\text{installation}} = \frac{P_{\text{installation,baseyear}}}{P_{\text{sector,baseyear}}} \times A_{\text{sector}}$$

"Allocation to an installation is calculated as the installation's portion of the total production within the sector multiplied by the total allocation for all installations in the sector."

It is therefore not necessary to know the sector's actual emissions if the sector's total allocation, A_{sector}, has been defined.

Figure 5 shows schematically how benchmarking works. Each bar represents an installation. The height of the bar shows the specific emission in tonnes CO_2/tonnes product. The width of the bar shows the size of production. The bar's area therefore represents total emissions. Emissions before allocation are shown on the left of the figure. In this example, we have assumed that the total number of allowances are the same as the historical emissions. The specific emission factor is the sector's total emissions divided by the sector's total production, which in this case is 2 tonnes of CO_2/tonnes of product. After allocation, all installations are given the same specific emission. Those installations that had below-average emissions receive more allowances than their emissions, while those installations that had above-average emissions receive fewer allowances than their emissions.
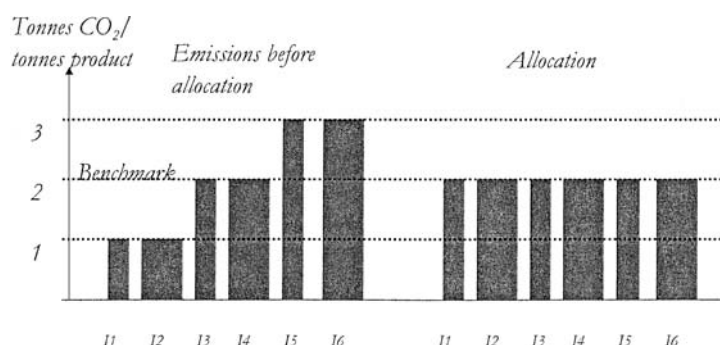
Figure 5. Schematic description of benchmarking.

8.1.1. *Important Issues*

Important issues that have to be dealt with by the policy maker when designing an allocation schem on the basis of benchmarking are:

  (i) What measure should be used for production?
 (ii) How should installations be grouped when developing benchmarks?
(iii) Can benchmarking based upon an economic measure be used for allocation?
 (iv) Can international benchmarking be used?
  (v) How is the problem of divided production chains dealt with?

8.1.2. *Measure of Production*

The principle of benchmarking is that the carbon dioxide emissions should be set in relation to some comparable measure that is shared for a group of actors. All actors should be allocated the same amount of allowances per comparable unit. This will benefit those actors that are more carbon dioxide efficient than others.

The idea of a scheme that benefits the most efficient installations is attractive for many, but the problem is how to find and define the comparable measures that will be used. The choice of these will have great importance for who the winners and losers in the trading scheme will be.

The comparison can be based upon three principally different parameters:
 – output/physical production
 – input
 – economic measures

Output. Physical production means amount of produced product, such as kWh energy or tonnes pig iron. A number of "unit products" are identified, whereby each unit product is used for comparison within a group of installations, such as a sector or sub-sector. In the energy sector, 1 kWh electricity or heat is a unit product that does not have any large complications, but in most sectors, it is much more difficult

to find such unit products. The differences between products' characteristics and the fact that installations may be responsible for certain parts of a process chain creates complications for comparison. A finer division of a sector, into sub-sectors, should decrease such problems, but they will never be completely solved. In addition, there is a clear risk that the number of installations in each sub-sector in Sweden would be so small that the average value would have no real meaning.

Input. The problem with different product characteristics and different stages in a process makes it interesting to study alternative comparison measures. One such alternative is input of resources. A conceivable alternative is allocation based upon amount of energy used. Allowances are thus in the form of CO_2/used energy, and the scheme would benefit actors that generate less carbon dioxide per amount of energy used than their competitors.

There is a risk that input-based allocation can be disadvantageous for process efficiency and high efficiency in energy conversion processes. Take for example a case with two electricity-producing installations that both use natural gas as their energy resource. One installation uses 100 MJ of gas to produce 40 MJ of electricity, the other uses 100 MJ of gas to produce 45 MJ of electricity. With a supply-based allocation scheme, both installations would be allocated the same amount of allowances despite the fact that the second installation has a higher efficiency and produces more electricity. We feel that it is more effective to have an allocation scheme based upon the amount of produced goods and have therefore not analysed input-based allocation schemes further.

Economic measure. An economic measure can be, for example, gross value added, production value, and working hours. Such measures would make it much easier to have benchmarks in industries, such as pulp and paper, refineries, or iron and steel, where there are large differences in the quality of products. Instead of having benchmarks in the form of CO_2/tonnes product, there could be benchmarks in the form of CO_2 per gross value added, CO_2 per production value, or CO_2 per number of work hours. A higher quality would result in a higher price or higher gross value added, which would be shown in the specific emission. Problems can, however, arise in determining gross value added and production value for individual installations.

8.1.3. *Sector Benchmarking Based on Economic Measures*
So far we have only discussed benchmarking between installations within a sector. But is it possible to use benchmarking for allocation at the national level or sector level? The definition becomes analogous to the previous one for installations:

Allocation for a sector is calculated as the sector's production multiplied by the specific emission for all trading sectors, expressed in tonnes of CO_2 per produced unit, possibly adjusted with a scale factor.

Or as a mathematical formula:

$$A_{\text{sector}} = P_{\text{sector}_{\text{prod.year}}} \times e_{\text{industry}_{\text{baseyear}}} \times f$$

$$e_{\text{industry}_{\text{baseyear}}} = \frac{E_{\text{industry}_{\text{baseyear}}}}{P_{\text{industry}_{\text{baseyear}}}}$$

It is, however, very difficult to define a measure for production that is relevant across all sectors. One possible solution is to use economic measures such as gross value added, production value, or total number of work hours. Previous studies have shown that is it possible to define specific emissions at the sector level as tonnes of CO_2/gross value added, tonnes of CO_2/production value, or tonnes of CO_2/number of work hours (Åhman et al. 2001). For the purpose at hand, however, it might be necessary to develop economic measures at the installation level, which is possible for work hours, but difficult for gross value added and production value.

8.1.4. *Benchmarking within Sweden–Grouping of Installations with the Same Benchmark*

In Sweden, there are 12 installations within the iron and steel sector that are expected to be included in emissions trading. Three installations reduce iron ore to metallic iron, a process that leads to large scale carbon dioxide emissions, while the other installations produce alloys from steel slabs. The Table III shows the result of allowance allocation within the iron and steel industry based upon a benchmark of 800 kg carbon dioxide/tonnes product produced. This benchmark was calculated from the sum of all 12 installations' CO_2 emissions divided by their total production in tonnes.

In this example, SSAB will receive approximately 2,8 M tonnes CO_2, despite the fact that it currently has much higher carbon dioxide emissions. This is because SSAB's reduction processes are more energy-consuming than the other installations in the group. This example shows that if benchmarking is to be used, it is important for the group for which the benchmark is calculated to have comparable processes and products.

There are also examples from other sectors. How can a refinery that produces low-sulphur fuel be compared with another? How do you compare a paper mill that produces chlorine-free paper with another? What should be done when several products are produced at the same installation? Our example shows the importance of creating sub-groups within sectors, sub-sectors, with similar activities.

In Sweden, there are examples of products that are produced at only one or two installations, such as at SSAB. In these cases, benchmarking within Sweden is not meaningful. In order to gain a clear picture of where benchmarking is possible, we asked several trade organisations for which products in their sector benchmarking would be meaningful. The list of installations expected to be included in the EU directive is from the ÅF-Group (Remes et al. 2002). The results are shown below:

Iron and Steel: There are 12 installations that are expected to be included in the trading scheme. The Swedish steel trade organisation argues that installations should be divided into smaller groups of 2–3 installations. The *Swedish Steel Producers' Association* therefore argues that benchmarking would not be meaningful in this sector (Lindblad, 2002).

Coke plant: There are two coke plants in Sweden. We argue that benchmarking in Sweden is not applicable.

Roasting, Sintering: There are three installations in Sweden, which we argue is too few for benchmarking to be applicable within Sweden.

Electricity and Heating: Just under 200 installations in this sector are expected to be included in the trading scheme (Remes et al. 2002). A district heating network is regarded as one installation by the EU Directive. The sector, in general, produces two products – electricity and heating. Heating can be of high or low quality, depending upon the temperature, but it should be able to group producers into a limited number or categories. This is confirmed by the sector (Granström 2002). Our conclusion is that a benchmark can be created for the sub-groups' electricity production and heating production. In addition to these there are combined heat and power installations, that is, installations that produce both electricity and heat.

Pulp and paper: There are a total of 61 installations that are expected to be included in the EU Directive. Of these, 26 are integrated plants, 13 are pulp mills, 15 are paper mills and 7 are recycled paper mills. The trade organisation *Swedish Forest Industries Federation* argues that benchmarking should not be used in this sector because the products and mix of products vary too widely between installations (Axelsson 2002).

Refineries: Five installations are expected to be included in the trading scheme, all of which differ in produced product. According to Scanraff and Shell, it is therefore impossible to set any sector-wide benchmark within Sweden (Brinck 2002; Wallin 2002).

Cement and other mineral industry: The EU Directive includes 20 installations within the mineral industry, which include the following product groups: cement, lime, ceramic, porcelain, gypsum boards, brick, rock wool, fibreglass and glass. Benchmarks would have to be defined for each product group, but we argue that the number of installations within each product group is too small in order for benchmarking within Sweden to be applicable.

In summary, it could be possible to use benchmarking within the electricity and heating sector in Sweden.

8.1.5. *International Benchmarking*

One of the problems with benchmarking within Sweden is that certain sectors have too few installations to set a relevant benchmark. One way to get by this problem is to define international benchmarks that can be used as the basis for determining allocation. A benefit with this method is that it makes it possible to take into consideration a sector's or installation's efficiency in comparison with the rest of the world. Greater consideration is then also given to international competition and the sector's ability to reduce emissions. In addition, installations with more carbon dioxide-efficient technology are benefited. However, it is not always evident that it is relevant to compare Swedish installations with installations in other countries. For instance, the Swedish conditions can be different with respect to raw materials and energy, or an installation may not be subject to international competition.

Allocation can be defined in analogue with benchmarking as:

$$A_{\text{installation}} = P_{\text{installation,calc.year}} \times e_{\text{sec } tor,\text{baseyear}} \times f$$

where e = the average specific emission (in tonnes CO_2 per tonne product) for a group of international installations with comparable processes/products.

Figure 6 shows schematically how allocation based on bencemarking is done. For simplicity's sake, it is assumed that the installations in the figure are the same size in terms of production. The process is as follows:

Step 1. Every bar in the figure represents an installation (I1, I2, ... I5). The set of bars on the left shows emissions for each installation. An international benchmark is set for each installation (BM1, BM2, ... BM5), that is the specific emission for each produced unit based on comparable installations around the world.

Step 2. An initial allocation is done, which is the installation's production multiplied by the international benchmark (middle set of bars).
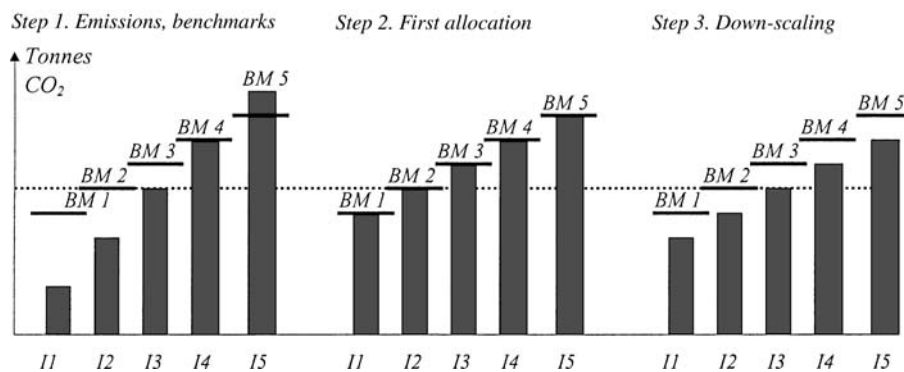

Figure 6. Schematic description of production-based allocation with emission factors based on international benchmarks.

Step 3. The initial allocation can result in the total number of allowances allocated to the installations exceeding the amount given to the group. This amount may, for example, be determined by the group's total emissions. The number of allowances allocated must then be downscaled, which can be done in the following ways:

(a) Allocation adjusted so that the total number of allowances meets the amount given to the group (downscaling). See the set of bars on the right in the figure.

(b) Allowances from other groups or sectors are transferred over through a trading scheme, for example by multiplying all sectors by a scale factor.

(c) The total number of allowances in the trading scheme are increased (increasing the bubble) at the cost of non-trading sectors.

(d) Increasing the total number of allowances in the trading scheme and thereby increasing the national goals.

(e) The state buys more allowances on the open market.

Transferring allowances from other sectors. This involves transferring allowances from other sectors, not using international benchmarks. However, it is difficult to see the rationale in only using international benchmarks in certain sectors and then using the results as a reason to transfer allowances from other sectors. It is likely that such an approach would be perceived as unfair by sectors that do not use international benchmarking.

8.1.5.1. *Can International Benchmarking be Used for Sector Allocation?* International benchmarking can, in comparison with Swedish benchmarking, be done without first having to do a sector allocation. Allocation is done directly at the installation level and is based upon the installation's production and an international benchmark. In principle, allocation for all installations in a trading scheme can be based upon international benchmarks without knowing what the sector's allocation will be. If the result at the sector level, or national level, does not agree with the desired sector and national goals, the allocations can be adjusted as described above. The method is analogous to the description for installations above.

Step 1. International benchmarks at the installation level are determined. As many installations as possible should be included here.

Step 2. Allocation is calculated as an installation's production multiplied by the international benchmark. Allocation is aggregated to the sector level and national level.

Step 3. If there is a deficit of allowances, allocation can be adjusted as described above.

The principle is simple, but involves a number of complications:

– A large number of benchmarks must be set. It is not sufficient to only have one benchmark for each of the five trading sectors. A benchmark is needed for each product group. Operations differ greatly within sectors and a benchmark is necessary for each unique product that is produced. We believe that certain

sectors (iron and steel, mineral and refineries) will likely need to determine benchmarks for each installation.

– It is likely that there will be a group of installations where no benchmarks can be determined (eg. because of high costs, difficulties in finding international comparisons). These installations need to be handled in another way. In The Netherlands, the EU country that has come the farthest with benchmarking, benchmarks have been set for 60% of all installations. Installations that have benchmarks could possibly represent installations with similar operations. Problems in setting scheme boundaries and how to group installations could arise here. It should be possible to avoid some problems by, within a sector, clearly defining what supply and production are, and which processes are included.

Allocation between sectors will probably be different if benchmarking is used, compared with a case where emission-based allocation is applied.

8.2. THE PROBLEM WITH DIVIDED PRODUCTION CHAINS

An allocation scheme based on production has a built-in risk for manipulation when a value chain or a production chain is split between several installations. For example, an installation that produces 100,000 tonnes of paper has the right to allowances that reflect its production. In principle, the installation could be split into two installations put into series – where the first factory produces a semi-manufactured product, which is then sold to the second installation that manufactures the final product. 100,000 tonnes are produced by both installations and the total production has increased to twice as much, 200,000 tonnes. They thus have the right to receive twice as many allowances as before the split.

8.3. SUMMARY ASSESSMENT OF PRODUCTION-BASED ALLOCATION WITH SECTOR-SPECIFIC EMISSION FACTORS-BENCHMARKING

8.3.1. *Swedish Benchmarking*
  (a) Takes into consideration installations' possibilities to reduce their emissions.
  (b) Benefits installations that use carbon dioxide-efficient technologies.
  (c) Takes into consideration previously implemented carbon dioxide reduction actions.
  (d) Moderate costs to compile basic data.
  (e) Increased risk for stranded costs.
  (f) Installations with more carbon dioxide-efficient technologies than benchmarks receive more allowances than they need.
  (g) Can be difficult to define comparable product groups with a sufficient number of installations.
  (h) Can be difficult to define measures for production.
  (i) Can probably not be used for allocation between sectors.

8.3.2. *International Benchmarking*
 (a) Takes into consideration installations' possibilities to reduce their emissions
 (b) Benefits installations that use carbon dioxide-efficient technologies
 (c) Takes into consideration previously implemented carbon dioxide reduction actions
 (d) High costs to compile basic data (benchmarks)
 (e) Can possibly be used for allocation between sectors.

9. Production-Based Allocation with Emission Factors Based on Best Available Technology (BAT)

9.1. What is bat?

The EU Directive on Integrated Pollution Prevention and Control (IPPC) deals with co-ordinated actions to prevent and limit pollution. There are a number of documents presenting the results of the discussions between member states and affected industries regarding best available technology (BAT) and therefore also about continuous control of processes and emissions, as well as on development in both these respects. The IPPC directive states that these BAT documents should be taken in consideration when defining what the "best available technology" is.

The term "best available technology" is defined in the Directive's article 2.11 as "the most effective and advanced stage in the development of activities and their methods of operation which indicate the practical suitability of particular techniques for providing in principle the basis for emission limit values designed to prevent and, where that is not practicable, generally to reduce emissions and the impact on the environment as a whole". This definition is further clarified in article 2.11 with the following:
 – 'techniques' shall include both the technology used and the way in which the installation is designed, built, maintained, operated and decommissioned,
 – 'available techniques' shall mean those developed on a scale which allows implementation in the relevant industrial sector, under economically and technically viable conditions, taking into consideration the costs and advantages, whether or not the techniques are used or produced inside the Member State in question, as long as they are reasonably accessible to the operator,
 – 'best' shall mean most effective in achieving a high general level of protection of the environment as a whole.

9.2. Definition

We define BAT-based allocation in the following way:

BAT-based allocation for an installation is calculated as the installations' production multiplied by the specific emission, in tonnes of CO_2 per produced

unit, applicable for the best available technology for comparable installations, possibly adjusted with a scale factor

Mathematically, this can be described as:

$$A_{\text{installation}} = P_{\text{installation,prod.year}} \times e_{\text{BAT}} \times f$$

A = allocated allowances
P = production
e_{BAT} = the specific emission, in tonnes CO_2 per produced unit, applicable for the best available technology for such installations.
prod. year = the year chosen to define the production levels of an installation upon which allocation is based.
f = scale factor to adjust the installation's allocation so that the total number of allocated allowances is the same as the sector's allocation, $A_{\text{installation}}$.

There are large similarities between BAT-based allocation and allocation based upon international benchmarks. However, whereas international benchmarks are based upon the average emissions in the sector, BAT-based allocation is based upon what can be attained if the best available technology is used.

Figure 8 shows schematically how BAT-based allocation is done. For simplicity's sake, it is assumed that the installations in the figure are the same size in terms of production. The process is as follows:

Step 1. Each bar in the figure represents an installation (I1, I2,… I5). The set of bars to the left shows emissions for each installation. The BAT level of each installation is shown by the horizontal lines across the bars.
Step 2. Allocation is calculated as the installation's production multiplied by their BAT level (right set of bars).
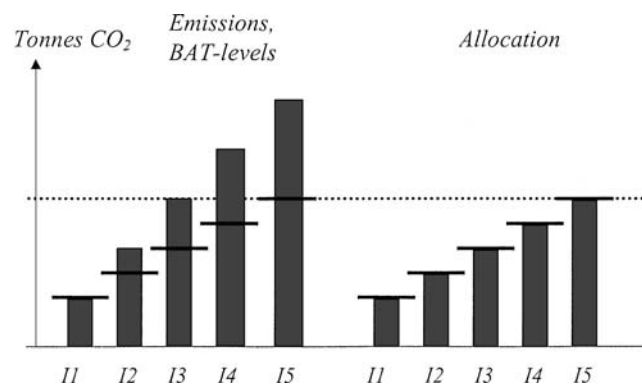

Figure 8. Schematic description of production-based allocation based on BAT.
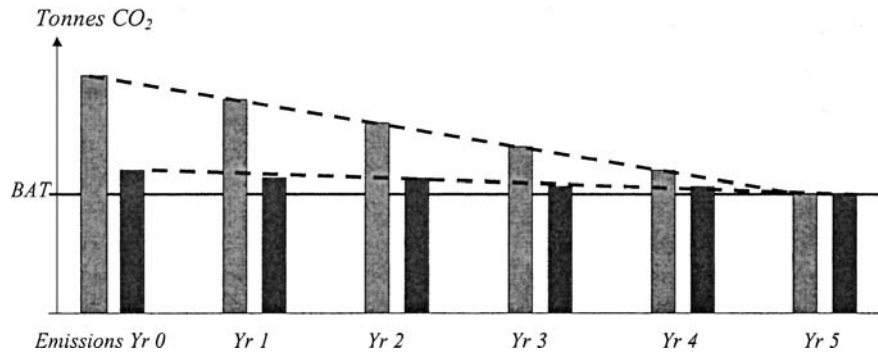
Figure 9. Schematic description of production-based allocation based on BAT with a "soft landing".

9.2.1. *BAT-based allocation with gradual reduction of specific emissions*
BAT-based allocation can lead to large one-time costs for those installations that are far from the BAT levels. In order to avoid this, an allocation can be designed such that it gradually shifts from the current emissions to the BAT levels, a kind of "soft landing". Figure 9 shows schematically how this can work. Two installations with the same BAT level have different emission levels at the start. The installation to the left is 100% above the BAT level, while the installation to the right is 20% over the BAT level. The allocation to each installation is decreased every year such that the BAT level is reached after five years.

Up-scaling within a group: The initial allocation can result in installations receiving fewer allowances than what the group is allowed to emit. This can be resolved by scaling up allocation for all installations so that, together, they meet the group allocation (see Figure 10).

Up-scaling means that installations 1 and 2 receive more allowances than what their emissions were, which can conflict with the EU Directive's Annex III criteria
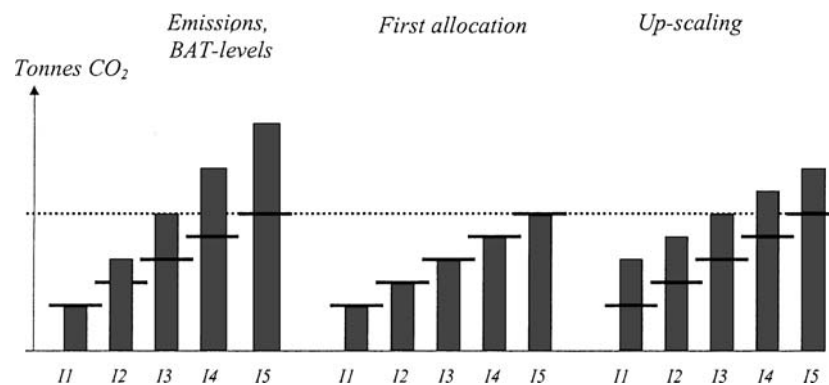

Figure 10. Schematic description of up-scaling after BAT allocation.

(see section 4). On the other hand, perhaps they have implemented emission-reducing actions earlier that they are now rewarded for.

9.2.2. *Important Issues*

  (i) Are there any basic data on BAT available today?
 (ii) Can BAT data be developed in the future?
(iii) Grouping installations with the same BAT levels.
(iv) Can BAT be used for allocation between sectors?

9.2.3. *Can Existing BAT Data be Used for Allocation of Allowances?*

The EU has published reference documents for BAT for, among others, the sectors to be included in the trading scheme. The documents are divided into different sections that describe various processes within the respective sectors. For each process, the relevant environmental parameters are examined, such as emissions of NO_x, SO_x, hydrocarbons, heavy metals and PAH. The emissions data for the various installations are also given. Suggestions are provided for the BAT levels for these different parameters, or technical solutions to help reach BAT levels. The document does not provide any final BAT levels, but rather aims to give a background document that can help member states set BAT levels in the future.

Carbon dioxide, in general, is not included as an environmental parameter in the document, but data on energy use is usually provided. These data, however, often do not specify lowest possible energy use for the various processes but instead provide typical energy use levels for an installation.

Our conclusion is that the existing BAT reference documents cannot be used to set BAT levels upon which allocation of allowances for the period 2005–2007 could be based.

9.2.4. *Development of BAT Data in the Future*

The majority of installations in Sweden that will be included in the trading scheme must apply for permits as per the Swedish Environmental Code, no later than 2007. The Swedish Environmental Code states that during this application process, demands will be placed on the installations to use the best available technology and decrease energy use. See the Swedish Environmental Code, chapter 2, sections 3 and 5:

3 § Persons who pursue an activity or take a measure, or intend to do so, shall implement protective measures, comply with restrictions and take any other precautions that are necessary in order to prevent, hinder or combat damage or detriment to human health or the environment as a result of the activity or measure. For the same reason, the best possible technology shall be used in connection with professional activities.

Such precautions shall be taken as soon as there is reason to believe that an activity or measure may cause damage or detriment to human health or the environment.

5 § Persons who pursue an activity or take a measure shall conserve raw materials and energy and reuse and recycle them wherever possible. Preference shall be given to renewable energy sources.

In other words, if it is applied strictly, The Swedish Environmental Code makes it possible for BAT levels to be set for all installations within the trading scheme, no later than 2007. BAT-based allocation could therefore be an interesting alternative as of 2008.

However, it should be kept in mind that BAT levels are not defined entirely unambiguously. BAT levels will be set for each unique installation in a dialogue between the company and responsible agency. The costs and benefits will have to be weighed. BAT levels do include a component of interpretation and valuation, and there is a clear risk of "regulator capture", ie. the regulator depends on information that can only be provided by the regulated installation itself.

It is unlikely that BAT levels will be set for the majority of installations in time for the first allocation (2005–2007). The cost for that to happen would be very high. There might, however, be some installations that already have basic data that can help set the BAT levels.

9.2.5. *Grouping of Installations with the Same BAT Levels*
We believe that it will be impossible to set BAT levels for each of the five sectors that will be included in the trading scheme. Operations differ greatly within each sector and a BAT level will have to be set for each unique product that is produced. We believe that certain sectors (iron and steel, mineral and refineries) will probably have to set unique BAT levels for each installation if they are to be relevant.

9.2.6. *Can BAT be Used for Allocation Between Sectors?*
There are large similarities with international benchmarking in this issue. BAT-based allocation occurs directly at the installation level without having to set the sector allocation first. If BAT-based allocation is used for all installations in a sector, the sector allocation will then be the sum of allocations for the sector's installations. The procedure is as follows:

Step 1. BAT-levels are set at the installation level.
Step 2. Allocation is calculated as an installation's production multiplied by the BAT level. Allocations are aggregated to the sector level and to the national level.
Step 3. It is likely that the sum of the initial allocation will be less than the bubble. An adjustment can then be made as described earlier by, for example, scaling up all allocations by a certain percentage.

If BAT is to be used to set allocation goals for all sectors, the BAT levels must be set for all sectors and for all relevant product groups. It will be a costly process. There will likely be a group of installations where no BAT level can be set. The

main advantage is that, as with international benchmarking, it opens the possibilities to do an allocation for installations and sectors that, better than emission-based allocation, reflects installations' and sectors' efficiencies in comparison with BAT. In this way, better consideration is taken for international competition and a sector's ability to reduce emissions, while also benefiting those installations with carbon dioxide-efficient technologies.

9.3. SUMMARY ASSESSMENT OF PRODUCTION-BASED ALLOCATION WITH EMISSION FACTORS BASED UPON BAT

(a) Takes into consideration installations' abilities to reduce their emissions.
(b) Benefits installations that use carbon dioxide-efficient technologies.
(c) Takes early action into consideration.
(d) Installations do not receive more allowances than they need. Can possibly happen if up-scaling is used to adjust the sector allocation.
(e) Can possibly be used for allocation between sectors.
(f) Installations far from the BAT levels risk having stranded costs.
(g) High costs for compiling basic data on best available technologies (BAT levels).

10. Other Allocation Schemes

There are many possible allocation schemes that are not described in this article. The selection of what allocation schemes that should be investigated in the study was done very pragmatically. We simply tried to choose those schemes that seemed most likely to be considered by the authorities.

10.1. UPDATING

10.1.1. *Definition*
An updated allocation is based on activities that the actor can affect. Allocation can, for example, be based upon activities in the current year, or the previous year, or on predicted activities for the coming year. As activities change over time, the allocation changes as well.

> *An updated allocation is based on activities that actors in the trading scheme can consciously affect and that affects allocation.*

10.1.2. *Important issues*
The most important issues with an updated allocation scheme are
  – Which activities will be the basis for the updating?
  – How and when will the total number of allowances be determined?
  – Can the scheme distort the market?

The choice of activity for an updated allocation is principally the same as the case with allocation based on historical activities. An updated allocation can be based upon emissions, production or supply (see section 6.1.1).

In the literature, production is the most common measure of activity, which is why updating is often, by default, considered as an output-based allocation method, even though this does not have to be the case.

Allocation can be done in advance, based upon predicted production. The actual production can be checked against the predicted production afterwards, and surplus allowances, if any, can then be returned. Allocation can also be done retroactively, based upon the actual production levels. However, as the proposed EU Directive requires the total number of allowances to be known in advance, this alternative may be difficult to implement.

In fact, any changes in an allocation scheme based on activities that occurred after it became known that an allocation scheme would be implemented, turn the scheme into an updated allocation scheme. If, for example, Sweden decides in 2005 to use an emission-based allocation scheme with base year 1990 and then in 2008 redoes the allocation using the same scheme but with base year 2006, this would also be an updated allocation scheme. The central issue is whether the actors in the scheme are aware that they can affect their future allocation by changing their activities.

Updated allocation can either be done within a given emission cap or as a scheme where the number of allocated allowances are unlimited. In the former, there is uncertainty for individual actors, as the number of allocated allowances is affected by other actors' actions. In the latter, an uncertainty is created for the authorities, since they do not know in advance how many allowances are to be allocated. The advantage with not having a cap on the number of allowances that can be allocated is that it is easier for new actors to enter the market, and that changes in the structure of the industry are not restrained by emission limits. However, as the EU Directive is written today, it is not allowed to have an allocation scheme where the total number of allowances is not known in advance.

Several studies on the trade of allowances have concluded that updating is unnecessary, that it distorts market forces and is economically inefficient, assuming that trade of allowances is allowed (Kerr 1999; Burtraw et al. 2002, Harrison and Radov 2002). This conclusion is based primarily upon the following reasoning: the most efficient strategy to cost-effectively reduce greenhouse gas emissions is to create a scheme with incentives to both decrease emissions per produced unit, and to decrease production of such units that give rise to emissions. Output-based allocation gives a scheme that encourages the reduction of specific emissions (emissions per produced unit) but it also creates an incentive for increased production; extra production is rewarded with extra allowances. In practice, this means production support. No pressure is put on moving from the production of carbon dioxide-intensive products towards products that produce less greenhouse gases. This means that in order to meet a given reduction goal, greater reductions in specific emissions must

be implemented than would be the case if, for example, a scheme based solely on historical emissions had been used. Since reductions in specific emissions are more costly the greater the reductions are, an updated output-based scheme would be economically inefficient and give higher costs to reach the reduction goal. Harrison and Radov (2002) refer to studies that show that the cost for a given reduction, measured as decreased surplus for consumers and producers, is two to three times higher in an updated scheme than in a grandfathered scheme or if auctioning is used. An updated production-based scheme can thus lead to inefficiencies in the market and create distortions in the trade between states.

However, although the reasoning given by, for example, Harrison and Radov is convincing, it is based on the idea of a perfect or near-perfect market situation. Since no such market exists, and it is difficult to tell how the real market will react, it may be that the advantages with an updated system may be larger than what theory predicts. One such advantage with updating is that adjustments can be made for changes in the structure of the industry, products or processes. Many of the problems that have been named in connection with output-based allocation become larger over time. An updated scheme can decrease these to a certain extent. It is also difficult to see how any scheme could not be updated at all, such as between commitment periods. This said, if updating is done too often, the problems with this scheme can be exacerbated.

10.1.3. *Summary Assessment of Updated Allocation*
  (a) Handles changes in the structure of the industry or production best of all studied schemes
  (b) Requires regular updating of data
  (c) Creates incentives that can reduce cost efficiency of reduction actions
  (d) Creates incentives that can give distortions in the market forces

10.2. AUCTIONING

When the work with this article commenced, it was believed that auctioning would not be an alternative as an allocation approach for the trading starting in 2005. Therefore, the study was limited to allocation schemes in which the allowances are allocated for free. However, the final version of the EU ETS Directive gives Member States the opportunity to auction 5% of the allowances allocated 2005–2007, and upto 10% of the allocated allowances 2008–2012. It can thus be of interest to do a thorough analysis of auctioning as an alternative allocation method.

The literature shows almost unequivocally that auctioning from a strictly economic perspective is more efficient than allocation based on historical activities, assuming that the returns from the auction are not wasted (FIELD 2002; Harrison et al. 2002.). Auctioning with recycling of the incomes through tax alleviation, for example, can reduce the costs to society by 95% in comparison with allocation based on historical activities (Kerr 1999). An argument against auctioning is

competition neutrality. If companies – Swedish or European – within the trading scheme are forced to pay for their allowances, while their competitors that have their operations outside the trading scheme but sell their products within the EU can avoid equivalent costs, an imbalance is created in the market.

10.3. ALLOCATION HANDLED BY TRADE ORGANISATIONS

A proposed alternative is for the state to decide upon the allocation of allowances between sectors, but allow the trade organisations to have responsibility for allocation within each sector. The argument for this is predominantly that trade organisations should be the most suited for developing a system that does not unfairly benefit certain actors over others. The system could gain greater acceptance within the sector, even if it is doubtful that fewer actors will feel mistreated than would be the case under any other allocation scheme.

11. Information Needs and Data Availability

The availability and quality of data is a clear bottleneck for the evaluation and implementation of an allocation scheme. In this study, data necessary to perform the allocation at the installation level, given the various allocation schemes has not been available. Table IV provides an overview of which data are necessary for the respective allocation schemes, which data exist today, possible ways to obtain remaining information, and estimates of what resources will be needed in order to do this.

With the exception of electricity and heat production and refineries, there is no information on emissions and production at the installation level, and there are data gaps at the sector level as well. It has also been shown that information varies depending on what sources are used. Trade organisations' data, for example, do not always agree with data from Statistics Sweden. The differences can sometimes be attributed to known differences in calculating methods or scheme boundaries, but not always.

Emission-based allocation requires the least data of the studied schemes. Of the production-based allocation schemes, those based on actor-specific emission factors or Swedish benchmarking have medium data needs, while allocation schemes based on international benchmarking and BAT require the highest levels of basic data.

12. Flow of Capital in Different Allocation Schemes

There are few empirical studies in the literature that look at the effects of the different allocation schemes on trade of allowances. The main reason for this is simply that there are too few real applications of such trading schemes in the world.

TABLE IV

Overview of data needs, status, solutions, and estimated resource needs to fill the data holes in the respective allocation schemes

Information	Status	Solutions	Required time
Bubble	Is not defined	For the time being, we assume a bubble size in order to be able to examine how allowances can be distributed between sectors and installations, given different allocation scheme.	–
Emission-based allocation to sectors			
Emissions sector 1990–2001	Electricity, heating, combined heat and power: 1990–2000 (Naturvårdsverket 2002), 2000 (Remes et al. 2002)	Two possible ways:	
	Refineries: 1990–1996 (SCB, 2002) 1990, 1995–2000 (Diczfalusy, 2002), 2000 (Remes et al. 2002)	i) Statistics Sweden compiles the information	1 month
	Pulp and paper: 1990–2000 (Naturvårdsverket 2002), 2000 (Remes et al. 2002)	ii) Contact all installations (approx. 300) and request the data. It is unlikely that the data set will be complete, particularly for early years.	2–4 months
	Iron & steel: 1990–2000 (Naturvårdsverket 2002), 2000 (Remes et al. 2002.		
	Cement, mineral: Data exists for 75 % of installations for the years 1990–1996 and 2000.		
	Industrial boilers: 2000 (Remes et al. 2002)		
Emission - based allocation to installations			
Emission installations 1990–2001	Electricity, heating, combined heat and power: 2000 (Remes et al. 2002)	Two possible ways:	
	Refineries: 1990, (Diczfalusy 2002), 1990–1996 (SCB, confidential)	i) Contact all installations (approx. 300) and request the data. It is unlikely that the data set will be complete, particularly for early years.	2–4 months
	Pulp and paper: 1990–1996, 2000, 54 of 61 installations (SCB, 2002, confidential)	ii) Revoke statistical secrecy and have Statistics Sweden deliver the data. NB: It is impossible to separate emissions from small boilers (larger than 20 MW) from Statistics Sweden's statistics.	1 month
	Iron & Steel, 1990–1996, 2000, (SCB, 2002, confidential)		
	Cement and Mineral: 1990–1996, 2000, 15 of 20 installations (SCB, 2002, confidential)		
	Industrial boilers: no data on installation level		

Production-based allocation to installations–Actor-specific emission factors

Product definition	Electricity, heating, combined heat and power: A preliminary product group definition has been done and checked with the sector. A definitive product group definition needs to be done.	Contact all installations (approx. 300) and request the data. Co-ordinate with data collection of emissions.	2–4 months
Production installations 1996–2000	Heating, combined heat and power: Data exists at the installation level (Swedish District Heating Association, 2002) Condensing power, gas turbines: Data on installed capacity exists at the installation level for 2000 (ÅF, 2002).	Contact all installations (approx. 300) and request the data. Co-ordinate with data collection of emissions.	2–4 months
Emissions installations 1990–1993	Refineries: Data exists for 1990. Source: Confederation of Swedish Enterprise	Two possible ways: i) Contact all installations (approx. 300) and request the data. Given the results from previous years, it is unlikely that the data set will be complete. ii) Maintain statistical secrecy and have Statistics Sweden deliver the data. NB: It is impossible to distinguish emissions from small boilers (less than 20 MW) from Statistics Sweden's statistics.	2–4 months

Production-based allocation to installations with Swedish sector-specific emission factors (Swedish benchmarking). Only the energy sector.

Product group definition	Electricity, heating, combined heat and power: A preliminary product group definition has been done and checked with the sector. A definitive product group definition needs to be done.	Contact Swedenergy AB, Swedish District Heating Association.	1 month
Production installations 1996–2001	Heating, combined heat and power: Data exists at the installation level (Swedish District Heating Association, 2002) Condensing power, gas turbines: Data on installed capacity exists at the installation level for 2000 (ÅF, 2002).	Contact Swedenergy AB, Swedish District Heating Association. Likely no large difficulties to compile the production data.	1 month
Emissions sector 1996–2001	Electricity, heating, combined heat and power: Data exists for all years. (Swedenergy AB, Swedish District Heating Association)		None

TABLE IV
(*Continued*)

Information	Status	Solutions	Required time
Production-based allocation to installations with international sector-specific emission factors (International benchmarking)			
Benchmarks	No benchmarks defined yet. Several installations probably have basic data that could help when setting benchmarks (e.g., Scanraff, Fortum).	If benchmarking is to be used for all installations, 100–200 benchmarks will have to be set. In the Netherlands, it is estimated that approximately 130 benchmarks will cover the whole trading scheme. Their first benchmark took approx. 2 yrs to set, while their latest took about one year. The cost has been around 1 million SEK per benchmark, a cost that is often shared by the various installations. Sweden might be able to learn from the Netherlands and save both time and resources.	6–24 months per benchmark.[a]
Production installations 1996–2001	Electricity, heating, combined heat and power, industrial counter pressure: Data on installed capacity at the installation level for the year 2000 exists (ÅF). Refineries: No data for production yet. Steel, roasting, sintering: Production data for 9 of 15 installations for the year 2001 exists. (ÅF) Mineral: Production data for 18 of 20 installations for the year 2001 exists (ÅF). Pulp and paper: Production data for all 61 installations for the year 2001 exists (ÅF).	Contact the installations as the benchmarks are being determined. No likely large difficulties to compile the production data.	3–6 months

Production-based allocation to installations based on BAT levels (BAT-based allocation)

BAT levels	EU's reference documents on BAT can not be used for allocation today. Some installations have basic data that could help when determining the BAT levels.	According to the IPPC Directive, a large number of installations within the trading scheme will be adjusted to BAT by 2007. This work can be used as basic data in future allocations. BAT levels are not clearly defined and will give a technical level that will minimise environmental impacts at a reasonable cost. BAT levels will be installation-specific. BAT also includes a component of interpretation and valuation. If BAT is to be used as the basis for allocation, it is important that different installations receive BAT levels under the same conditions. Determining the BAT levels for the first allocation period (2005–2007) will likely be very costly and comparable to setting international benchmarks.	Earliest 2008
Production installations 1996–2001	See International Benchmarking		

The bubble, i.e. total amount of allowances in the Swedish trading scheme.

[a] In The Netherlands, international benchmarks have been developed for many installations in connection with the Energy Covenants and the long term agreements between the Dutch Government and industry. The amount of time required to determine a benchmark varies, but in general it takes approximately one year to set a benchmark (Vencken 2002; Brinkhoff 2002). The Dutch scheme is based upon energy efficiency, but the same strategy should be feasible for carbon dioxide efficiency, even if it would be somewhat more complicated.

One of the goals with this study was to calculate the effects of different allocation schemes, in the form of capital flows between actors in the trading scheme.

In theory, a rough modelling of capital flows is quite simple. A market price for allowances is set and then combined with the costs for reduction actions for the different actors. The driving force being that each installation will strive to implement actions as long as they are cheaper than the market price for allowances. If this means that the actor's total emissions drop lower than their allocation, the actor will then be able to sell their allowances.

Data on the costs of reducing greenhouse gas emissions from installations in Sweden have, however, proven to be difficult to obtain. There is little published material and those materials that do exist are not complete and not always relevant to the purpose at hand. The most comprehensive studies of European reduction costs were done on behalf of the European Commission and published in 2000. In three reports, the costs for different sectors to implement reduction actions and the reduction potential of the respective actions are presented (Hendriks and de Jager 2001; de Beer et al. 2001; Hendriks et al. 2001). From these data, it is possible to compile estimated costs for the Swedish sectors. The majority of the presented actions are very cheap and some can even generate positive returns.

Sector experts, trade organisations, and individual companies that were contacted were almost in full agreement that the data presented by the European Commission's studies are for the most part non-representative or irrelevant for the Swedish situation. In addition, the basic assumptions for some of the concepts, such as energy prices and discounting rate can be questioned. For these reasons, the contacted people argued that the published costs for reduction actions should not be used for the purpose of this study.

The conclusion from the literature review and sector contacts is that there are no existing estimates of reduction costs of sufficient quality to be useful in calculating capital flows between sectors and installations. In order to be able to make such calculations with reasonable precision, a more detailed study of the costs for reduction actions of those installations that will be included in the Swedish trading scheme is necessary. Such a study is probably possible to do, but is beyond the scope of the current investigation.

An alternative to specific reduction costs could be to assume that the marginal cost for reducing carbon dioxide emissions today is equal to the marginal cost for the emission of carbon dioxide. This implies that the reduction cost is the same as the carbon dioxide tax in those sectors where such a tax is paid, and zero in other sectors. The problem is that it is very difficult to forecast how reduction costs would develop once reductions are carried through. The question is "how fast will the costs per reduced tonne of carbon dioxide increase after the first kilo is reduced?"

With today's data, it is only possible to estimate the *maximum* costs for an actor. These are equal to the reduction actions multiplied by an estimated market price for allowances. Those sectors that pay carbon dioxide tax today will likely be subjected to the maximum cost, because the price for allowances will likely be lower than

today's taxes. It is still possible to see that the actual costs for the actor will be lower than in the current scheme. For those actors that do not pay any carbon dioxide tax, it is harder to foresee how large a portion of the maximum cost the actor will have to pay.

The reader should be aware that, albeit very important, technical costs are not the only factor determining capital flows. The shape of a cost curve is dependent on what factors are included in the cost, and what basic assumptions are made when determining the cost. It can also be argued that the costs for carbon dioxide-reducing actions are likely to be lower than what is indicated by actual implemented actions in the industry. Since the time required to pay off reduction measures may be longer than for other investments they tend to choose investments in other kinds of actions that do not affect carbon dioxide emissions. The fact that companies have limited financial resources and that the emission market is still undeveloped which adds risk to investment, further increases the likelihood that companies may prioritise other types of investments.

In summary, we are forced to conclude that within this study, it is impossible to model capital flows in the trading scheme. In order for modelling to be possible, additional efforts would need to be made to determine reduction costs.

13. Other Issues

13.1. New entrants and changes in existing installations' operations

"New entrants" means those that come into the market after the year upon which the allocation of allowances is based. These installations must either be allocated allowances for free, or be forced to buy allowances on the open market in order to be able to run their operation. Even an existing installation that wants to increase its operation will require more allowances. In principle, there is no difference between a new entrant and increased operations at an existing installation.

It can thus be difficult to treat new installations differently from increased production. If, for example, a new installation where to receive its allowance for free, while an existing installation were required to pay for its increase in production, the result could be that changes in operations would instead be said to be new operations and installations. The same reasoning can be transferred to closures of operations or decreases in production.

The main argument brought forward for new entrants receiving free allowances is that new actors should not be excluded from the market because of costs that existing actors did not have (assuming they received their allowances for free). If new entrants are hindered, the development of new and better technologies could be curbed, for example, and ultimately the efficiency of the trading scheme as a way to decrease emissions could decrease. In the same way, it can be argued that the potential growth of a company should not be limited by the cost of new allowances.

The most important counter-argument in the literature builds upon economic

theory. The economic incentives to develop new and more CO_2-efficient technologies are the same regardless of whether a new installation is given allowances for free or if it is forced to pay for them. An actor has much to gain by reducing its CO_2 emissions regardless of whether allowances are free or not. Either the actor can sell an allowance and make money, or avoid buying allowances because it doesn't need them. The economic effect, the so-called opportunity cost, is identical in either case. In addition, new installations do not carry the cost of previous investments in CO_2-inefficient technologies, so-called stranded costs. If the allocation scheme does not fully compensate stranded costs, this could compensate new installations to a certain extent for having to buy allowances.

In summary, economic theory shows that the choice between free allocation or not will not affect CO_2 efficiency in the trading scheme, or the result in the form of investments and technological advances. Cost effectiveness and environmental impacts are not affected either, because from this point of view, it makes no difference if it is a new or old installation that is releasing the carbon dioxide or investing in the new technology, respectively. On the other hand, whether new entrants have to pay for their allowances or not could have an important distributional effect, that is which actors will be the winners and the losers.

When new entrants are allocated allowances, there are several alternatives:

 (i) New entrants must buy allowances on the open market.
 (ii) The state "saves" a number of allowances from the first allocation. These saved allowances can then be offered to later installations or increased productions, either free or for sale. Such a process would put higher reduction requirements on existing installations. There is still the possibility of surplus or deficit of allowances if the number of saved allowances does not match the need from new installations.
(iii) The state is allowed to buy new allowances on the open market, either from the EU or in the form of project-related allowances (JI/CDM project). These allowances can then be allocated to new installations, or installations that want to increase their production. The costs for such a scheme are difficult to estimate. There is also a risk that allocation of allowances in this way would be counted as state aid and therefore not be allowed under EU-legislation.
(iv) The state issues as many allowances as the new entrants need, without setting a ceiling for the total number of allocated allowances. The reduction requirements are then increased to the equivalent amount in non-trading sectors. Given the size of the sectors included in the scheme, this could lead to unreasonable reduction requirements on the other sectors.
 (v) The value of allowances is decreased during the trading period, in pace with the rate at which allowances are allocated to new installations. This could, however, create uncertainty in the scheme and also benefit installations that enter the scheme late.

The EU Directive states that the national allocation plans must include information on how new installations will gain access to the market. There are, however,

no direct instructions on *how* this should be achieved. There are other criteria in the proposed Directive that very likely rules out some of the alternatives presented above. The most important is that it should be known *in advance* how many allowances will be allocated and to which installations. This can eliminate alternatives four and five in the list above.

13.2. COMBINED HEAT AND POWER

In Annex III of the EU Directive, it is stated that allocation should give consideration to carbon efficient technologies such as combined heat and power. Allocation to combined heat and power is difficult because the same installation produces two different products from energy resources that cannot physically be allocated to one or the other product. In other words, it is impossible to specify what proportion of the input, the fuel, is used to produce the electricity or heat. There are several different ways to deal with this problem. Furthermore, there are also combined heat and power installations that are not considered as part of the energy sector. These are usually called industrial combined heat and power. Whether these installations will be considered as part of the energy sector or not has not been decided, and this can affect how an allocation scheme should be formed. There can be reason to treat industrial combined heat and power differently compared with conventional combined heat and power. Industrial combined heat and power does not have the same possibilities to optimise its energy use as does a conventional combined heat and power plant, whose only purpose is to produce electricity and heating. Which sector industrial combined heat and power should be considered to belong is not dealt with in this study.

14. What Can a Complete Allocation Scheme Look Like?

Allocation will in part be an iterative process. First, it is necessary to define what types of installations that should be included in the trading scheme. After that, the allocation process can be divided into the following steps:
  (a) Definition of sectors, ie., grouping of the trading installations, and initial allocation to the sectors.
  (b) Initial allocation to the installations. Sum up the allocated number of allowances at the sector level and for the entire trading scheme.
  (c) Adjustment of the allocation at installation level so that the sector goals and national goals are met.
    The criteria for the allocation under the EU ETS Directive also stipulates a public consultation process. The consultation process will in practice be focussed on the participants in the ETS and will contain a significant amount of lobbying activities. This may well affect the final outcome of the allocation, but will not be discussed further here.
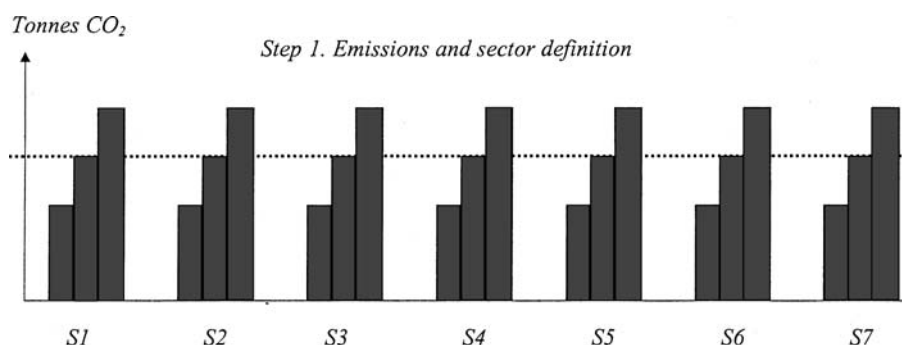
Figure 11. Schematic description of emissions in seven sectors S1–S7, each with three installations.

Note that the first step is not necessary in all allocation schemes. It is possible to do an allocation at installation level directly, without defining groups or sectors first.

It is not necessary to use the same allocation method throughout the whole scheme. One method can be used for allocation between sectors, while allocation between installations within a sector can be based on another method. The choice of the method will be a balance between how appropriate it is to its purpose, and how much it costs.

The example below shows what a complete allocation scheme can look like. We assume that we start with a given, fixed, number of allowances and must allocate these to the sectors and installations.

Step 1. Define the sectors. Figure 11 shows schematically the installations in the trading scheme. Each bar represents an installation and the bar height illustrates the installation's emissions. The bars are sorted into seven sectors (S1, S2,... S7), with three installations in each sector. For simplicity's sake, all sectors have equal total emissions. All installations have the same size production, but the installations' emissions, and therefore their specific emissions, differ within each sector.

Step 2. Do an allocation to the sectors. In our example, we have used emission-based allocation with the same base year as the year the emissions data were given for. This means that the sector allocation reflects exactly the emissions. Other allocation methods other than emission-based allocation are, of course, possible here too (Figure 12).

Step 3. Do an initial allocation to the installations. Here, we have used several different allocation methods. For each sector, the allocation method that keeps the highest level of quality as possible (as per the Flex Mex delegation's criteria and Annex III criteria) at an acceptable cost is chosen. Sectors 1 and 2 use emission-based allocation with the same base year as for sector allocation. This means that each installation receives exactly as many allowances as they had emissions during the base year. Sector 3 uses benchmarking within the sector as its allocation method. This means that all installations receive allowances
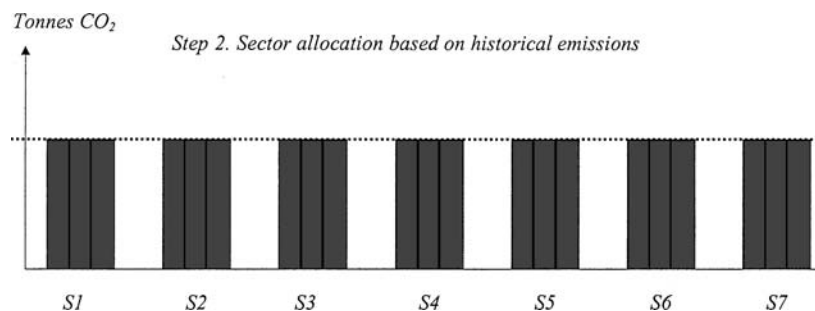
Figure 12. Schematic sector allocation based on historical emissions. Because sectors S1–S7 all had the same amount of emissions, their allocations are all alike as well.
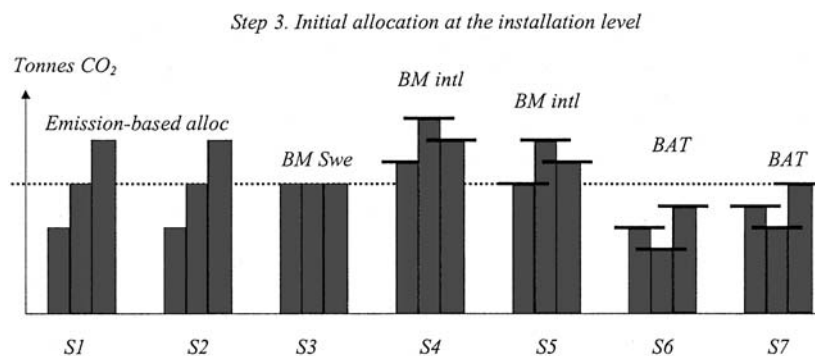

Figure 13. Schematic allocation at the installation level, before adjustments.

reflecting their sector's specific emissions. Sectors 4 and 5 use international benchmarking. This means that a benchmark must be set for each installation. Since the total number of allowances is higher than the original sector allocation, there is a deficit of allowances in the scheme. Sectors 6 and 7 use installation-specific BAT levels for their allocation. The initial allocation resulted in the total number of allowances being less than the original sector allocation and a surplus of allowances in the scheme (Figure 13).

Step 4. Adjust allocations at the installation level so that the sector goals and national goal are met. In the above allocation, sectors 4 and 5 receive more allowances, and Sectors 6 and 7 fewer allowances than in the original sector allocation. This can be solved in several ways that have been described earlier: (i) through scaling within the sector; (ii) through transferring allowances between sectors; (iii) through increasing/decreasing the total number of allowances in the trading sectors (i.e., changing the size of the bubble) at the cost of non-trading sectors in Sweden; or (iv) through increasing/decreasing the total number of allowances in the trading sectors by changing the national goal. In the figure below, we have made adjustments in each sector so that the original sector allowances are kept (Figure 14).
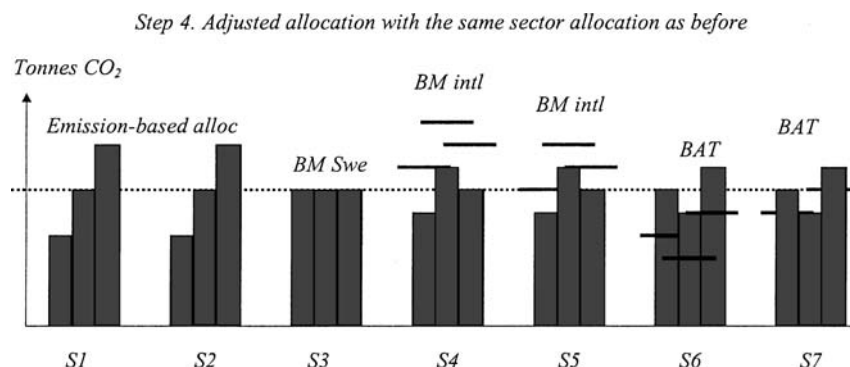
Figure 14. Schematic allocation at the installation level, after adjustments.

Is it not feasible to transfer allowances between sectors? In our example, we see a surplus of allowances in sectors with international benchmarking (S4, S5) that is exactly equal to the deficit in sectors with BAT-based allocation (S6, S7). If we accept the initial allocations, the size of the bubble remains unchanged, but in practice, it means that the BAT sectors (S6, S7) have transferred their allowances from their original sector allocation to the sectors that used benchmarking (S4, S5). Doing this is does have problems, however. To transfer allowances from the BAT sectors to the benchmarking sectors could conflict with the Annex III criteria that state that consideration should be taken to the installations' possibilities to reduce their own emissions. The installations in the BAT sectors (S6 and S7), by definition, have difficulties reducing their emissions below the BAT-levels. The installations in the benchmarking sectors (S4 and S5) have much greater possibilities of reducing their emissions below their respective benchmarks, since the international benchmarks give an average value for specific emissions, not the lowest possible level.

15. Conclusions

Each alternative allocation scheme has its own advantages and disadvantages. None is perfect, and it is unlikely that all actors will perceive any allocation scheme as fair.

The administrative costs associated with implementing emission-based allocation are lower than those associated with production-based schemes, but emission-based allocation do not meet all Annex III criteria. The production-based allocation schemes better fulfil those criteria but demand more resources, particularly schemes based upon international benchmarking or BAT. The production-based allocation schemes are also more difficult to understand and communicate, and require more time and preparation to be possible to implement.

The characteristics of each investigated allocation scheme are summarised in Table V and Table VI.

TABLE V
Assessment of emission-based allocation

Annex III –Criteria[a]	Emissions, early base year	Emissions, late base year
1. The total quantity of allowances to be allocated for the relevant period shall be consistent with the Member State's obligation under the burden sharing agreement.	Not analysed	Not analysed
2. The total quantity of allowances to be allocated shall be consistent with assessments of actual and projected progress towards fulfilling the Member States' contributions to the Community's commitments made pursuant to Decision 93/389/EEC.	Not analysed	Not analysed
3. Allocation shall be consistent with the potential to reduce emissions.	Not OK	Not OK
4. The plan shall be consistent with other EC legislative and policy instruments.	Not analysed	Not analysed
5. The plan shall not unduly favour certain undertakings or activities.	Can pose problems	OK
6. The plan shall contain information on the manner in which new entrants will be able to begin participating in the Community scheme in the Member State concerned.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.
7. The plan may accommodate early action.	OK	Not OK
8. The plan shall contain information on the manner in which clean technology, including energy efficient technologies, are taken into account	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.
9. The plan shall include provisions for comments to be expressed by the public.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.
10. The plan shall contain a list of the installations covered by this Directive with the quantities of allowances intended to be allocated to each.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.

(*Continued on next page*)

TABLE V

(*Continued*)

Annex III–Criteria[a]	Emissions, early base year	Emissions, late base year
11. The plan may contain information on the manner in which the existence of competition from countries/entities outside the EU will be taken into account	Not analysed	Not analysed
Other issues		
Feasibility	OK	OK
Data availability	Medium	OK
Costs for compiling data	Medium	Low
Risk of Stranded Costs	High	Low

[a]There are a number of other criteria that a National Allocation Plan must meet. Among these can be mentioned aspects like state aid, legislation on competition, compatibility with energy policy, how the trading scheme work in parallel to other relevant taxes, subsidies and regulations, and the concerned parties possibilities to give comments on the allocation plan.

It is not possible to accurately model capital flows between the trading sectors, the main reason being lack of abatement cost curves.

Data availability will most probably limit the options available to the authorities designing the allocation schemes.

16. Further Work

The most important issues still to resolve in the context of designing a National Allocation Plan are

(a) Data on production and emissions needs to be assembled at the installation level in order to be able to calculate allocation at that level. The most important issue to resolve before a final allocation scheme can be formed is compiling basic data that show allocation at the installation level under the different potential allocation schemes. This is for the most part a question of information availability. In this study, data on emissions and production have not been available to the extent that is necessary to determine allocation at the installation level. Confidentiality is a problem for individual installations that must be dealt with if new data are to be assembled. It is unclear how far back in time data exists at the installation level. For a summary of which data are available, and an estimate of what resources will be needed to assemble the remaining data, see Table VI.

(b) International benchmarking and BAT need to be examined further, especially with respect to the resources necessary to implement these schemes. What

TABLE VI
Assessment of production-based allocation

Annex III-Criteria[a]	Production, actor-specific emission factors	Production, sector specific emission factors (Swedish Benchmarking)	Production, international emission factors (International Benchmarking)	Production, BAT
1. The total quantity of allowances to be allocated for the relevant period shall be consistent with the Member State's obligation under the burden sharing agreement.	Not analysed	Not analysed	Not analysed	Not analysed
2. The total quantity of allowances to be allocated shall be consistent with assessments of actual and projected progress towards fulfilling the Member States' contributions to the Community's commitments made pursuant to Decision 93/389/EEC.	Not analysed	Not analysed	Not analysed	Not analysed
3. Allocation shall be consistent with the potential to reduce emissions.	Not OK	OK	OK	OK
4. The plan shall be consistent with other EC legislative and policy instruments.	Not analysed	Not analysed	Not analysed	Not analysed
5. The plan shall not unduly favour certain undertakings or activities.	Can pose problems[b]	Can pose problems[b]	Can pose problems[b]	Can pose problems[b]
6. The plan shall contain information on the manner in which new entrants will be able to begin participating in the Community scheme in the Member State concerned.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.
7. The plan may accommodate early action.	OK	OK	OK	OK
8. The plan shall contain information on the manner in which clean technology, including energy efficient technologies, are taken into account	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.
9. The plan shall include provisions for comments to be expressed by the public.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.

TABLE VI
(Continued)

Annex III-Criteria[a]	Production, actor-specific emission factors	Production, sector specific emission factors (Swedish Benchmarking)	Production, international emission factors (International Benchmarking)	Production, BAT
10. The plan shall contain a list of the installations covered by this Directive with the quantities of allowances intended to be allocated to each.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.	No differences between the studied allocation schemes.
11. The plan may contain information on the manner in which the existence of competition from countries/entities outside the EU will be taken into account	Not analysed	Not analysed	Not analysed	Not analysed
Other issues				
Feasibility	Energy: OK Pulp & paper: Not OK Steel, mineral, refin.: Possibly Between sectors: Not OK	Energy: OK Pulp & paper: Not OK Steel, mineral, refin.: Not OK Between sectors: Not OK	OK[c]	Not OK
Data availability?	Medium	Energy: OK Pulp & paper: unknown Steel, mineral, refin.: n/a	Not OK	Not OK
Costs for compiling data	Low	Low	High	High
Risk of Stranded Costs	Medium	Low[d]	Low[d]	Low[d]

[a]There are a number of other criteria that a National Allocation Plan must meet. Among these can be mentioned aspects like, stringency with national and EU commitments under the Kyoto protocol, state aid, legislation on competition, compatibility with energy policy, how the trading scheme work in parallel to other relevant taxes, subsidies and regulations, and the concerned parties possibilities to give comments on the allocation plan.

[b]The EU Commission states in its non-paper on the National Allocation Plan that early action that has already received government support cannot be compensated again for those actions. This could mean that benchmarking may not be allowed by the Commission.

[c]Feasible, but high costs.

[d]Installations with specific emissions that are far from benchmark or BAT may be subject to stranded costs.

costs are associated with these methods? What are the experiences from the Dutch scheme with long-term agreements for energy efficiency, a scheme that resembles the approach used in this study? For which installations is this a viable solution? What advantages can be expected in the actual application? By learning more from the experiences of the Dutch scheme with

benchmarks, and from the American RECLAIM trading-scheme (RECLAIM 2003), it should be possible to get good indications of the costs associated with application, how long it takes to set benchmarks, and for which types of installations these methods are viable. Discussions should therefore be taken up with companies, trade experts and sector representatives to identify, which installations in Sweden can use benchmarks.

(c) What happens if an installation that has been allocated allowances shuts down or moves abroad? This question needs to be examined further. If, for example, allocations discontinue upon moving, how do you distinguish between a normal decrease in production and moving? Experiences can be gained from the American trading scheme and a deeper analysis beyond what was done in this study can be done.

(d) Each sector has specific problems and issues that should not be underestimated. An example is combined heat and power, which was discussed briefly above and is specifically mentioned in the ETS Directive. Another problem that is named in the ETS Directive is installations that can change their operations as a consequence of other EU legislation. Refineries, for example, have changed their products considerably over the past decade, in part as a result of changed legal requirements. Further, it is plausible that individual installations can have changed their entire operation between the planned base year and the start of the trading scheme. The list is long and specific issues exist in all sectors. The character of the problems and their solutions will probably vary from political deliberation to pure technical assessments.

(e) Data on abatement costs must be improved in order to be able to estimate the economic consequences.

(f) Other allocation schemes, notably auctioning, and their consequences, need to be examined further.

Acknowledgements

References

Axelsson, L.-E.: 2002, *Personal Communication,* Swedish Forest Industries Federation, Box 16006, SE-103 21 Stockholm, Sweden.

Brinck, L.: 2002, *Personal Communication*, Scanraff Skandinaviska Raffinaderi AB Scanraff, SE-453 81 Lysekil, Sweden.

Brinkhoff, J.: 2002, *Personal Communication,* Ministry of Economic Affairs, The Netherlands.

Burtraw, D., Palmer, K., Bharvikar, R. and Paul, A.: 2002, 'The effect on asset value of the allocation of carbon dioxide emission allowances', *Discussion Paper 02-15,* Resources for the Future, Washington D.C.

Capros, P. and Mantzos, L.: 2000, *The Economic Effects of EU-Wide Industry-Level Emission Trading to Reduce Greenhouse Gases – Results from PRIMES Energy Schemes Model.* Institute of Communications and Computer Schemes of National Technical University of Athens.

de Beer, J., Phylipsen, D. and Bates, J.: 2001, *Economic Evaluation of Carbon Dioxide and Nitrous Oxide Emission reductions in Industry in the EU, Bottom up Analysis,* Ecofys, The Netherlands, AEA Technology Environment, United Kingdom.

Diczfalusy, B.: 2002*, Personal Communication*, Confederation of Swedish Enterprise, SE-114 82 Stockholm, Sweden.

EU: 2003, *Directive 2003/.../EC of the European Parliament and of the Council Establishing a Scheme for Greenhouse Gas Emission Allowance Trading Within the Community and Amending Council Directive 96/61/EC*, Provisional, unofficial version.

FIELD: 2002, *Designing Options for Implementing an Emissions Trading Regime for Greenhouse Gases in the EC,* Commission for the EU Commission, Brussels, Belgium.

Granström, P.-O.: 2002, *Personal Communication,* Swedenergy AB, SE-101 53 Stockholm, Sweden.

Harrison, D. Jr. and Radov, D.B.: 2002, *Evaluation of Alternative Initial Allocation Mechanisms in a European Greenhouse Gas Emissisions Allowance Trading Scheme,* NERA, London.

Hendriks, C., de Jager, D., de Beer, J., van Brummelen, M., Blok, K. and Kerssemeekers, M.: 2001, *Economic Evaluation of Emission Reduction of Greenhouse Gases in the Energy Supply Sector in th EU, Bottom- up Analysis,* Ecofys, The Netherlands.

Hendriks, C. and de Jager, D.: 2001, *Economic Evaluation of Methane Emissions Reduction in the Extraction, Transport and Distribution of Fossil Fuels in the EU,* Ecofys, The Netherlands.

Kerr, S.: 1999, *Allocation of Greenhouse Gas Reduction Responsibilities Among and within the Countries of the European Union,*Center for Clean Air Policy, Scoping paper No. 5 to the European Commission, DG XI.

KPMG: 2002, *Allocation of CO_2^- Allowances – Distribution of Emission Allowances in a European Emissions Trading Scheme,* Report to the Ministry of Economic Affairs, The Netherlands.

Larsson, E.: 2002, *Personal Communication*, Swedish District Heating Association, SE-101 53 Stockholm, Sweden.

Lindblad, B.: 2002, *Personal Communication,* Swedish Steel Producers' Association, Box 1721, SE-111 87 Stockholm, Sweden.

Naturvårdsverket.: 2002, Data from Swedish National Report to UNFCCC.

Remes, M., Wiklund, S.-E. and Åkesson, H.: 2002, *Kartläggning av Svenska Anläggningar Som Omfattas av ett Scheme för Handel Med Utsläppsrätter,* Report to the FlexMex 2 Commission. ÅF-Energikonsult, Flemminggatan 7, Box 833, SE-104 20 Stockholm, Sweden.

RECLAIM: 2003, For a description of the Regional Clean Air Incentives Market, see e.g. www.aqmd.gov/reclaim/reclaim.html

SCB: 2002, Rudander, H. and Palm, V.: *Personal Communication,* Environmental programme, Statistics Sweden, Box 24 300, SE-104 51 Stockholm, Sweden.

SFT: 2000, *Reduksjon av Klimagassutslipp i Norge, en Tilltaksanalyse for 2010,* Statens forurensningstilsyn, Norway.

SOU: 2003, *Handla för Bättre Klimat,* Delbetänkande av FlexMex2-Utredningen, Statens Offentliga Utredningar SOU2003:60, Fritzes Förlag, 106 47 Stockholm, Sweden.

Swedish Environmental Protection Agency.: 2002, *Processutsläpp för Mineralindustrin och Stålindustrin*, Report to UNFCCC, Swedish EPA website: www.naturvardsverket.se

Vencken, J.: 2002, *Personal Communication,* Verificatiebureau Benchmarking Energie-Efficiency (Vbe), The Netherlands.

Wallin, S.: 2002, *Personal Communication*, Shell.

Åhman, M., Zetterberg, L., Enroth, M., Widing, A., Brohammer, G., Stadig, M., Beyer, G. and Johansson, R.: 2001, *Miljöindikatorer för Näringslivet,* IVL-rapport B1450, IVL, Box 210 60, 100 31 Stockholm, Sweden.

Paper VIII

CLIMATE IMPACT FROM PEAT UTILISATION IN SWEDEN

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Abstract. The climate impact from the use of peat for energy production in Sweden has been evaluated in terms of contribution to atmospheric radiative forcing. This was done by attempting to answer the question 'What will be the climate impact if one would use 1 m^2 of mire for peat extraction during 20 years?'. Two different methods of after-treatment were studied: afforestation and restoration of wetland. The climate impact from a peatland – wetland scenario and a peatland – forestation – bioenergy scenario was compared to the climate impact from coal, natural gas and forest residues. Sensitivity analyses were performed to evaluate which parameters that are important to take into consideration in order to minimize the climate impact from peat utilisation. In a 'multiple generation scenario' we investigate the climate impact if 1 Mega Joule (MJ) of energy is produced every year for 300 years from peat compared to other energy sources.

The main conclusions from the study are:

- The accumulated radiative forcing from the peatland – forestation – bioenergy scenario over a long time perspective (300 years) is estimated to be 1.35 mJ/ m^2/ m^2 extraction area assuming a medium-high forest growth rate and medium original methane emissions from the virgin mire. This is below the corresponding values for coal 3.13 mJ/ m^2/ m^2 extraction area and natural gas, 1.71 mJ/ m^2/ m^2 extraction area, but higher than the value for forest residues, 0.42 mJ/ m^2/ m^2 extraction area. A 'best-best-case' scenario, i.e. with high forest growth rate combined with high 'avoided' methane (CH$_4$) emissions, will generate accumulated radiative forcing comparable to using forest residues for energy production. A 'worst-worst-case' scenario, with low growth rate and low 'avoided' CH$_4$ emissions, will generate radiative forcing somewhere in between natural gas and coal.

- The accumulated radiative forcing from the peatland – wetland scenario over a 300-year perspective is estimated to be 0.73 –1.80 mJ/ m^2/ m^2 extraction area depending on the assumed carbon (C) uptake rates for the wetland and assuming a medium-high methane emissions from a restored wetland. The corresponding values for coal is 1.88 mJ/ m^2/ m^2 extraction area, for natural gas 1.06 mJ/ m^2/ m^2 extraction area and for forest residues 0.10 mJ/ m^2/ m^2 extraction area. A 'best-best-case' scenario (i.e. with high carbon dioxide (CO$_2$)-uptake combined with high 'avoided' CH$_4$ emissions and low methane emissions from the restored wetland) will generate accumulated radiative forcing that decreases and reaches zero after 240 years. A 'worst-worst-case' (i.e. with low CO$_2$-uptake combined with low 'avoided' methane emissions and high CH$_4$ emissions from the restored wetland) will generate radiative forcing higher than coal over the entire time period.

- The accumulated radiative forcing in the 'multiple generations' – scenarios over a 300-year perspective producing 1 MJ/year is estimated to be 0.089 mJ/ m^2 for the scenario 'Peat forestation – bioenergy', 0.097 mJ/ m^2 for the scenario 'Peat wetland with high CO$_2$-uptake' and 0.140 mJ/ m^2 for the scenario 'Peat wetland with low CO$_2$-uptake'. Corresponding values for coal is 0.160 mJ/ m^2, for natural gas 0.083 mJ/ m^2 and for forest residues 0.015 mJ/ m^2. Using a longer time perspective than 300 years will result in lower accumulated radiative forcing from the scenario 'Peat wetland with high CO$_2$-uptake'. This is due to the negative instantaneous forcing that occurs after 200 years for each added generation.

- It is important to consider CH_4 emissions from the virgin mire when choosing mires for utilization. Low original methane emissions give significantly higher total climate impact than high original emissions do.
- Afforestation on areas previously used for peat extraction should be performed in a way that gives a high forest growth rate, both for the extraction area and the surrounding area. A high forest growth rate gives lower climate impact than a low forest growth rate.
- There are great uncertainties related to the data used for emissions and uptake of greenhouse gases in restored wetlands. The mechanisms affecting these emissions and uptake should be studied further.

1. Introduction

The global change impact from using peat as an energy source has been discussed intensely during the last years. Research has also been made on mechanisms (decomposition processes, biomass growth etc.) that govern the magnitude of the climate impact. In December 1998, the Swedish Environmental Protection Agency arranged a hearing where new research findings on greenhouse gas (GHG) emissions connected to such mechanisms were discussed. This study was partly initiated as a follow-up to that hearing.

2. Objective of the Study

The objective of this study is to evaluate the climate impact from the use of peat for energy production in Sweden. Two different options for after-treatment are studied: afforestation and restoration of wetland.

The global change impact from the use of peat for energy production has been studied before (e.g. Rodhe and Svensson 1995; Savolainen et al. 1994; Zetterberg and Klemedtsson 1996; Åstrand et al. 1997). The aims of this study are to:

- update results from previous reports by using the latest research findings available as input for the calculations
- give the results as possible ranges for different scenarios, not as exact figures
- assess which parameters are the most important to consider when attempting to minimize the climate impact from peat utilisation; and
- compare the climate impact from peat utilisation with alternative energy sources

The time span studied here is up to 300 years, which is longer than what has been studied in previous reports. The purpose of this is to make it possible to assess long-term trends (e.g. over several forest generations) as well as impacts over a shorter time period.

3. Methods

In this report we approximate the contribution to climate impact by using the concept of radiative forcing. Radiative forcing, measured in W/ m2 (instantaneous radiative forcing) or J/ m^2 (accumulated radiative forcing), can be described as the change in radiative balance at the tropopause (the boundary between the troposphere and the stratosphere) due to for example emissions of GHGs. In other words, it is the change in the difference between incoming and outgoing radiation through the tropopause. A positive radiative forcing tends to warm the earth's surface, a negative radiative forcing tends to cool it.

The impact chain can be simplified as: emissions lead to increased atmospheric concentrations which lead to radiative forcing, which leads to climate change. Since we only calculate the radiative forcing in this study, one can say that the potential global change impact is calculated. For a full explanation of the concept of radiative forcing and the relationship between GHG concentrations and radiative forcing (Zetterberg 1993, Uppenberg and Åhman 1998).

The United Nations IPCC (Intergovernmental Panel on Climate Change) recommends that the GWP-concept (Global Warming Potential) should be used to calculate and compare GHG emissions on national and international level. With that method global change impact is calculated as the amount of GHG emitted, multiplied by the corresponding GWP-index. The GWP-indeces for different GHGs are defined as the cumulative radiative forcing between the present and some chosen later time horizon, caused by a unit mass of gas emitted now, expressed relative to some reference gas (usually CO_2). The reason that radiative forcing (RF) is used in this study instead of GWP is that:

- RF can describe the impact of an emission scenario that stretches over a long time, which the GWP-concept can't.
- GWP is a relative measure. A GWP today is not the same as a GWP year 2100.
- According to model studies performed by the IPCC, there seems to exist a direct relation between RF and global average temperature.

Three GHGs have been studied: CO_2, CH_4 and Nitrous Oxide (N_2O). The net emissions have been calculated by adding the emissions from different activities or processes connected to the extraction of peat. The unit used is g/(m2*year). All included activities and processes are described in the following sections.

The calculation of radiative forcing was made using a dynamic computer model. The model calculates the radiative forcing from an emission scenario in two steps. First the changes in atmospheric GHG concentrations are calculated, and then the radiative forcing is calculated from these concentrations. The computer model also includes an 'overlap term' that is a result of that CH_4 and N_2O absorbs infrared radiation partly in the same wavelength range. The relations between emissions and changes in concentrations are expressed by exponential functions, and the relations

between changes in concentration and radiative forcing are expressed by functions based on parametrisations of model results.

The calculated radiative forcing from the use of peat is compared to the radiative forcing resulting from the use of coal, natural gas and forest residues. The comparison is based on the assumption that equal amounts of energy are supplied in all the different fuel systems (see section 5.7).

In this study, we present the results both as instantaneous radiative forcing (W/ m^2) and accumulated radiative forcing (J/ m^2). The accumulated radiative forcing describes the time integrated radiative forcing. One can say that the GWP-concept is a simplified way of calculating accumulated radiative forcing.

It is difficult to give the definite answer to whether instantaneous or accumulated radiative forcing is the most correct measure to use when evaluating the global change impact from peat utilisation. However, the inertia of the climate system gives a significant time lag in the cause-effect chain. This means that changes in radiative forcing does not immediately result in changes in global average temperature. The temperature changes more slowly, and as a result of the radiative forcing over a longer period of time. In that sense, the climate system can be described as having a memory.

All this considered, it is our view that one should focus on the accumulated radiative forcing in the evaluation of long term climate impact. But the instantaneous radiative forcing should also be considered since it describes how the climate system is affected in every given moment.

4. System Boundaries and Underlying Assumptions

The scope of this study is peat utilisation under the conditions that are valid in Sweden. The different stages involved in the process of extracting peat for energy are described in Table I below.

The following assumptions for peat characteristics and impact area have been used in the study:
- The area affected by the drainage that is performed, is assumed to be twice the size of the extraction area (Larsson pers. comm.; Åstrand pers. comm.). This assumption is based on the fact that the influence distance of the drainage ditches is approximately 20–30 m outside the extraction area. The drained area that is not used for extraction, the surrounding area, is used for e.g. storage piles and access roads. Based on this assumption, every m^2 of mire that is used for peat extraction will cause 1 m^2 of drained surrounding area in the calculations.
- Average extracted peat depth on the extraction area: 1.4 m (Larsson pers. comm.) The real extracted peat depth vary significantly for specific locations within the extraction area. Extracted peat depth does not include the surface layer of low-humified peat that is removed before extraction (often used as

TABLE I

Description of the different stages of energy peat production in Sweden.

Stage	Year	Description
Virgin mire	before 0	The mire has not been affected by activities connected to peat extraction.
Drained mire, before extraction	0–5	During year 0, the covering vegetation is stripped off and ditches are made on the extraction area, with about 20 meters between each other. The area is then drained to lower the water content from 90–95% to 80–85%. This will normally take 1–5 years. We assume 5 years.
Extraction, transport and combustion of peat	6–25	When the water content has been lowered enough for the ground to carry the machines, extraction of peat can be started. This can be done either as milled peat or as sod peat. The extracted peat is dried lying in the field, and thereafter transported to large storage piles close to the extraction area. The peat extraction is carried out during the summer months. During the winter months, peat is transported directly from the storage piles to plants for heat/power production.
After-treatment	26–	When the peat extraction has been finished after approximately 20 years, the area can be converted to agricultural land (not common in the modern peat industry) or forest, or it can be restored to new wetland.

soil improver or for horticultural use), nor what is oxidized during extraction (see 5.3.1) or what is left after extraction (see 5.5.1). All these parameters included, average peat depth would be approximately 1.9 m.

- Peat density: 1000 kg/ m^3 (Råsjö Torv 2000; HMAB 2000)
- Dry weight: 8% (Råsjö Torv 2000; HMAB 2000)
- Carbon content: 50% on dry solids (Råsjö Torv 2000; HMAB 2000)
- Energy content: 20 MJ/kg dry solids (Råsjö Torv 2000; HMAB 2000)

Three scenarios are studied:

1. *Peat – Forestation – Bioenergy*. After the peat is extracted and used as energy, the area is forested. When the forest is mature, it is harvested and used for energy, paper and wood products. All these products are finally used for energy production. After the first forest generation is harvested the area is re-forested producing a second and third generation of forest products and energy.

2. *Peat – Wetland restoration*. After the peat is extracted and used for energy, the area is restored as a new wetland, with the consequent uptake of carbon from the atmosphere.

3. *Peat multiple generations*. In our scenarios 1 and 2, peat is extracted and used for energy production during a period of 20 years. In our scenario 3, after 20 years, when the first generation of peat is extracted, we add a new generation.

A total of 15 generations are added, with a time delay of 20 years, thus adding up to 300 years of continuous peat extraction supplying 1 MJ/year. We consider two different types of after treatment:

– Forestation, but unlike our scenario 1, we do not consider harvesting the forest or the possible bio-energy production. The reason for this is to investigate the isolated effect of producing 1 MJ/year from peat exclusively.
– Wetland restoration, as in our scenario 2.

5. Greenhouse Gas Fluxes for Peat

The net GHG emissions from peat utilisation are calculated as the difference between emissions from a utilised mire and a virgin mire, i.e. (net emissions) = (emissions from drained mire, before extraction) + (emissions from extraction of peat) + (emissions from combustion of peat) + (emissions from after-treatment) − (emissions from virgin mire), as described in the following sections.

5.1. VIRGIN MIRES

Different virgin peat bogs can have very different characteristics and it is therefore very difficult to generalise for an 'average mire'. The figures used in this study are average values for Sweden, and not connected to a specific mire type.

5.1.1. *Carbon dioxide*

Most virgin mires accumulate C in the growing biomass and thereby act as sinks for atmospheric CO_2. The uptake can vary significantly depending on geographic location (climate) and age of the bog. A Finnish overview of GHG fluxes from peatlands (Crill et al. 2000) uses an average uptake rate of 75 g/ m^2*year to calculate the national C accumulation in undisturbed peatlands. That figure is based on Turunen et al. (1999), in which the range is 62–96 g CO_2/ m^2*year. A similar Swedish overview of GHG flux from peatlands (Kasimir-Klemedtsson et al. 2001) uses uptake rates of 51, 62 and 77 g/ m^2*year for fens, mires and bogs respectively. Those figures are based on Turunen and Tolonen (1996). Previous Swedish studies for peat (Rodhe and Svensson 1995; Zetterberg and Klemedtsson, 1996; Åstrand et al. 1997) have used an uptake rate of 37–48 g/ m^2*year based on Tolonen et al. (1992). However, the range in uptake rates does not affect the results in any significant way why the different levels are not tried here. Since Kasimir-Klemedtsson et al. (2001) presents the latest findings assumed to be representative for Swedish conditions we use their values as input to the model. Based on that, an area-weighted mean value of 58 g/ m^2*year is used as best estimate.

5.1.2. *Methane*

Methane emissions from virgin mires have been studied in several research programmes in different countries. An extensive study concerning CH_4 emissions

from Swedish mires was performed during 1994. Methane emissions were then measured at more than 600 sites all over Sweden. The results from that study are here assumed to be the most representative data available for Swedish conditions.

- The average CH_4 emissions from Swedish mires can vary between 2 – 40 g CH_4/ m^{2*}year depending on mire type and geographic location (Nilsson et al. 2000).

- The magnitude of the methane emissions also depends on if trees are growing on the mire or not (on the surrounding area). In this study we assume a reversed proportional relation between forest fraction and CH_4 emission from surrounding area.

- The long-time average emission for Swedish mires has been modelled in Nilsson et al. (2000), considering data over a 17-year period. The calculated long-time average is 21 g CH_4/ m^{2*}year (Nilsson, pers. comm.) which is used as best estimate here.

5.1.3. *Nitrous oxide*

The emission of N_2O from a virgin mire is assumed to be negligible (Kasimir-Klemedtsson et al. 2001).

5.2. DRAINED MIRE, BEFORE EXTRACTION (YEAR 0 TO 5)

Emissions from working machines connected to drainage of the ground take place during year 0, but they are relatively small and are ignored here.

5.2.1. *Carbon dioxide*

- The drainage causes an oxidation of the peat which results in a net emission of CO_2. Sundh et al. (2000) state that the emissions can be 230–1020 g CO_2/ m^{2*}year, mean value 600 g CO_2/ m^{2*}year, based on field measurements in Sweden. Finnish field measurements have given similar results; 880 g CO_2/ m^{2*}year including the loss of average annual C accumulation occurring in natural mires (Nykänen et al. 1996). Here we assume a net emission of 1000 g CO_2/ m^{2*}year, which can be seen as a worst case. The emission is assumed to increase linearly from 0 to 1000 g CO_2/ m^{2*}year during year 0–3 and remain constant at 1000 g CO_2/ m^{2*}year during year 4–5. We have chosen not to include the range in CO_2-emission from peat oxidation in the sensitivity analysis. This decision is based on the fact that the possibilities to actively limit the oxidation during drainage and extraction are very limited.

- The drainage will also cause an increased growth of trees and other vegetation which result in an uptake of CO_2. However, this uptake is negligible in comparison to the oxidation (Rodhe and Svensson 1995) and is therefore ignored here.

5.2.2. *Methane*

When a mire is drained, the water table is lowered and the decomposition processes in the upper parts of the ground changes from anaerobic to aerobic. Thereby, the CH_4 emissions from the ground are reduced substantially.

- **Extraction area:** The remaining CH_4-emission after drainage is in Sundh et al. (2000) estimated to be 0.4–4.5 g CH_4/ m^2 (approximately 10% of the emissions from virgin mires). These figures include area weighted emissions from drainage ditches, which can be substantially higher than emissions from the extraction area because of vegetation in the ditches. The emissions from ditches can probably be kept low by keeping the ditches clear from vegetation (Sundh et al. 2000). Nykänen et al. (1996) estimate the methane emissions from peat mining areas to be approximately 0.32 g CH_4/ m^2 both from the extraction area and from the ditches. Based on Sundh et al. (2000) we assume the methane emission to be 10% of the original emission. This gives 0.2–4 g CH_4/ m^{2*}year with 2.1 g CH_4/ m^{2*}year as average.
- **Surrounding area:** The remaining CH_4-emission after drainage is assumed to be 25% of the original emission because of more vegetation in the drainage ditches than in the production area (poorer maintenance of the ditches, see discussion above). This gives 0.5–10 g CH_4/ m^{2*}year with 5.25 g CH_4/ m^{2*}year as average.

5.2.3. *Nitrous oxide*

The N_2O-emission is assumed to be 0.02–0.1 g N_2O/ m^{2*}year (Klemedtsson pers. comm.).

5.3. EXTRACTION OF PEAT (YEAR 6–25)

5.3.1. *Carbon dioxide*

- **Extraction area:** The oxidation of peat continues to stay high because of the working of the ground during extraction. This gives a CO_2-emission of 1000 g CO_2/ m^{2*}year (see section 5.2.1) during the extraction period (year 6–25). Note that this oxidised peat is not subtracted from the average extracted peat depth (see section 4.), and thereby not included in the calculated total energy content. Approximately 6% of the extractable peat is lost because of the oxidation (Sundh et al. 2000).
- **Surrounding area:** The oxidation of peat gives an emission of 1000 g CO_2/ m^{2*}year (see section 5.2.1) during year 5–10. After that, the oxidation is assumed to decrease because of less working of the ground compared to the production area. The CO_2-emission is assumed to decrease linearly from 1000 to 300 g CO_2/ m^{2*}year during year 11–25. 300 g CO_2/ m^{2*}year corresponds approximately to oxidation of 3 mm peat per year. There will be a certain spontaneous growth of forest on the surrounding area, which will give an uptake of CO_2. Here we assume that a possible future forestry will be coordin-

ated for the extraction area and the surrounding area, and therefore ignore forest growth at this stage.

- CO_2 is also emitted because of oxidation in stockpiles and other losses (e.g. dusting and self-burning). According to Nykänen et al. (1996) emissions from stockpiles are 175 g CO_2/ m^{2*}year. These emissions are not explicitly included in this study, but can be seen as included since we have chosen a high value for emissions from the extraction area (1000 g CO_2/ m^{2*}year). For a Finnish site Nykänen et al. (1996) estimated total emissions including emissions from stockpiles, ditches and the loss of average annual carbon accumulation in natural mires to 1064 g CO_2/ m^{2*}year. Based on the mean value for emissions from the extraction area in Sundh et al. (2000), total emissions including emissions from stockpiles would be 775 g CO_2/ m^{2*}year.

- Working machines and transports of peat are assumed to give an emission of 1 g CO_2/MJ extracted peat, based on an energy demand of 1.3% of the extracted peat as diesel oil (Larsson pers. comm.).

5.3.2. *Methane*

- **Extraction area:** The remaining CH_4-emission after draining is assumed to be 10% of the original emission (see section 5.2.2). This gives 0.2–4 g CH_4/ m^{2*}year with 2.1 g CH_4/ m^{2*}year as average.

- **Surrounding area:** The remaining CH_4-emission after draining is assumed to be 25% of the original emission because of more vegetation in the drainage ditches than in the production area (poorer maintenance of the ditches). This gives 0.5–10 g CH_4/ m^{2*}year with 5.25 g CH_4/ m^{2*}year as average. This emission is assumed to continue for a few years until tree growth has increased somewhat. The emission is assumed to be 0 g CH_4/ m^{2*}year during year 8–25.

- Emissions from working machines and transports of peat are very small in comparison to emission from ground processes, and are ignored here.

5.3.3. *Nitrous oxide*

- **Extraction area:** During year 6–7, the emission is assumed to be 0.2–1 g N_2O/ m^{2*}year. After that, the emission is assumed to decrease linearly down to 0.01–0.05 g N_2O/ m^{2*}year during year 8–25 (Klemedtsson pers. comm.).

- **Surrounding area:** The emission is assumed to be 0.2–1 g N_2O/ m^{2*}year during year 5–25 (Klemedtsson pers. comm.).

- Emissions from working machines and transports of peat are very small in comparison to emission from ground processes, and are ignored here.

5.4. Combustion of peat (year 6–25)

5.4.1. *Carbon dioxide*
The emission from combustion of peat has been calculated to 91–96 g CO_2/MJ peat depending on moisture content (6–50%) based on heating values and elementary analysis from Swedish mires (Åstrand 2000). These analyses have been performed on peat from the two largest peat operators in Sweden HMAB and Råsjö Torv. The samples from HMAB have been taken from central stocks of mixed peat originating from all of HMAB's mires, thus being representative for the production of HMAB. The samples from Råsjö Torv have been taken from ten different mires at different geographical areas, deliberately chosen to be representative for the production of Råsjö Torv (HMAB 2000; Råsjö Torv 2000). Varying the emission level within the range does not affect the calculated radiative forcing in any significant way, and thus the average value (93.5 g CO_2/MJ) was used in the calculations. The IPCC recommends a higher emission factor for peat, 106 g CO_2/MJ (IPCC 1997). However, in this study we have concluded that the value 93.5 g/MJ is more representative for Swedish conditions, well aware that this value is 12 percent lower than the IPCC value. Given the disparity in these values a new investigation on the CO_2 emission factor is being performed and will be published in 2004 (Nilsson, Zetterberg forthcoming).

5.4.2. *Methane*
The average emission from combustion of peat in Swedish power/heat plants is assumed to be 0.005 g CH_4/MJ peat (Uppenberg et al. 1999).

5.4.3. *Nitrous oxide*
The average emission from combustion of peat in Swedish power/heat plants is assumed to be 0.006 g N_2O/MJ peat (Uppenberg et al. 1999).

5.5. After-treatment – Afforestation (year 26–)

The following assumptions for timber characteristics have been used in the study:
    Basic wood density (dry matter) = 430 kg/ m3 (Norway spruce (*Picea*)/birch (*Betula*); 9/1) Carbon content = 50%

5.5.1. *Carbon dioxide*
- **Oxidation on extraction area:** The oxidation of peat will continue until the remaining peat layer has disappeared. Assuming an average peat depth of 0.2 m after the extraction has finished, it will take approximately 22 years (until year 47) until the remaining peat has disappeared if the oxidation rate is 1000 g CO_2/ m^2*year. That calculation is based on the assumption that approximately 50% of the remaining peat is easily oxidable material (consisting mainly of hemicelluloses or cellulose). The other 50% is assumed to be made up of resistant, highly humified material which decompose very slowly (this fraction will eventually decompose, but over a much longer time than this study covers). From year 48, the emission caused by oxidation is assumed to

be 0. The fraction of easily oxidable peat in the deeper part of the bog can be lower than what is assumed here (30% is stated by Åstrand (pers. comm.)).

- *Oxidation on surrounding area:* The oxidation of peat will continue as long as there is peat left and the water table is kept low through drainage. Here we assume that the oxidation will continue at the same rate as in the preceding stage, 300 g CO_2/ m^2*year, throughout the study period.

- *Accumulation of C in humus:* Accumulation of carbon in humus: Following afforestation, a new forest floor (litter+ humus layer) will accumulate on the soil as a result of litter formation. The accumulation of C in the upper part of the soil is a balance between litter input and decomposition, and assuming a constant C input the forest floor will reach an approximate steady-state after 50–100 years (Lillesköld and Nilsson 1997). Thus, as a rough estimate we assume that the C accumulation will occur linearly during the first forest rotation and thereafter turns to zero. In a cold climate with low decomposition the steady-state will be reached later, perhaps after several hundred years, but, on the other hand, the absolute accumulation rate will be lower here.

  Olsson et al. (1996) estimated the C pools of the humus layer of four clear-felled coniferous forests in Sweden to 1.7–4.4 kg C/ m^2. At the two sites in southern Sweden the topsoil contained 2.9 and 4.4 kg C/ m^2. This could be compared with data from Gärdenäs (1998), who reported a mean value for the organic matter store in the forest floor of Norway spruce sites in northern Europe to 40 mg/ha, i.e. about 2 kg C/ m^2. Based on the presented data we can assume that, as a maximum, afforested peatlands in southern Sweden could accumulate 3–4 kg C/ m^2 during a forest generation. Here we use a value of 3.5 kg C/ m^2, which over a 70-year period gives a mean uptake of 183 g CO_2/ m^2*year. This is about half of the uptake rate used in the study by Zetterberg and Klemedtsson (1996). In a low-productive site (3.0 m^3/ha year) the wood production and litter fall would be about one third of that found on the high-productive. However, also the decomposition rate would be somewhat lower and we assume a C uptake of 100 g/ m^2year for the low-productive site.

- *Forest growth:* The range for C uptake in the forest planted on the extraction area as well as on the surrounding area has been calculated from Hånell (1997) According to Hånell the regional mean forest growth can reach between 3 m^3/ ha*year (northern Sweden, Härjedalen and Västerbotten) and 8.5 m^3/ ha*year (southern Sweden, Småland). For parts of Småland, the growth rate can be up to 10 m^3/ha*year (Anon 1992). These figures imply a forest management withoutN fertilization but where the nutrient status of the soil is improved by wood-ash or other mineral fertilizer. The lower value would underestimate the expected growth rate in areas of northern Sweden where peat is extracted (Hånell (pers. comm.) and Åstrand (pers. comm.)) but still 3 m^3/ha*year was used as a worst-case scenario.

  The C net uptake in forest biomass for two sites, one with low and one with high growth rate, was calculated from data on volume production in Hånell

(1997). The high-productive site (8.5 m^3/ha*year) was thus assumed to have a total stem volume production of 595 m^3/ha, harvested in thinnings and final cutting, over a forest rotation of 70 years, while the corresponding figure for the low-productive site (3.0 m^3/ha* year) was assumed to be 350 m^3/ha over a 100 year rotation. Furthermore, the total standing biomass at thinnings and final cutting, including stem, branches, needles, stump and roots, was assumed to amount to 1.5 times the stem biomass. The same assumption was also used by Zetterberg and Klemedtsson (1996) and Åstrand et al. (1997). Lundmark (1988) stated that the total biomass/stem biomass ratio of mature conifer trees is 1.5–1.7, but since younger trees harvested in thinnings has lower ratios the lower part of the range was used here. Using the above-mentioned figures and assumptions, an accumulation of 414–1006 g CO_2/ m^{2}year is estimated. If the growth rate 10 m^3/ha*year is used as the higher value, an accumulation of 1180 g CO_2/ m^{2}year is estimated following the calculations above. In the simulations we use the range 414–1180 g CO_2/ m^{2}year for carbon accumulation in forest growing on cut-away peatlands.

Note that the average growth rates for northern and southern Sweden are used in this study as examples of low and high growth rates. It is possible to get a high growth rate also in northern Sweden depending on site specific geographic location and management conditions, and vice versa. The different growth rate scenarios are therefore named 'low' and 'high' in the model input and results sections, and should not be seen as specific for afforestation on utilised peat land in different parts of Sweden.

- *Forest rotation period:* The generation period for a forest with 'low' growth rate is assumed to be 100 years and for a 'high' growth rate, 70 years. The generation period for 'best estimate' in the simulations is assumed to be 85 years.

- *After felling:* All harvested forest biomass (20% of the standing biomass is assumed to be roots and stumps) is assumed to be combusted immediately after felling which results in an instantaneous emission of 33120 g CO_2/ m^2 (low) to 66080 g CO_2/ m^2 (high). This assumption is based on Eriksson (1994) who state that approximately 45% of the harvested biomass goes directly to energy production, 35% or more of the remaining harvested biomass goes to pulp and paper production and the rest (20%) end up as sawmill products. The latter is partly used in buildings and other long-lived objects, and the rest is used as consumption material in e.g. construction work. Based on these assumptions, one can also argue that the CO_2-emissions should be distributed out over a time period of at least 10–15 years, with the major part of the emissions during the first 5 years. On the other hand, Eriksson (pers. comm.) states based on data from Statistics Sweden (SCB), that there is no significant change of the C pool (wooden buildings etc.) in the Swedish society. That assumption would also lead to an instantaneous emission of the forest biomass C in the calculations. However, the different scenarios do not change the results

in any significant way, so these differences are not investigated further. The roots and stumps are assumed to decompose at a constant rate during 20–100 years, thereby emitting CO_2 during 20–100 years after felling.

In this study, it is assumed that all the harvested forest biomass including branches and needles is used for energy production in some way. This assumption is valid if the forest is explicitly grown for energy production. However, it is not possible to restrict land owners to a specific after-treatment alternative for hundreds of years. In an average scenario, a large part of the forest biomass will probably go directly to energy production in both heat/power plants and industry, and some of it will first be transformed to paper and building material, and then gradually be used for energy production. But there will always be losses of biomass during processing of wood products, and some of the wood waste will end up in landfills and thereby decompose without being used for energy production. These losses are uncertain and highly influenced by legislation concerning waste handling. We have not tried to quantify these losses in this study, and thus the climate impact from the peat-afforestation scenario probably are underestimated for average forestry, compared to the other studied energy sources.

5.5.2. *Methane*
There will be a small uptake of CH_4 in forest soils during forest growth, but that uptake is ignored here.

5.5.3. *Nitrous oxide*
- **Extraction area:** The emission is assumed to decrease linearly from 0.1–0.5 g N_2O/ m^2*year to 0.02–0.1 g N_2O/ m^2*year during year 26–47 and thereafter stays constant (Klemedtsson pers. comm.).
- **Surrounding area:** The emission is assumed to stay constant at 0.14–0.7 g N_2O/ m^2*year from year 26 and throughout the study period (Klemedtsson pers. comm.).

5.6. AFTER-TREATMENT – RESTORATION OF WETLAND (YEAR 26–)

5.6.1. *Carbon dioxide*
There are very few studies on C uptake in a restored wetland and the data on uptake rates are uncertain. Tuittila et al. (1999) state that a restored wetland after only a few years will have a net accumulation of C and that the accumulation rate can be 108–160 g C/ m^2*year. In Crill et al. (2000) these figures are used to calculate the total C sink capacity in restored Finnish cut-away peatlands. Savolainen et al. (1994) use an uptake rate of 64 g C/ m^2*year for a restored peatland at the beginning of paludification, and 37 g C/ m^2*year for lake formation, both values from Hillebrand and Wihersaari (1993). In Kasimir-Klemedtsson et al. (2001) it is shown that the C accumulation rate in a mire decreases significantly as the mire gets older. An example for a 2400 year old mire show that the accumulation rate during the first 400 years can reach between 60 and 120 g C/ m^2*year. Tolonen and

Turunen (1996) show mean accumulation rates from 77 to 126 g C/ m^2*year for peat layers 9–102 years old (maximum 290 g C/ m^2*year), and rates from 40 to 81 g C/ m^2*year for peat layers 100–200 years old (Alm et al. 1992).

Since the uncertainties connected to the C uptake rate and how it will develop over time are very large, wide ranges has been studied –37 g C/ m^2*year as low value and 160 g C/ m^2*year as high value, corresponding to 136–587 g CO_2/ m^2*year. The C uptake rate is probably high in a young mire during the first decades or centuries whereafter it decreases gradually (probably exponentially) as the mire gets older (during several thousand years). The change in uptake rate is highly governed by climate and the data available are uncertain (Bohlin pers. comm.; Tolonen and Turunen 1996) since the process stretches over such long time periods. In this study we have not tried to quantify the temporal change in uptake rate. We assume that the uptake increase linearly from 0 to 136–587 g CO_2/ m^2*year during year 26–31 and thereafter stays constant at that level throughout the study period.

5.6.2. *Methane*

The uncertainties on CH_4 emissions from a restored wetland are also very large. Tuittila et al. (2000) state that the CH_4 emissions can stay at a low level for a long period of time, even after sites have become fully vegetated and colonized by mire plants. We assume that the CH_4-emissions will increase linearly from 0 up to the level assumed for virgin mires, 2–40 g CH_4/ m^2*year, during year 26–45, and thereafter stay constant at that level.

5.6.3. *Nitrous oxide*

No information on N_2O-emissions from restored wetlands were found.

5.7. ASSUMPTIONS FOR THE COMPARISON TO COAL, NATURAL GAS AND
        FOREST RESIDUES

The energy content of the peat layer under 1 m^2 of the extraction area is 2240 MJ based on the assumptions in section 4. This peat is extracted during 20 years, which gives 112 MJ/year. In the comparison of global change impact from the different fuels, the same amount of energy as coal, natural gas and forest residues is assumed to be combusted during the same 20 years.

All fuel systems are assumed to supply equal amounts of energy, also for the after-treatment stages. For the system with restoration of wetland as after-treatment this will have no effect on the calculations since no energy is produced in the peatland – wetland scenario after peat extraction has stopped. On the other hand, for the system based on afforestation as after-treatment this will have effect on the calculations since all the forest biomass that is produced on the utilised area is assumed to be used for energy production (see 5.5.1). In this peatland – forestry scenario forest biomass will in fact produce more than half of the total energy during 300 years. In the comparison equal amounts of energy are assumed to be produced by the other energy systems. The relevant amounts of energy produced from utilisation of 1 m^2 of mire are displayed in Table II below.

The differences in efficiency between different fuels are small and vary with plant type. The plant efficiency is therefore not considered here.

TABLE II

Energy produced from utilisation of 1 m^2 mire for peat extraction during different stages in the process.

Forest growth rate	Low		Best est.		High	
	MJ/year	*year*	MJ/year	*Year*	MJ/year	*Year*
Peat extraction	112	*6–25*	112	*6–25*	112	*6–25*
Forestation 1st generation	662	*126*	992	*111*	1322	*96*
Forestation 2nd generation	662	*226*	992	*196*	1322	*166*
Forestation 3rd generation	–	–	992	*281*	1322	*236*

5.8. EMISSIONS SUMMARY – MODEL INPUT

5.8.1. *Ground preparation and extraction of peat*

TABLE III.

Emission ranges for CO_2, CH_4 and N_2O from virgin mires and from ground processes and activities during preparations and extraction of peat. 'b.e.' means best estimate.

Stage/ activity	Year	CO_2 (g/m^2*year)*		CH_4(g/m^2*year)*		N_2O (g/m^2*year)*	
		Extraction area	Surrounding area	Extraction area	Surrounding area	Extraction area	Surrounding area
Virgin mire	before 0	–58	–58	2–40 b.e.=21	2–40 b.e.=21	0	0
Drained mire, before extraction	0–5	*year 0–3:* linearly increasing from 0 to 1000 *year 4–5:* 1000	*year 0–3:* linearly increasing from 0 to 1000 *year 4–5:* 1000	0.2–4 b.e.=2.1	0.5–10 b.e.=5.25	0.02–0.1 b.e.=0.06	0.02–0.1 b.e.=0.06
Extraction of peat	6–25	1000	*year 6–10:* 1000 *year 11–25:* linearly decreasing from 1000 to 300	0.2–4 b.e.=2.1	*year 6–7:* 0.5–10 b.e.=5.25 *year 8–25: 0*	*year 6–7:* 0.2–1 b.e.=0.6 *year 8–25:* linearly decreasing to 0.01–0.05 b.e.=0.03	0.2–1 b.e.=0.6
Working machines and transports	6–25	1 g/MJ peat		0		0	
Combustion of peat	6–25	93.5 g/MJ peat		0.005 g/MJ peat		0.006 g/MJ peat	

* The units for emissions from working machines, transports and combustion of peat are defined in the table.

5.8.2. *After-treatment – Afforestation*

TABLE IV.

Emission ranges for CO_2, CH_4 and N_2O from ground processes and activities during after-treatment by afforestation. 'b.e.' means best estimate.

Process/ activity	Year	CO_2 (g/m^2*year)*		CH_4 (g/m^2*year)*		N_2O (g/m^2*year)*	
		Extraction area	Surrounding area	Extraction area	Surrounding area	Extraction area	Surrounding area
Oxidation of remaining peat and other ground processes	26–300	*year 26–47:* 1000 *year 48–300:* 0	300	0	0	*year 26–47:* linearly decreasing from 0.1–0.5 b.e.=0.3 to 0.02–0.1 b.e.=0.06 *year 48–300:* 0.02–0.1 b.e.=0.06	0.14–0.7 b.e.=0.42
Accumulation of carbon in humus	26–300	*year 26–95:* −100 − −183 *year 96–300:* 0	*year 26–95:* −100 − −183 *year 96–300:* 0	x	x	x	x
Forest growth, emissions due to self-thinning are included	*low growth rate:* 26–125, 126–225, 226–300 *high growth rate:* 26–95, 96–165, 166–235 236–300	−414 − −1180 b.e.= −797	−414 − −1180 b.e.= −797	x	x	x	x
Combustion of forest biomass after felling	*low growth rate:* 126, 226 *high growth rate:* 96, 166, 236	33120– 66080 b.e.=49600	33120– 66080 b.e.=49600	x	x	x	x
Oxidation of roots and stumps	*low growth rate:* 126–145, 226–245 *high growth rate:* 96–115, 166–185 236–255	414–826 b.e.=620	414–826 b.e.=620	x	x	x	x

x = not relevant.

5.8.3. *After-treatment – Restoration of wetland*

TABLE V

Emission ranges for CO_2, CH_4 and N_2O from ground processes during after-treatment by restoration of wetland. 'b.e.' means best estimate.

Stage	Year	CO_2 (g/m^2*year)*		CH_4 (g/m^2*year)*		N_2O (g/m^2*year)*	
		Extraction area	Surrounding area	Extraction area	Surrounding area	Extraction area	Surrounding area
Restored wetland	26–300	*year 26–31:* linearly increasing from 0 to −136 – −590 *year 32–300:* −136 – −590	*year 26–31:* linearly increasing from 0 to −136 – −590 *year 32–300:* −136 – −590	*year 26–45:* linearly increasing from 0 to 2–40 b.e.= 21 *year 46–300:* 2–40 b.e.= 21	*year 26–45:* linearly increasing from 0 to 2–40 b.e.= 21 *year 46–300:* 2–40 b.e.= 21	no data available	no data available

6. Greenhouse Gas Fluxes for Coal, Natural Gas and Forest Residues

6.1. COAL

The direct (combustion) and indirect (production and transports) emissions from the coal fuel cycle are summarised in Table VI.

TABLE VI

Summary of greenhouse gas emission factors per MJ fuel for different stages in the coal fuel cycle (Uppenberg and Zetterberg 1999).

Greenhouse gas	Indirect emissions (g/MJ)	Direct emissions (g/MJ)	Total emissions (g/MJ)
CO_2	3.2	91	94.2
N_2O	–	0.012	0.012
CH_4	1.1	0.0005	1.1

Indirect emissions from coal mainly originates from extraction and transportation. In this study the indirect emissions are based on current conditions. However, in 300 years from now extraction sites and costs may be quite different from now

and consequently, the indirect emissions may be different. If coal resources become scarcer, more fragmented and more unavailable geographically, this is likely to lead to increased emissions from extraction and transportation. We have not attempted to quantify these effects for coal, nor for natural gas.

6.2. NATURAL GAS

The direct (combustion) and indirect (production and transports) emissions from the natural gas fuel cycle are summarised in Table VII.

TABLE VII

Summary of GHG gas emission factors per MJ fuel for different stages in the natural gas life cycle (Uppenberg and Zetterberg 1999). (b.e.= 'best estimate')

Greenhouse gas	Indirect emissions (g/MJ)	Direct emissions (g/MJ)	Total emissions (g/MJ)
CO_2	4.9	56	61
N_2O	0.0001	0.0005	0.0006
CH_4	0.003–0.41 (0.04 b.e.)	0.0001	0.003–0.41 (0.04 b.e.)

6.3. FOREST RESIDUES

According to Zetterberg and Hansén (1998) the gross emissions from collecting and combusting forest residues are approximately 103 g CO_2/MJ; 0.0056 g CH_4/MJ and 0.005 g N_2O/MJ whereof 3 g CO_2/MJ come from the use of fossil fuels in forest machines. To calculate the net emissions of CO_2 one must also consider the decomposing of the forest residues that would have occured if they had been left on the ground. In Eriksson and Hallsby (1992) it is assumed as a rough approximation to estimate the climate impact of forest residues, that the forest residues are decomposed totally after 20 years and that the decomposition process is linear. Hyvönen et al. (2000) present a model for the decomposition based on empirical studies. The model describes an exponential decomposition process. This will result in a fast decomposition rate in the beginning but a slower rate after some years, and the process will continue for a long time, approaching zero. The net emissions of CO_2 over time for the use of forest residues for energy, based on both Eriksson and Hyvönen et al. are illustrated in Figure 1. The model from Hyvönen et al. used here gives a halving of the initial C content in the forest residues after nine years. The model has been modified to give complete decomposition of the forest residues (no remaining C) after approximately 100 years. Other models suggest that the decomposition of organic matter in forest is not complete. Berg and Ekbohm (1993) e.g. developed a model based on asymptotic functions, indicating that 10% of Scots
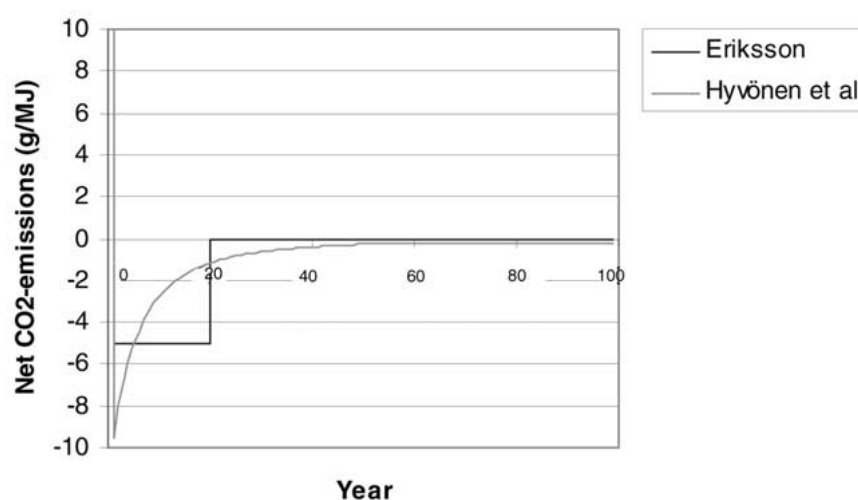
Figure 1. Net emissions from the use of forest residues for fuel. The initial value (year0) for both scenarios is 103 g/MJ. The range for the y-axis is decreased here to make it possible to see differences between the scenarios.

pine (*Pinus*) needle litter is resistant to further decomposition. If this is true also for forest residues, the global change impact of using forest biomass as fuel would be somewhat higher than calculated in this study.

The differences in global change impact from the use of forest residues for energy due to the different decomposition scenarios are displayed in Figures 3 and 4 in section 7.1. As shown in these figures, the differences in global change impact due to different decomposition scenarios are marginal compared to the global change impact of the other fuels. Based on that result and with consideration to technical difficulties in the model implementation of Hyvönen et al. (2000), only the decomposition scenario according to Eriksson and Hallsby (1992) is used for comparison in the other result figures.

6.4. Discussion on oil

Oil is not included in the comparison of climate impact from the different fuels. The reason for this is that the fuels for the comparison were chosen to represent max and min values for net greenhouse gas emissions. Coal has the highest specific emissions of carbon dioxide while forest residues has the lowest. Above that, natural gas was included to represent GHG emissions in between the max and min values.

Fuel oil has a specific CO_2-emission of 76 g/MJ which puts it in between natural gas and coal (56 g CO_2/MJ and 91 g CO_2/MJ respectively) considering global change impact. This will result in a global change impact that will also lie in between natural gas and coal. So, to compare the results to oil, imagine a line in
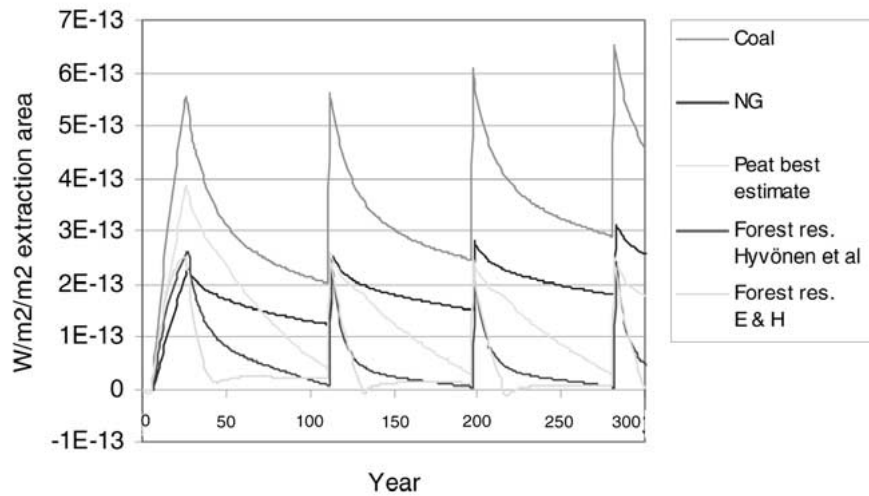
Figure 3. Immediate RF from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages of the peatland – forestation – bioenergy scenario. Best estimate for forest growth is 6.5 m^3/ha*year.

the diagrams between the lines for natural gas and coal, somewhat closer to coal than natural gas.

7. Results

7.1. PEATLAND – FORESTATION – BIOENERGY SCENARIO

A comparison between best estimate for the peatland – forestation – bioenergy scenario and the other different fuel systems are presented in Figures 3 and 4 as instantaneous and accumulated RF. Equal amounts of energy are produced and the timing of energy use is the same in the different fuel systems (see Table II).

The immediate RF from peat lies between the RF from natural gas and coal during the extraction of peat, but the fact that forest is grown on the peat area after extraction makes the RF from peat decrease faster than from natural gas. The instantaneous RF from peat is equal to natural gas after approximately 70 years and approaching the RF from forest residues at the end of the first forest generation. In year 111, the forest is cut down and combusted making the RF increase instantaneously. The accumulated RF from peat is approximately equal to natural gas after 180 years and approximately 20% lower than natural gas after 300 years. Figure 4 shows that the there is a marginal difference in global change impact between the two emission scenarios for forest residues.

The radiative forcing from CO_2, CH_4 and N_2O respectively is shown in Figure 5.
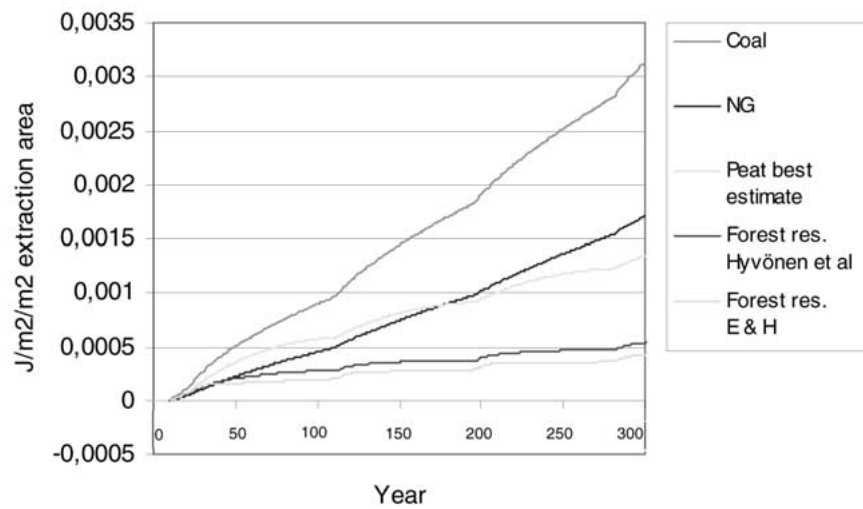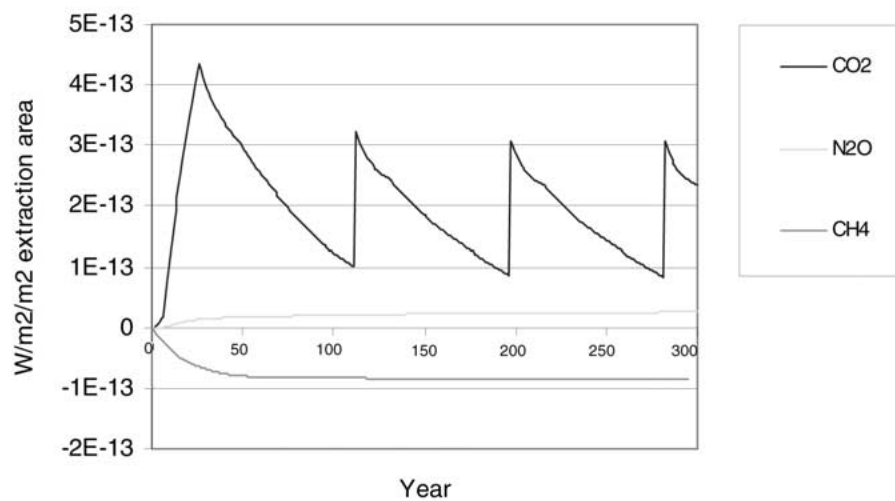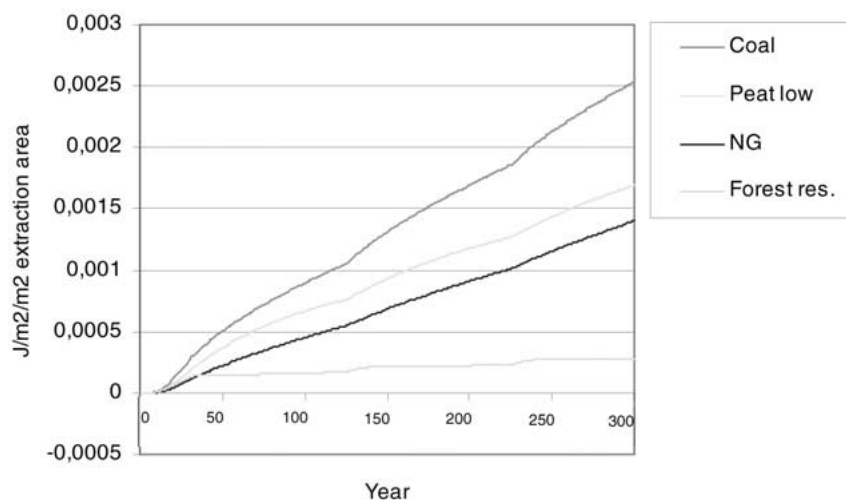
Figure 4. Accumulated radiative forcing from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages of the peatland – forestation – bioenergy scenario. Best estimate for forest growth is 6.5 m^3/ha*year.

Figure 5. Immediate emissions of CO_2, CH_4 and N_2O from the use of 1 m^2 mire in the peatland – forestation – bioenergy scenario with forest growth rate 6.5 m^3/ha*year.

Figure 6. Accumulated RF from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages. The forest growth rate is assumed to be low, corresponding to 3 m^3/ha*year.

The negative emissions and radiative forcing from methane is due to the fact that the mire has been drained which has substantially reduced the methane emissions.

7.1.1. *The importance of forest growth rate*

The differences in accumulated radiative forcing from the peatland – forestation – bioenergy scenario depending on different CO_2-uptake rates in the forest that is planted after extraction, is shown in Figures 6 and 7.

If the forest that is planted after peat extraction accumulates C at a low rate (3 m^3/ha*year), the long term accumulated radiative forcing from the use of peat lies in between coal and natural gas, closer to natural gas. If on the other hand the forest is assumed to accumulate C at a high rate (10 m^3/ha*year), the accumulated radiative forcing from peat is equal to natural gas after 120 years and approximately 1/3 lower than natural gas after 300 years.

These examples are based on the assumption that forest is planted and grows at the same rate on both the extraction area and the surrounding area. Different growth rates for the extraction and surrounding area will result in a climate impact in between best and worst case. Assuming for example that the forest on the surrounding area will have a low growth rate while the extraction area will have a high growth rate (depending on differences in ground properties) makes the climate impact similar to best estimate.
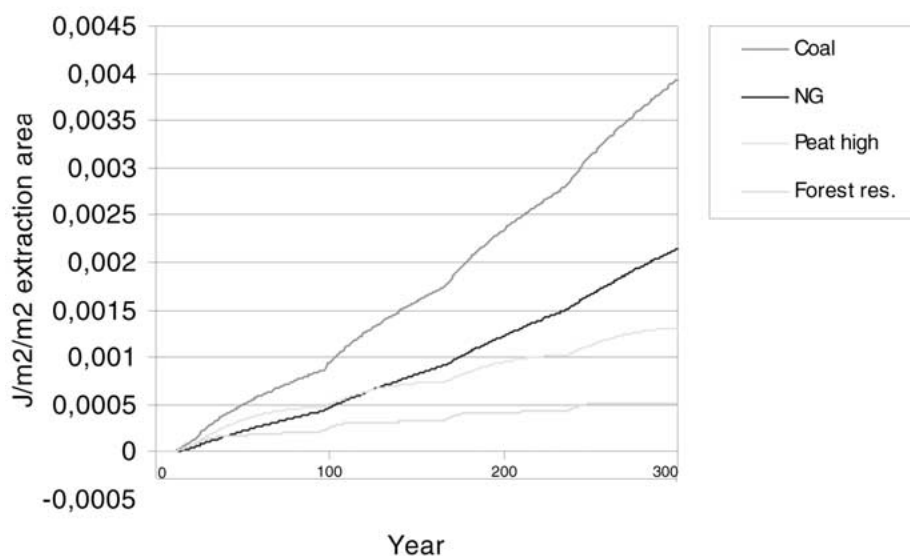
Figure 7. Accumulated radiative forcing from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages. The forest growth rate is assumed be high, corresponding to 10 m^3/ha*year.

7.1.2. *The importance of methane emissions from virgin mires*

The accumulated radiative forcing assuming different emission rates of CH_4 from virgin mires is displayed in Figure 8. CH_4 emissions from a virgin mire is dramatically reduced. Therefore, using a mire for peat extraction means avoiding a certain amount of CH_4 emissions. The differences in long term accumulated radiative forcing between 'best case' and 'worst case' depending on CH4 emissions from virgin mires are approximately ± 50% compared to best estimate. Very high methane emissions from virgin mires (best case) makes the accumulated radiative forcing from peat lie between forest residues and natural gas (closer to forest residues), while very low methane emissions (worst case) makes the long term accumulated radiative forcing from peat lie between natural gas and coal (closer to natural gas).

7.1.3. *The importance of nitrous oxide emissions*

The accumulated radiative forcing assuming different emission rates of N_2O from ground processes during the entire study period is displayed in Figure 9. The differences in long term accumulated radiative forcing between 'best case' and 'worst case' depending on N_2O emissions from ground processes during extraction and afforestation are approximately ± 10% compared to best estimate.
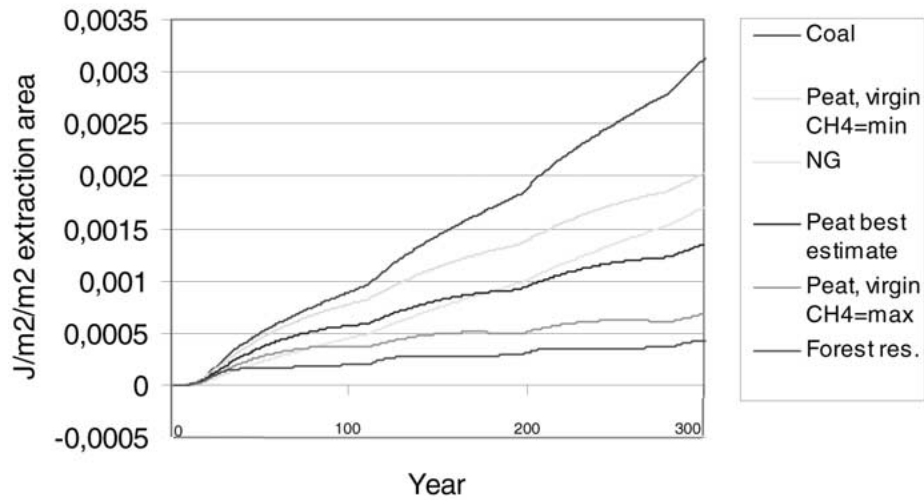
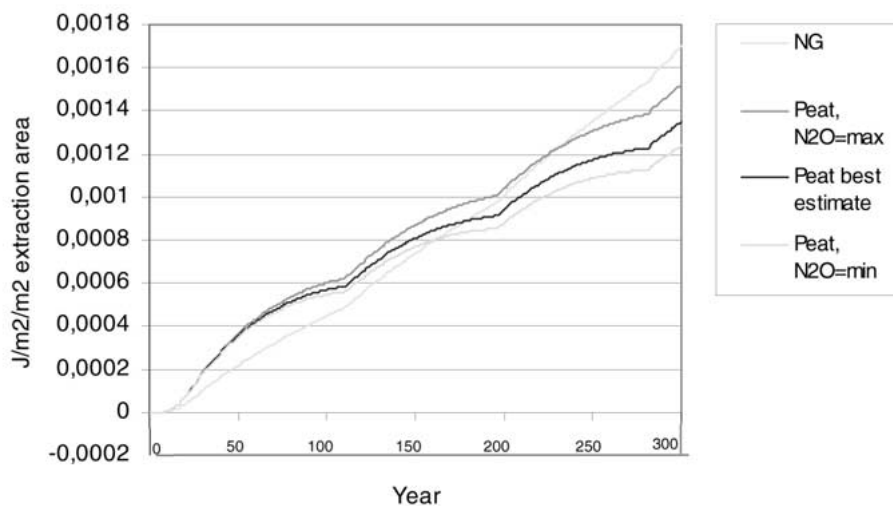Figure 8. Accumulated RF for different emission rates of CH_4 from virgin mires.

Figure 9. Accumulated RF for different emission rates of N_2O from ground processes.

7.1.4. Summary of the results

The simulated climate impact of peat utilisation with afforestation as after-treatment, depends significantly on the assumed growth rate of the trees and the assumed original CH_4 emissions from the virgin mire (emissions that are avoided when the mire is drained).

A 'best-best-case' scenario (i.e. with high growth rate combined with high (avoided) CH_4 emissions) will generate accumulated radiative forcing comparable

to using forest residues for energy production. The 'best-best-case' scenario is constructed by combining *Peat high* in Figure 7 with *Peat, virgin CH$_4$ = max* in Figure 8, and is not displayed explicitly in a figure.

A 'worst-worst-case' scenario, with low growth rate and low (avoided) CH$_4$ emissions, will generate radiative forcing somewhere in between natural gas and coal (closer to coal). The 'worst-worst-case' scenario is constructed by combining *Peat low* in Figure 6 with *Peat, virgin CH$_4$ = min* in Figure 8, and is not displayed explicitly in a figure.

The best-case figures for CH$_4$ emissions from virgin mires are very high and not representative as average values. For forest growth rate, however, the best-case figures are not at all unlikely to achieve according to Hånell (pers. comm.), on the condition that wood ashes are returned to the forest ground as fertilisation (no nitrogen fertilisation is assumed). The growth rate can, according to Hånell, under favourable growth conditions even be higher than the best-case scenario in this study.

The 'best-best-case' and 'worst-worst-case' scenarios described above are to be considered as extreme values for the climate impact of peat utilisation. It is unlikely that those conditions will occur very often. The 'normal' climate impact of peat utilisation will lie somewhere in between the extreme values.

7.2. PEATLAND – WETLAND SCENARIO

A comparison of RF from the peatland – wetland scenario and the other different fuel systems are presented in Figures 10 and 11 as immediate and accumulated RF respectively. For the peatland – wetland scenario, there is no 'best-estimate' scenario displayed since there are great uncertainties connected to C uptake in restored wetlands. Equal amounts of energy are produced in all the different systems, i.e. 112 MJ/year during years 6–25.

Looking at immediate RF (figure 10), the low value for C uptake rate in the wetland makes the potential global change impact from peat similar to coal over the entire study period. The high value for C uptake rate makes peat comparable to natural gas after approximately 120 years and the RF reaches zero after 200 years and continues to decrease below zero thereafter. With the high C uptake rate, all the CO$_2$ emitted during peat extraction will have been accumulated again after approximately 300 years. The C uptake combined with avoided CH$_4$ emissions from the original virgin mire results in negative instantaneous radiative forcing after 200 years.

The results for accumulated RF (Figure 11) also show that the potential climate impact from peat is comparable to coal if the low C uptake rate for the restored wetland is used. With the high C uptake rate however, accumulated radiative forcing from peat is comparable to natural gas after approximately 225 years and decreases below natural gas at the end of the study period.
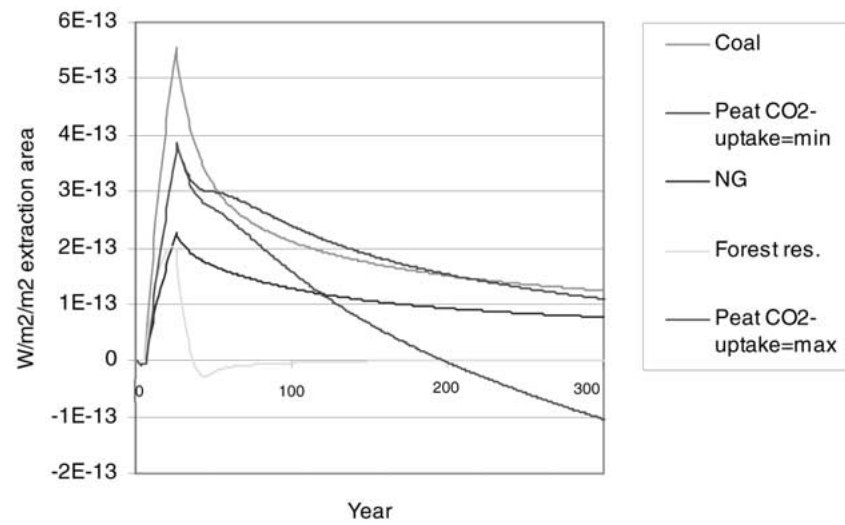
Figure 10. Immediate RF from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages.
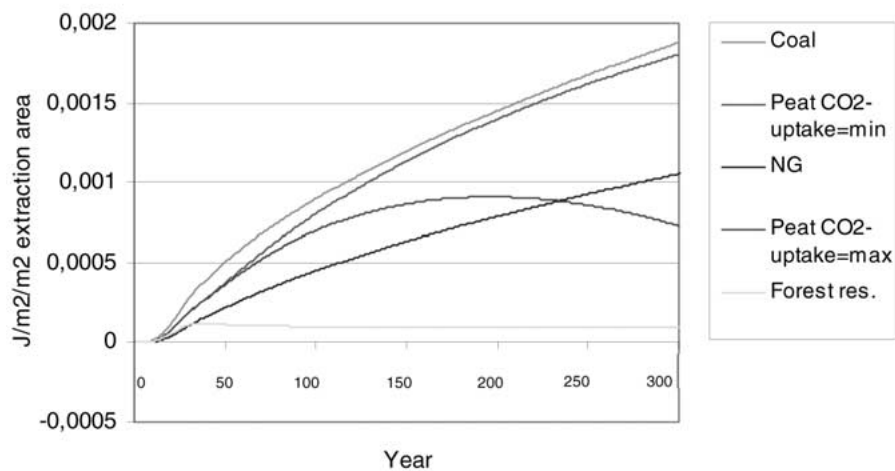

Figure 11. Accumulated RF from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages.
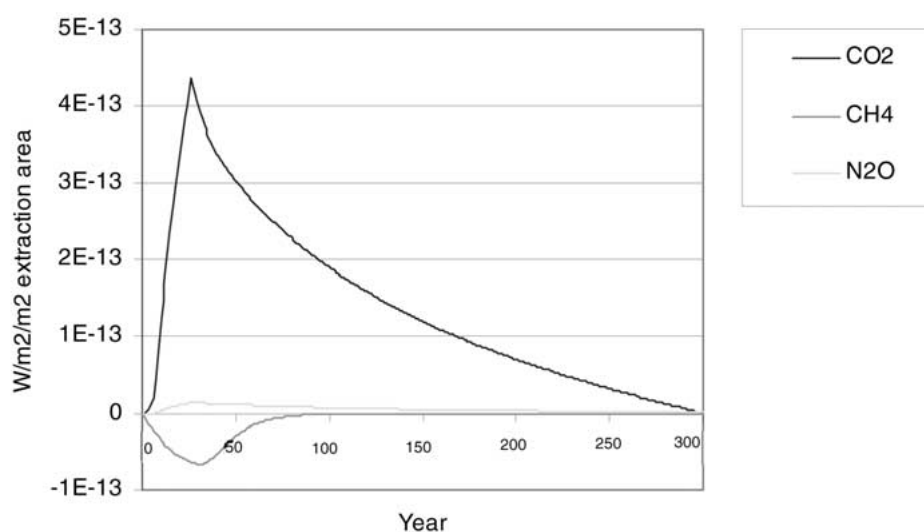
Figure 12. Immediate RF from the emissions of CO_2, CH_4 and N_2O connected to the use of 1 m^2 mire for peat production. Carbon accumulation in the restored wetland assumed to be a mean value of high and low carbon uptake rate.

The differences in long term accumulated RF between 'best case' and 'worst case', depending on the magnitude of C uptake in the restored wetland, are approximately $\pm$ 50% compared to the mean value.

The negative emissions and RF from CH_4 during the first 25 years depend on the fact that CH_4 emissions from the virgin mire have been substantially reduced because of the drainage of the mire. The restoration of wetland is assumed to cause increasing CH_4 emissions again. After 50 years, the net CH_4 emissions from the restored wetland compared to virgin mires are assumed to be zero.

7.2.1. *The importance of methane emissions from restored wetland*

The differences in immediate and accumulated RF from the peat system depending on different CH_4-emission rates from the restored wetland is shown in Figure 13.

The differences in long term accumulated RF between 'best case' and 'worst case' depending on the magnitude of CH_4 emissions from the restored wetland are approximately $\pm$ 50% compared to best estimate (mean value for carbon uptake rate and 'best estimate' for CH_4 emissions). A very low CH_4-emission rate makes the accumulated RF from peat equal to natural gas at year 200 and about 2/3 of natural gas after 300 years. A very high CH_4-emission rate makes the accumulated RF from peat lie close to coal.
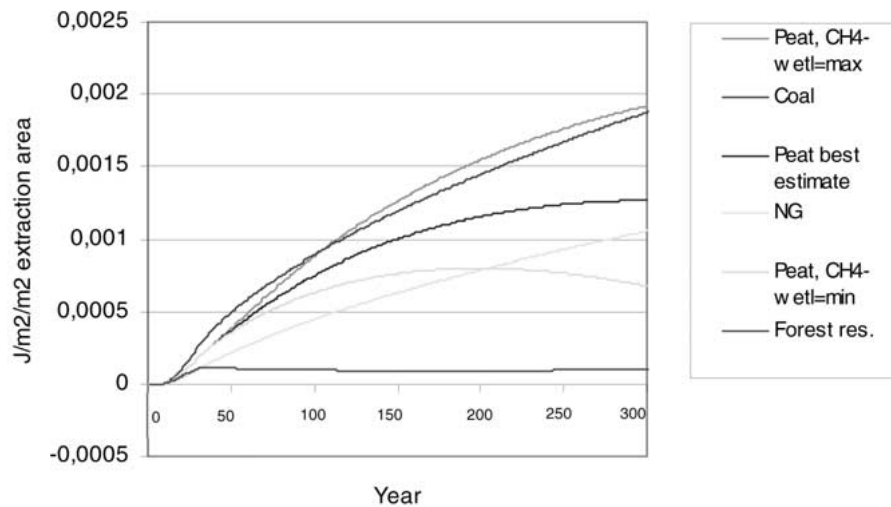
Figure 13. Accumulated RF for different emission rates of CH$_4$ from the restored wetland.

7.2.2. *The importance of methane emissions from virgin mires*

The immediate and accumulated RF assuming a mean value for C uptake rate and different emission rates of CH$_4$ from virgin mires are displayed in Figure 14. When a mire is drained, the CH$_4$ emissions from a virgin mire is dramatically reduced. Therefore, using a mire for peat extraction means avoiding a certain amount of CH$_4$ emissions. The differences in long term accumulated RF between 'best case' and 'worst case' depending on the magnitude of CH$_4$ emissions from virgin mires are approximately $\pm$ 50% compared to best estimate. A very high CH$_4$-emission rate makes the accumulated RF from peat equal to natural gas at year 170 and less than 2/3 of natural gas after 300 years. A very low CH$_4$-emission rate makes the accumulated RF from peat lie close to coal.

7.2.3. *Summary of the results*

The simulated global change impact of peat utilisation with restoration of wetland as after-treatment, is significantly dependent on the CO$_2$-uptake rate in the wetland as well as (avoided) CH$_4$ emissions from the virgin mire and CH$_4$ emissions from the restored wetland.

   If we assume that a high CO$_2$-uptake rate is combined with high (avoided) CH$_4$ emissions from the virgin mire and low CH$_4$ emissions from the restored wetland (a really 'best-best-case'), this will generate an accumulated radiative forcing comparable to natural gas in a 100-year perspective. After that the accumulated forcing will start to decrease, reaching zero after approximately 240 years. The 'best-best-case' scenario is constructed by combining *Peat CO$_2$-uptake = max* in Figure 11 with *Peat, CH$_4$-wetl = min* in Figure 13 and *Peat, virgin CH$_4$=max* in Figure 14,
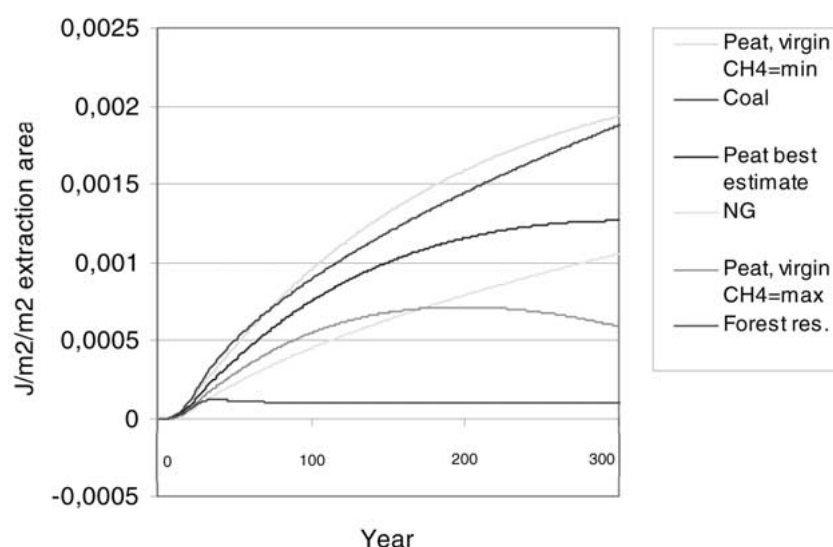
Figure 14. Accumulated RF for different emission rates of CH$_4$ from virgin mires. Mean value for C uptake rate assumed.

and is not displayed explicitly in a figure. The two last options can possibly be seen as contradictive to some degree if the peatland is not managed during the whole after-treatment period.

If on the other hand a 'worst-worst-case' is assumed (i.e. with a low CO$_2$-uptake rate combined with low (avoided) CH$_4$ emissions from the virgin mire and high CH$_4$ emissions from the restored wetland), this will generate an accumulated RF higher than coal over the entire time period. The 'worst-worst-case' scenario is constructed by combining *Peat CO$_2$-uptake = min* in Figure 11 with *Peat, CH$_4$-wetl = max* in Figure 13 and *Peat, virgin CH$_4$=min* in Figure 14, and is not displayed explicitly in a figure.

The 'best-best-case' and 'worst-worst-case' scenarios described above are to be considered as extreme values for the global change impact from the peatland – wetland scenario. It is unlikely that those conditions will occur very often. The 'normal' global change impact of peat utilisation with restoration of wetland as after-treatment will lie somewhere in between the extreme values. The data quality in this study for CH$_4$ emissions and accumulation of C in a new, restored wetland is however unsatisfactory. The range in C uptake rate used here is very wide and the data are uncertain. The results for restoration of wetland as after-treatment are therefore to be considered more uncertain than for afforestation.
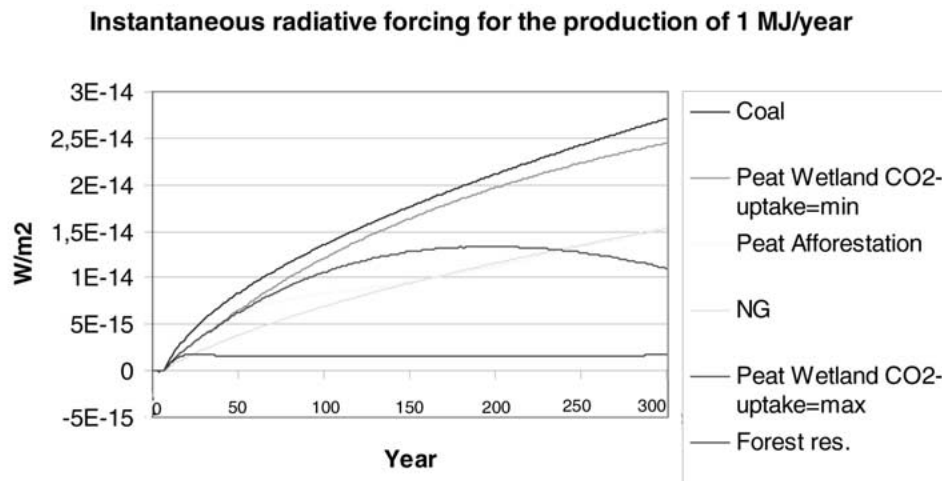
Instantaneous radiative forcing for the production of 1 MJ/year

Figure 15. Immediate RF from the use of coal, natural gas, forest residues and peat for energy assuming the production of 1 MJ/year for 300 years. Three peat fuel systems are presented. 1) Peat afforestation: After the peat bog is mined, the area is forested and the consequent uptake of CO_2 is credited to the peat. We have not considered future use of this forest. 2) Peat Wetland CO_2-uptake = min. After the peat bog is mined the area is turned into a managed wetland. In this scenario we assume a minimum uptake of CO_2 in the wetland. 3) Peat Wetland CO_2-uptake = max. As above, but with the assumption of a maximum uptake of CO_2 in the wetland.

7.3. MULTIPLE GENERATIONS SCENARIO

In our earlier scenarios (1. Peat – Forestation – Bioenergy and 2. Peat – Wetland Restoration) peat is extracted and used for energy production during a period of 20 years. In this third scenario, when the first generation of peat is extracted after 20 years, we add a new generation. A total of 15 generations are added, with a time delay of 20 years, thus adding up to 300 years of continuous peat extraction supplying 1 MJ/year. We consider two different types of after-treatment:

– Forestation, but unlike our scenario 1, we do not consider harvesting the forest or the possible bio-energy production. The reason for this is to investigate the isolated effect of producing 1 MJ/year from peat exclusively. We here assume what is called 'best estimate' for forest growth rate and original methane emissions from the virgin mire.
– Wetland restoration, as in our scenario 2. We here assume for forest growth rate, for original methane emissions from the virgin mire and for CH_4 emissions from the restored wetland.

Figure 15 and 16 below show the resulting immediate and accumulated RF respectively. In the figures, these fuel systems are compared with coal, natural gas and forest residues. Each energy system produces 1 MJ per year for 300 years.

The immediate RF from 'Peat – Afforestation' is comparable to natural gas. The immediate radiative forcing from 'Peat – Wetland Restoration' with minimum

Accumulated radiative forcing for the production of 1 MJ/year for 300 years
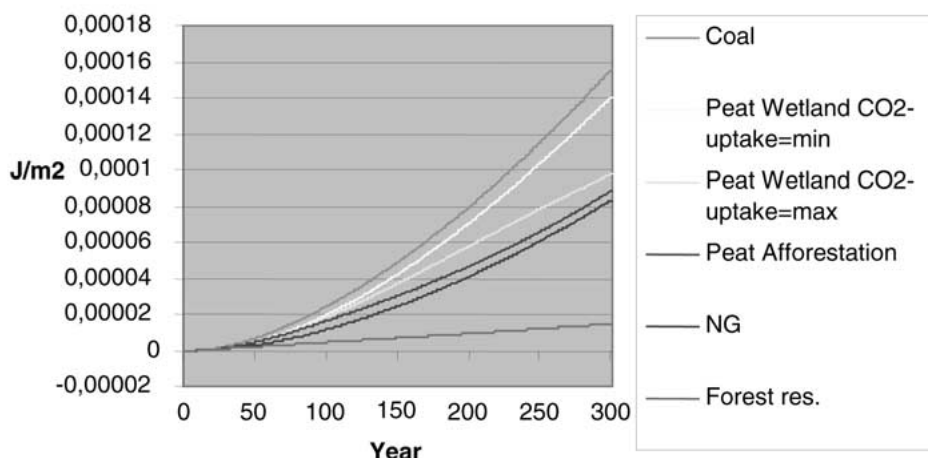

Figure 16. Accumulated RF from the use of coal, natural gas, forest residues and peat for energy assuming the production of 1 MJ/year for 300 years. Three peat fuel systems are presented. 1) Peat afforestation: After the peat bog is mined, the area is forested and the consequent uptake of CO_2 during 85 years is credited to the peat. We have not considered future use of this forest. 2) Peat Wetland CO_2-uptake = min. After the peat bog is mined the area is turned into a managed wetland. In this scenario we assume a minimum uptake of CO2 in the wetland. 3) Peat Wetland CO_2-uptake = max. As above, but with the assumption of a maximum uptake of CO_2 in the wetland.

CO_2-uptake is slightly lower than coal. The instantaneous radiative forcing from 'Peat – Wetland Restoration' with maximum CO_2-uptake is initially higher than natural gas, but starts decreasing at 200 years, is equal to natural gas at ca 240 years and continues to decrease thereafter. This is due to the negative immediate forcing that occurs after 200 years for each added generation in this scenario (see Figure 10), which is a result from the uptake of CO_2 in the restored wetland and avoided CH_4 emissions from the virgin peatland.

The accumulated radiative forcing from all the three peat scenarios lies between coal and natural gas in a 300 year perspective. Using a longer time perspective than 300 years will result in a decreasing accumulated RF from the scenario 'Peat wetland with maximum CO_2-uptake'. This is due to the negative instantaneous forcing that occurs after 200 years for each added generation (see Figure 10).

7.4. COMPARISON OF RADIATIVE FORCING WITH THE CONCEPT OF GLOBAL WARMING POTENTIALS

In this study we have chosen to use radiative forcing as an approximation to the contribution to global change impact. The IPCC recommends that the GWP-concept

TABLE VIII

Using the GWPs for approximating the global change impact of peat scenarios compared to using RF. Fifteen 20-year generations of peat extraction are added with a time delay of 20 years, thus adding up to 300 years of continuous peat extraction supplying 1 MJ/year. Resulting emissions are multiplied with corresponding GWP-values and summarised over the 300 year period. For comparison accumulated RF values are presented.

	CO2-equivalents (GWP 500) summarised over 300 years		Accumulated radiative forcing at t = 300	
	kg CO2-equiv.	Relative coal	mJ/m2	Relative coal
Coal	31	1.00	0.160	1.00
Peat – Wetland CO2-uptake = min	31	1.00	0.140	0.90
Peat – Wetland CO2-uptake = max	14	0.46	0.098	0.63
Peat – Afforestation	23	0.73	0.090	0.57
Natural Gas	18	0.59	0.083	0.54
Forest residues	2	0.07	0.015	0.09

should be used to calculate and compare GHG emissions on national and international level. The reason RF used in this study instead of GWP is that:

- RF can describe the impact of an emission scenario that stretches over a long time, which the GWP-concept isn't meant to be used for.
- GWP is a relative measure. A GWP today is not the same as a GWP year 2100.
- According to model studies performed by the IPCC, there seems to exist a direct relation between RF and global average temperature.

In order to investigate the difference between using radiative forcing versus the GWP-concept we have performed a GWP-calculation of the 'multiple generations scenario', earlier described in section 7.3. In this scenario a total of 15 generations of peat extraction are added with a time delay of 20 years, thus adding up to 300 years of continuous peat extraction supplying 1 MJ/year. The GWP-calculation from this scenario has been performed in the following way: Annual emissions and uptake of CO_2, CH_4 and NO_2 are compiled, multiplied by corresponding GWP-factor and finally summarised over the 300 year time period. GWP-values for a 500-year perspective have been used: 1,0 for CO_2, 7,0 for CH_4 and 156 for N_2O. The results are shown in Table VIII expressed in g CO_2-equvalents and as a value relative coal. For comparison accumulated RF values are given for the multiple generations scenario, as earlier presented in Figure 15.

Discussion

Global Warming Potentials are defined as the accumulated RF between t = 0 and some chosen time horizon, in our case t = 500, caused by a unit mass of gas

emitted at t = 0. In the hypothetical case that an emission scenario consists of an emission at t = 0 and no further emissions after t = 0, using the GWP-concept would give exactly the same result as calulating the accumulated RF at t = 500. However, in our multiple generations scenario the emissions are spread over time, so the GWP-concept will give a different result compared to using the concept of accumulated RF. This difference in results is mainly due to the following: With the GWP-concept one kg gas emitted at year 0 gives the same contribution to the total impact as one kg gas emitted at year 299. Both emissions are time integrated over 500 years. In contrast, with the concept of accumulated RF one kg gas emitted at t = 0 is time integrated over 300 years, while one kg gas emitted at t = 299 is integrated over one year only. Therefore, with accumulated RF, one kg gas emitted at t = 0 gives a much larger contribution to the total climate impact than one kg gas emitted at t = 299. In other words, with the GWP-concept emissions occurring late in time are over-represented in the calculated climate impact compared to using concept of accumulated RF.

For the natural gas and forest residues scenarios the difference between using the GWP-concept and accumulated RF is small. This can be explained by the fact that the dominating emissions occur early in the scenario, at the combustion phase. In contrast, for the peat scenarios there is a considerable difference between using GWP-concept and using accumulated RF. This difference can be explained by the fact that there are important emissions/uptake that occur late in the peat scenario.

8. Discussion

The results from this study show that the range for RF from peat utilisation can be very large. This is also confirmed by the fact that previous studies on global change impact from peat utilisation have come to different conclusions, depending on the assumptions made for CH_4 emissions from virgin mires, C uptake in forest biomass etc. Savolainen et al. (1994) state that the global change impact from peat utilisation is comparable to coal, Rodhe and Svensson (1995) compares peat to fossil oil, in Zetterberg and Klemedtsson (1996) peat is comparable to fossil oil or natural gas and in Åstrand et al. (1997) peat is comparable to forest residues. The sensitivity analysis performed in this study is an attempt to find the parameters that are most significant for the global change impact.

Calculations of the global change impact from peat utilisation need to consider many uncertainties in data and assumptions that can affect the results in different ways that have not been assessed here. Some examples are:

- A higher assumed extracted peat depth would for example increase the relative global change impact, while a decreased depth would decrease the global change impact. This is because the increased peat extraction depth decreases the peatland area needed to produce a given amount of energy, and the decreased area lowers the amount of avoided CH_4 emissions.

TABLE IX

Global change impact from the peat-afforestation scenario. Accumulated RF from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages of the peatland – forestation – bioenergy scenario. For the peat-affrestation scenario energy is produced both from peat and from the forest that is consequently grown. Best estimate for forest growth is 6.5 m^3/ha*year.

Energy type	Case	Description	Accumulated radiative forcing at t = 300 years mJ/m^2/m^2 extraction area
Peat afforestation	Best estimate	Best estimate forest growth rate Best estimate virgin CH_4-emissions	1.35
Peat afforestation	High virgin CH_4-emissions	Best estimate forest growth rate High virgin CH_4-emissions	0.68
Peat afforestation	Low virgin CH_4-emissions	Best estimate forest growth rate Low virgin CH_4-emissions	2.02
Coal			3.13
Natural gas			1.71
Forest residues			0.42

- The original state of the mire that is utilised will have a significant impact on the results. If for example a forest-drained mire is utilised, the results will become quite different because of the original emissions of such a mire (no CH_4 emissions, emission of CO_2 due to peat oxidation).
- No range in CO_2 emissions due to peat oxidation has been investigated here (a 'worst case' was assumed). Lower emissions from peat oxidation would decrease the global change impact relative the other fuels.
- Different options for the use of the forest biomass produced on the utilised peatland has not been studied here (we assumed that all the harvested forest biomass will be used for energy production). The global change impact from peatland-forestation – bioenergy utilisation would increase relative the other fuels if a lower fraction of the forest biomass was assumed to be used for energy production.

9. Conclusions

The accumulated RF from the peatland – forestation – bioenergy scenario is below natural gas in a 300 year time perspective assuming what is here called 'best estimate' for forest growth rate and original CH_4 emissions from the virgin mire.

TABLE X

Global change impact from the peat-wetland scenario. Accumulated RF from the use of coal, peat, natural gas and forest residues for energy. The amount of energy produced with coal, natural gas and forest residues is the same as the amount of energy produced by 1 m^2 peat area at different stages.

Energy type and case	Case	Description	Accumulated radiative forcing at t = 300 years $mJ/m^2/m^2$ extraction area
Peat wetland	Best estimate	Best estimate CO_2-uptake Best estimate virgin CH_4-emissions Best estimate restored CH_4-emissions	1.27
Peat wetland	High CO_2-uptake	High CO_2-uptake Best estimate virgin CH_4-emissions Best estimate restored CH_4-emissions	0.73
Peat wetland	Low CO_2-uptake	Low CO_2-uptake Best estimate virgin CH_4-emissions Best estimate restored CH_4-emissions	1.80
Peat wetland	High virgin CH_4-emissions	Best estimate CO_2-uptake High virgin CH_4-emissions Best estimate restored CH_4-emissions	0.59
Peat wetland	Low virgin CH_4-emissions	Best estimate CO_2-uptake Low virgin CH_4-emissions Best estimate restored CH_4-emissions	1.93
Peat wetland	Low restored CH_4-emissions	Best estimate CO_2-uptake Best estimate virgin CH_4-emissions Low restored CH_4-emissions	0.68
Peat wetland	High restored CH_4-emissions	Best estimate CO_2-uptake Best estimate virgin CH_4-emissions High restored CH_4-emissions	1.92
Coal			1.88
Natural gas			1.06
Forest residues			0.10

The accumulated RF from the peatland – wetland scenario, will lie between coal and 2/3 of natural gas in a 300-year perspective, depending on the assumed C uptake rates for the wetland and assuming what is here called 'best estimate' for CH_4 emissions from a restored wetland.

The accumulated RF in the 'multiple generations' – scenario, show that all peat scenarios (Peat forestation – bioenergy, Peat wetland high CO_2-uptake and Peat wetland low CO_2-uptake) will lie between coal and natural gas in a 300-year perspective. We here assume what is called 'best estimate' for forest growth rate,

TABLE XI

Global change impact from the multiple generations scenario. Accumulated RF from the use of coal, natural gas, forest residues and peat for energy assuming the production of 1 MJ/year for 300 years. Three peat fuel systems are presented. 1) Peat afforestation: After the peat bog is mined, the area is forested and the consequent uptake of CO_2 during 85 years is credited to the peat. We have not considered future use of this forest. 2) Peat Wetland high CO_2-uptake. After the peat bog is mined the area is turned into a managed wetland. In this scenario we assume a maximum uptake of CO_2 in the wetland. 3) Peat Wetland low CO_2-uptake. As above, but with the assumption of a minimum uptake of carbon dioxide in the wetland.

Energy type	Case	Description	Accumulated radiative forcing at t = 300 years mJ/m^2
Peat afforestation	Best estimate	Medium forest growth rate Medium virgin CH_4-emissions	0.090
Peat wetland	High CO_2-uptake	High CO_2-uptake Medium virgin CH_4-emissions Medium restored CH_4-emissions	0.098
Peat wetland	Low CO_2-uptake	Low CO_2-uptake Medium virgin CH_4-emissions Medium restored CH_4-emissions	0.140
Coal			0.160
Natural gas			0.083
Forest residues			0.015

for original CH_4 emissions from the virgin mire and for CH_4 emissions from the restored wetland. Using a longer time perspective than 300 years will result in lower accumulated RF from the scenario 'Peat wetland high CO_2-uptake'. This is due to the negative instantaneous forcing that occurs after 200 years for each added generation.

There are great uncertainties related to the data used for emissions and uptake of GHGs in restored wetlands. The most important factors to consider in that matter are emissions of CH_4, uptake of CO_2 and how these two parameters will change over time. The mechanisms affecting these parameters in a restored wetland should be studied further.

The importance of N_2O-emissions from soil processes on the global change impact from peat utilisation, have often been discussed (e.g. at the hearing mentioned in section 1). The results from this study show that the magnitude of these emissions can affect the result by approximately $\pm$ 10%, but also that they are of minor importance compared to C uptake rate during after-treatment and methane emissions from virgin and restored mires.

All fuel systems studied here, except one, will generate an accumulated RF that is increasing over time. The only exception is the peatland – wetland scenario with a high C uptake rate in the restored wetland, where the results show that the accumulated RF will start to decrease after 200–250 years (as a result of that the instantaneous RF decreases below zero). It is however very difficult to predict how C uptake and CH_4 emissions in a restored wetland will develop over time why these results are uncertain.

Previous studies of the global change impact from peat utilisation state that the climate impact can be comparable both to coal (Savolainen et al. 1994) and fossil oil (Rodhe and Svensson 1995) as well as comparable to forest residues (Åstrand et al. 1997). The range between the different previous results is large. The results from the present study show that all these scenarios are possible depending on the characteristics of the specific mire. For the peatland – forestation – bioenergy scenario, the global change impact is highly dependent on the CH_4 emissions from the virgin mire, as well as the growth rate of the forest planted after peat extraction has finished. For the peatland – wetland scenario, the global change impact is highly dependent on the CH_4 emissions from the virgin mire and the CO_2-uptake rate and CH4 emissions of the restored wetland. It is possible and recommended to take mire characteristics and after-treatment alternatives into consideration when planning peat utilisation, in order to minimize the climate impact. Such analyses in the planning-stage could be made quite easily with the help of a model similar to the one used in this study combined with an emission database for different mire types and after-treatment alternatives.

10. Recommendations for Minimizing the Climate Impact from Peat Utilisation

- It is important to consider CH_4 emissions from the virgin mire when choosing mires for utilisation. High original CH_4 emissions will result in a significantly lower total climate impact than if we assume low original emissions (see for definitions of high and low emissions). In order to minimize the climate impact one should preferably choose mires with high CH_4 emissions for utilisation.
- If afforestation is chosen as after-treatment strategy, the goal should be to achieve a high forest growth rate, both for the extraction area and the surrounding area. A high forest growth rate gives lower climate impact than a low forest growth rate (see for definitions of high and low growth rates). This is because in the peatland-forestation – bioenergy scenario a considerable fraction of the energy is produced with wood. A high forest growth rate increases this fraction.
- The results from this study shows that restoration of wetland can reduce the climate impact from peat utilisation substantially. That is, however, based on

the assumption that the CO_2-uptake rate of the wetland is high and that the CH_4 emissions from the restored wetland remains on a quite low level. Those factors should be considered in the planning of the restoration.

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References

Alm, J., Talanov, A., Saarnio, S., Silvola, J., Ikkonen, E., Aaltonen, H., Nykänen, H. and Martikainen, P.J.: 1997. 'Reconstruction of the carbon balance for microsites in a boreal oligotrophic pine fen, Finland', *Oecologia* **110**, 423–431.

Alm, J., Tolonen, K. and Vasander, H.: 1992, *Determination of Recent Apparent Carbon Accumulation in Peat using Dated Fire Horizons*, Suo, Mires and Peat, pp. 191–194.

Anon: 1992, *Skogsdata 92 (in Swedish)*, Umeå, Swedish University of Agricultural Sciences.

Berg, B. and Ekbohm G.: 1993, 'Decomposing needle litter in lodgepole pine (*Pinus contorta*) and Scots pine (*Pinus sylvestris*) monocultural systems – is there a maximum mass loss?', *Scand. J. Forest Res.* **8**, 457–465.

Bohlin, E., personal communication April 2001, Umeå, Swedish University of Agricultural Sciences.

Crill, P., Hargreaves, K. and Korhola, A.: 2000, *The Role of Peat in Finnish Greenhouse Gas Balances,* Studies and Reports 10/2000, Helsinki, Ministry of Trade and Industry.

Eriksson, H., personal communication December 2000, National Board of Forestry.

Eriksson, H.: 1994, *Skogens roll i Sveriges koldioxidbudget*, Report from the Swedish Climate Delegation, SOU 1994, pp. 138.

Eriksson, H. and Hallsby, G.: 1992, *Biomass Fuels – Effects on the Carbon Dioxide Budget*, R1992: **10**, Stockholm, NUTEK.

Gärdenäs, A.I.: 1998, 'Soil organic matter in European forest floors in relation to stand characteristics and environmental factors', *Scand. J. Forest Res.* **13**, 274–283.

Hillebrand, K.: 1993, *The Greenhouse Effects of Peat Production and Use compared with Coal, Oil, Natural Gas and Wood,* VTT Communications 1494, Espoo, Technical Research Centre of Finland.

Hillebrand, K. and Wihersaari, M.: 1993, *The Greenhouse Effects of Pest Production and Use compared with Coal, Oil, Natural Gas and Wood (in Finnish)*, Reviews B: **143**, Helsinki, Ministry of Trade and Industry.

Hånell, B.: 1997, *Beräkningar av skogsproduktionens storlek efter beskogning på färdigbrutna torvtäkter,* appendix in Åstrand et al. (1997), Umeå, Swedish University of Agricultural Sciences.

Hånell, B., personal communication December 2000, Umeå, Swedish University of Agricultural Sciences.

HMAB 2000, Härjedalens Mineral AB, elementary analysis of peat, personal communication.

Hyvönen, R., Olsson, B.A., Lundkvist, H. and Staaf, H.: 2000, 'Decomposition and nutrient release from *Picea abies* (L.) Karst. and *Pinus sylvestris* L. logging residues', *Forest Ecol. & Manag.* **126**, 97–112.

IPCC: 1997, *Greenhouse Gas Inventory Reference Manual*, Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories, vol. 2, Bracknell, Intergovernmental Panel on Climate Change.

Kasimir-Klemedtsson, Å., Nilsson, M., Sundh, I. and Svensson, B.: 2001, in Å. Kasimir-Klemedtsson and M. Lilliesköld (eds.), *Växthusgasflöden från myrar och organogena jordar,* Report 5132, Stockholm, Swedish Environmental Protection Agency.

Klemedtsson, L., personal communication December 2000.

Larsson, L-E., personal communication December 2000. Swedish Peat Research Foundation.

Lilliesköld, M. and Nilsson, J.: 1997, *Kol i marken (Soil carbon)*, Swedish Environmental Protection Agency. Report 4782. (In Swedish, English summary)

Lundmark, J-E.: 1988, *Skogsmarkens ekologi. Ståndortsanpassat skogsbruk. Del* 2. The Forestry Board. Jönköping. (In Swedish)

Mälkki, H. and Frilander P.: 1997, *Life Cycle Assessment of Peat Utilisation in Finland,* VTT Publications **333**, Espoo, Technical Research Centre of Finland.

Nilsson, M., Mikkelä, C., Granberg, G., Sundh, I., Svensson, B. and Ranneby, B.: 2000, *Methane Emission from Swedish Mires – National and Regional Budgets and Dependence on Mire Vegetation,* in Press, Journal of Geophysical Research.

Nilsson, M., personal communication December 2000, Umeå, Swedish University of Agricultural Sciences.

Nykänen, H., Silvola, J., Alm, J. and Martikainen, P.: 1996, 'Fluxes of greenhouse gases CH_4, CO_2 and N_2O on some peat mining areas in Finland', in R. Laiho, J. Laine and H. Vasander (eds.), *Northern Peatlands in Global Climatic Change,* Proceeding of the International Workshop held in Hyytiälä, Finland, 8–12 October 1995, The Academy of Finland, Helsinki, pp. 141–147.

Olsson, B., Staaf, H., Lundkvist, H., Bengtsson, B. and Rosén, K.: 1996, 'Carbon and nitrogen in coniferous forest soils after clear-felling and harvests of different intensity', *Forest Ecol. & Manag.* **82**, 19–32.

Råsjö Torv 2000, elementary analysis of peat, personal communication.

Rodhe, H. and Svensson, B.: 1995, 'Impact on the greenhouse effect of peat mining and combustion', *Ambio* **24:4**, 221–225.

Savolainen, I., Hillebrand, K., Nousiainen, I. and Sinisalo, J.: 1994, *Greenhouse Impacts of the Use of Peat and Wood for Energy,* VTT Research Notes 1559, Espoo, Technical Research Centre of Finland.

Silvola, J.: 1986, 'Carbon dioxide dynamics in mires reclaimed for forestry in Eastern Finland', *Ann. Bot, Fennici* **23**, 59–67.

Sundh, I., Nilsson, M., Mikkelä, C., Granberg, G. and Svensson, B.H.: 2000, 'Fluxes of methane and carbon dioxide on peat-mining areas in Sweden', *Ambio* **29:8**, 499–503.

Tolonen, K., Vasander, H., Damman, A.W.H. and Clymo, R.S.: 1992, 'Rate of apparent and true carbon accumulation in boreal peatlands', in D. Fredriksson (ed.), *Proc. 9th Int. Peat Congr.* Vol. **1.**, Uppsala, Swedish Nat. Comm. IPS, pp. 319–333.

Tuittila, E.-S., Komulainen, V.-M., Vasander, H. and Laine, J.: 1999, 'Restored cut-away peatland as
    a sink for atmospheric CO_2], *Oecologica* **120**, 563–574.
Tuittila, E.-S., Komulainen, V.-M., Vasander, H., Nykänen, H., Martikainen, P.J. and Laine, J.: 2000,
    'Methane dynamics of a restored cut-away peatland', *Global Change Biol.* **6**, 569–581.
Turunen, J., Tolonen, K., Tolvanen, S., Remes, M., Ronkainen, J. and Jungner, H.: 1999, 'Carbon
    accumulation in the mineral subsoil of boreal mires', *Global Biogechem. Cycles* **13**, 71–79.
Turunen, J. and Tolonen, K.: 1996, 'Carbon accumulation in mires in Finland', in R. Laiho, J. Laine
    and H. Vasander (eds.), *Northern Peatlands in Global Climatic Phange*, Proceeding of the In-
    ternational Workshop held in Hyytiälä, Finland, 8–12 October 1995, Helsinki, The Academy of
    Finland, pp. 250–255.
Uppenberg, S. and Åhman, M.: 1998, *Climatic Impact of Future Swedish Transports,* IVL-Report
    B1286, Stockholm,Swedish Environmental Research Institute.
Uppenberg, S. and Zetterberg, L.: 1999, *Climatic Impact of Energy from Wind Power, Photovoltaics
    and Natural Gas Compared to Coal,* not published material, Stockholm, Swedish Environmental
    Research Institute.
Uppenberg, S., Brandel, M., Lindfors, L.-G., Marcus, H.-O., Wachtmeister, A. and Zetterberg, L.:
    1999, *Miljöfaktabok för bränslen,* IVL-Rapport B1334 A+B, Stockholm, Swedish Environ-
    mental Research Institute.
Zetterberg, L. and Hansén, O.: 1998, *Nettoemissioner av koldioxid till atmosfären vid använd-
    ning av hyggesrester för el- och värmeproduktion,* IVL-Rapport B1298, Stockholm, Swedish
    Environmental Research Institute.
Zetterberg, L. and Klemedtsson, L.: 1996, *The Contribution to the Greenhouse Effect from the Use
    of Peat and Coal for Energy,* IVL-Report B1237, Stockholm, Swedish Environmental Research
    Institute.
Zetterberg, L.: 1993, *A Method for Assessing the Expected Climatic Effects from Emission Scenarios
    Using the Quantity Radiative Forcing,* IVL-Report B1111, Stockholm, Swedish Environmental
    Research Institute.
Åstrand, L.: 2000, Swedish Peat Association, personal communication.
Åstrand, L., Ericson, S.-E. and Nyström, K.: 1997. *Torvbränsle och växthuseffekten,* Report 1997/8,
    Stockholm, Vattenfall Support AB.

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