The Allocation of European Union Allowances: Lessons, Unifying Themes and General Principles

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NOTA DI LAVORO 116.2006

SEPTEMBER 2006

CCMP – Climate Change Modelling and Policy

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Summary

This paper is the concluding chapter of *Rights, Rents and Fairness: Allocation in the European Emissions Trading Scheme,* edited by the co-authors and forthcoming from Cambridge University Press. The main objective of this paper is to distill the lessons and general principles to be learnt from the allocation of allowances in the European Union Emission Trading Scheme (EU ETS), i.e. in the world's first experience with allocating carbon allowances to sub-national entities. We discuss the lessons that emerge from this experience and make some comments on what seem to be more general principles informing the allocation process and on what are the global implications of the EU ETS. As has become obvious during the first allocation phase, the diversity of experience among the Member States is considerable, so that it must be understood that these lessons and unifying themes are drawn from the experience of most of the Member States, not necessarily from all. Lessons and unifying observations are grouped in three categories: those concerning the conditions encountered, the processes employed, and the actual choices.

Keywords: Climate Change, Emission Trading, Allocation, Fairness, EU Policy

JEL Classification: C72, H23, Q25, Q28

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The Allocation of European Union Allowances. Lessons, Unifying Themes and General Principles[1]

Barbara Buchner, Carlo Carraro and A. Denny Ellerman

1 Introduction

On January 1st, 2005, the EU Emissions Trading Scheme (EU ETS) scheme was officially launched, only two years after the European Council adopted the EU Emissions Trading Directive (European Community 2003). As a consequence of this formal start, the world's largest ever market in emissions has been established, and European companies now face a carbon-constrained reality in form of legally binding emission targets. Within essentially one year, 2004, the international carbon market has gained momentum through major policy developments and quick market responses, which among others have enabled the establishment of a framework for the EU carbon market.

The steady increase in market activity in the EU during 2004 has been substantial, both in terms of number of market participants and actual volumes traded. During 2004, as the start of the EU ETS came closer, it became evident that the market was highly sensitive to political developments (cf. Buchner 2006). Indeed, EU Member States needed to decide on the amount of GHG emissions allowances (EUAs) which are to be allocated for the period of 2005 to 2007 to large fixed sources of CO_2 - the national allocation plans (NAPs) - by March 2004[2].

[1] This is the concluding chapter of *Rights, Rents and Fairness: Allocation in the European Emissions Trading Scheme*, edited by the co-authors and forthcoming from the Cambridge University Press. The authors are, respectively, Researcher at the Fondazione Eni Enrico Mattei (FEEM); Professor of Economics at the University of Venice and Research Director of FEEM; and Senior Lecturer at MIT. The lessons and unifying themes presented here draw upon eleven contributions, which constitute the core of the book and which describe the experience of allocation in ten member-states of the European Union and as experienced in Brussels. We are endebted to the European Commission, FEEM, and MIT's Center for Energy and Environmental Policy Research and Joint Program on the Science and Policy of Global Change for the various forms of financial and other support that made this book possible. But our greatest debt is to the other participants of this project who were able to make sense of the allocation process in their countries and thereby provide the material from which these conclusions could be drawn. None of them should be held responsible for the views we express here and all errors of fact and interpretation are ours alone.

[2] A second allocation phase is scheduled for the period from 2008 to 2012.

As the uncertainty about the final allowance allocations represented on of the major reasons why significant activities in the EU ETS were still rare until the beginning of 2004, information on the allocation process had strong implications on the market developments. In particular, rumours and decisions on the National Allocation Plans (NAPs) spurred substantial volumetric activity and price movement. The reason for this development is that the initial allocation of emission rights determines the overall shortage of the market.

Given the importance of the allocation process for the overall efficiency of any potential emission trading scheme, the process of allocating the emission allowances in Europe has attracted world-wide attention. For this reason, the main objective of the present paper is to distill the lessons and general principles to be learnt from the world's first experience with allocating carbon allowances to sub-national entities based on insights emerging from ten Member States[3]. We discuss the lessons that emerge from this experience and make some concluding comments on what seem to be more general principles informing the allocation process and on what are the global implications of the EU ETS. As has become obvious during the first allocation phase, the diversity of experience among the Member States is considerable, so that it must be understood that these lessons and unifying themes are drawn from the experience of most of the Member States, not necessarily from all. For every lesson and each general principle, there is typically at least one Member State for which it does not apply, or it does so only weakly. Accordingly, as is true of all lessons and concluding comments, these will need to be applied carefully and some may not apply to future allocations in the EU ETS or to the circumstances surrounding future allocations of CO_2 rights by other countries. Still, it seems likely that most of the problems experienced by the Member States of the European Union will be encountered by others who follow this example and that the lessons and general principles drawn from them will be helpful.

These lessons and unifying observations are grouped in three categories: those concerning the conditions encountered (section 2), the processes employed (section 3), and the actual choices (section 4). The last section closes with some comments about the uniqueness of this policy experiment, both as regards more general principles and its global implications.

[3] The analysis has focused on ten representative Member States, including (in alphabetical order) Czech Republic, Denmark, Germany, Hungary, Italy, Ireland, Poland, Spain, Sweden and the United Kingdom. These Member States account for more than 70% of the total first phase allocation in the EU ETS.

2 The conditions encountered

2.1 Data availability limits allocation choices

The lack of data at the level of the installation was perhaps the biggest problem confronted in the allocation process by nearly all Member States. This came as a surprise to most people since all countries had developed reasonably good inventories of CO_2 emissions data[4]. The problem was that the inventory data were developed from statistics of aggregate energy use and they did not extend to the level of the installation, which was the mandated recipient of the allowance allocations by the EU Directive. Since all Member States were operating under very tight time constraints in submitting NAPs, obtaining installation-level data became the first major hurdle that had to be cleared and the final allocation choices were strongly influenced by considerations of what data could be obtained within the available time.

The data problem existed regardless of the history of data collection or the extent of pre-existing energy or environmental regulation. Member States with a long history of energy and environmental regulation such as Germany and Sweden faced as large a problem as those with less, such as Spain and Italy, not to mention the accession countries of Eastern Europe. Some Member States had collected installation-level data for many of the facilities covered by the EU ETS, but the discrepancies between the earlier data and that submitted in the allocation process could be as much as 20%. The only Member State that did not face a significant data problem was Denmark, which had already established a CO_2 emissions trading scheme that included most of the emissions to be included in the EU ETS.

The problem of data availability was compounded by the lack of legal authority to collect the relevant data. When combined with the pressing deadlines for NAP submission, governments had little choice but to rely heavily on voluntary submissions from industry, while they also initiated action to acquire the requisite legal authority. The surprising thing is that the affected firms cooperated as fully and in as good faith as appears to have been the case. This cooperation may have reflected recognition of the ultimate power of the government to compel performance, but it is also true that the production of the requested data established a claim on the allowances being distributed and a failure to produce data would have resulted in no allocation to the installation, as well as other sanctions.

[4] Amongst others, also because an early study commissioned by the European Commission (NERA, 2002) had focused on the importance of data and pointed out the limitations that data availability placed on the practicability of various types of allowance allocation.

The limitations imposed by data availability had important consequences in ruling out certain baselines and types of allocation for which an *a priori* preference may have existed. For instance, Germany had advocated that allocations be based on 1990 emissions. This would have been in keeping with the Kyoto Protocol and with the EU Burden Sharing Agreement and it would have recognised 'early action.' It soon became evident, however, that data on installation level emissions in 1990 were non-existent and, in 2003, irretrievable in any reliable or meaningful form. Some Member States with better data could choose baselines that extended as far back as 1998 (UK, Sweden, Denmark), but for most countries, the baseline or reference periods for allocation included only the most recent few years because these were the only years for which installation level data could be easily retrieved. Consequently, baselines that would automatically recognise 'early action' were infeasible. If any recognition was given to 'early action,' it was the subject of special provisions for those who had the data and could make a convincing case. Among the ten Member States examined (see footnote 3), only Germany, the Czech Republic, Hungary, and arguably the UK (in the baseline and rationalisation rules) made such provision. The more general pattern was to disregard early action not only because of the data problems but also on account of the conceptual problem of distinguishing 'early action' from emission reductions taken for other reasons.

While data availability limited allocation choices, a much noted by-product of the need to acquire installation-level data for allowance allocation was the resulting significant improvement in the quality of the data on emissions and energy use.

2.2 Inclusion of small facilities is not worth it

The EU Emissions Trading Directive established a very low level of heat input (20 MW thermal) as the threshold for inclusion in the ETS. If there is one refrain that arises from virtually every one of the ten NAP processes that we have analysed in detail, it is that the inclusion of small installations was not worth it. As noted in one contribution after another, a large proportion of CO_2 emissions originate from a small number of installations, while a very large number of the installations contribute only a small percentage of emissions. For instance, in the UK, 20% of the sites account for 94% of emissions and 80% of the sites contribute 6% of the emissions. Similar statistics are found in every Member State.

The problems relating to the size threshold are two-fold. First, and most evident in this first allocation cycle, data requirements are installation-specific. Therefore, much of the data problem discussed above was created by the small size threshold. Secondly, the reporting and verification requirements will impose costs on small installations that are disproportionate to their emissions or the abatement that could be expected from them.[5]

While the inclusion of small installations required more time and effort than would appear to be justified by their emissions or abatement potential, the alternatives are not obvious. The problem with any size threshold is that it has the potential to create a competitive disadvantage for covered installations and a perverse incentive to downsize in order to avoid regulation. And, the higher the threshold, the greater these problems are likely to be. Perhaps a staged approach, whereby small installations were brought in later, would have reduced the initial data problems, but those problems are now solved and no longer at issue.

Reporting and verification has just started so that the extent of the burden is not fully known. Similar size thresholds in US systems have not resulted in transaction costs that have created a noticeable problem despite similar reporting procedures.[6] The only way around this burden would seem to be an upstream point of regulatory obligation – at the refinery, gas terminal, or coal mine – that would result in a fuel price that included the price of carbon. This would have the same effects on abatement by small installations without the transaction costs involved in a downstream monitoring and reporting requirement.

Without a doubt, small installations will need to be part of successful emissions trading scheme in the longer run. In the medium term, the obvious burden for small installations deriving from the high transaction costs in relation to relatively low environmental benefits needs to be eased. Yet, the ultimate solution presented above presumes that the system eventually becomes more comprehensive, covering also other sectors. In order to partially resolve the problem of small installations in the shorter term, a judicious opt-out provision might be a promising way. Currently, the Directive foresees a temporary exclusion of installations only for the Phase 1 of the EU ETS. In order to avoid excessive transaction costs for small installations, the continuation of the opt-out possibility would be an attractive way

[5] Schleich and Betz (2004) discuss the problems of transaction costs for small and medium sized companies in more detail.

[6] Small installations in the US SO_2 program are generally exempt from the requirement to install a Continuous Emissions Monitoring System and instead report emissions based on fuel use and engineering calculations. For a more detailed discussion see e.g. Elleman *et al.* (2000).

to increase the efficiency of the EU ETS. As is the case with the Phase I opt-out provision, the exercise of this option could be coupled with a requirement of equivalent regulatory measures.

As a matter of fact, a number of these installations are already interested in participating in the EU ETS, for reasons related to the investments made for Phase 1, allowance allocations, or circumstances related to their advanced technologies or low energy intensity. Allowing installations below a certain threshold to opt-out appears to be a better way to lower the burden of these facilities and to increase the scheme's positive participation incentives and its overall efficiency than attempting to set new thresholds, given the potential danger of creating some discriminatory effects.

3 The processes employed

3.1 Emitters are involved in allocation decisions

In all the Member States examined in our project, except one, the allocation process can best be described as an extended dialogue between the government and industry. The involvement of industry in the process is not surprising given the data problem we have just described, but there were other factors as well. The Emissions Trading Directive mandated that at least 95% of the allowances in Member States be allocated to the installations that would be included in the scheme. Even had there been no data problems at the installation level, any democratically elected government would have considered it prudent to consult with the recipients, who were in addition well aware of the value of the endowments they were to receive. These two factors worked together to create an intense iterative process between the relevant parts of the Member State governments and the affected industry whereby data was collected, cross-checked, and refined at the same time that distribution proposals were made, evaluated against the data, and modified until a final NAP emerged. This interactive process was a key factor in successfully completing the NAP process.

The government role in the process was as much one of managing a process by which conflicting claims could be resolved as it was one of imposing any pre-conceived idea of how allowances should be distributed. The government was always the final arbiter of conflicts, but the actual exercise of this role was more the exception than the rule and it was always a last resort. On the part of industry, there was of course much lobbying, but the fixed total forced all players into a zero-sum game where a defensive concern about what competitors would receive became as important as offensive attempts to gain more for themselves.

Evidence of the government's role as organiser and arbiter of the process can be seen in several choices not taken. The 'pooling' option, which would have effectively delegated installation-level allocation to some industrial association, was never chosen. Similarly, in Spain, an early idea to have sector associations make installation-level allocations was set aside at an early date in favor of a process managed by the government. More generally, the frequency with which the word 'fair' appears in describing industry concerns indicates the extent to which the government's role was one of finding a reasonably equitable resolution of the conflicting claims that would permit a final NAP to emerge. The process was inevitably contentious, but it never broke down and there are at least as many comments on the cooperative aspects of the process as there are to the evident conflicts.

The government participants in this process were nearly always the environmental ministry in the lead with the ministry charged with economy or trade heavily involved. Sometimes the process started out as a more or less technical exercise within the environmental ministry, but the economics/trade ministry became heavily involved either as a means of obtaining the necessary data or at the instigation of industry. The relations between the environmental and economics/trade ministries could be contentious and even require resolution by the head of the government, but the relation was as often a cooperative one especially towards the end of the process when the prospect of confronting and persuading Brussels loomed. On the industry side, sector associations played a vital role in nearly every Member State both in obtaining the necessary historical data and in negotiating for the sector, although where few firms were involved and towards the end of the process the role of individual firms became greater.

Ireland is one exception to the active involvement of industry in the allocation process. Here, an ongoing investigation of an earlier scandal involving similar endowments led to the delegation of the task to the independent expert agency that issued environmental permits and was responsible for monitoring compliance with environmental regulations. The government retained the power to decide the total and the basic distribution principles upon the agency's recommendation after public consultation, but all the technical work was done within the agency with the help of consultants and some advisory groups that included industry representatives.

Two parties were noticeably absent from the distributive part of the NAP process, environmental non-governmental organisations (NGOs) and Brussels. Depending on the country, NGOs were either absent in any meaningful sense or they tended to focus on the total number of allowances. To the extent they were concerned about how the total was to be split, it was to ensure that favored activities, such as cogeneration or district heating received favorable treatment. Perhaps the most notable absence from the debate on internal allocation, given the frequent calls for 'harmonisation' was the European Commission. Aside from suggesting how the internal distribution might be done in an 'informal' non- paper and performing a perfunctory review for state aid, the Commission stayed out of the controversies about allocating allowances within Member States in keeping with the subsidiarity principle. In its NAP decisions, the Commission fixed the overall amount but explicitly allowed for redistribution within that envelope in case of data improvements.

3.2 Projections played a major role despite their unreliability

In all Member States, projections of CO_2 emissions and the associated modeling played a large role in determining national and sector totals. Although the use of predictions is sure to involve some error and be subject to subtle gaming, their use was unavoidable and they had the merit of narrowing debate about projected emissions to the underlying assumptions and imposing some top-side discipline on expansive bottom-up claims.

At the national level, projections became necessary because no Member State wished to deviate far from expected emissions in deciding the total to allocate to installations. BAU emissions were explicitly the constraint for Member States not facing a Kyoto constraint, as was the case for most of the accession countries in Eastern Europe. But even for the EU15, for whom compliance with the targets of the Kyoto Protocol or EU Burden-Sharing Agreement pose more of a problem and for whom the 'Path to Kyoto' provided an alternative criterion, only a gently constraining total was chosen, as will be discussed shortly, and that criterion necessitated the use of projections to determine what emissions could be expected to be.

The second major use of projections in the EU ETS was in establishing sector totals. Most Member States chose to allocate the national total in a two-step process whereby the national total was broken down into sector totals, which were then split among the installations in each sector. For these sector allocations, the use of projections followed from the decision to endow (non-electric) industrial sectors with as many allowances as 'needed,' as we will be discussed more fully in a subsequent lesson. Moreover, since all sectors were not expected to grow at the same rate, it became necessary to develop projections for each sector.

The use of projections also led to modeling problems. At the national level, no model captured the trading sector exactly so that every Member State had to revise existing sector models to more closely approximate the ETS sectors, and this in turn had to wait on the availability of data to define the baselines for those sectors. The heterogeneity of non-electric industrial production also led to a phenomenon of hyper-differentiation of sectors. Just as cement, steel, or pulp and paper would not be expected to experience the same rates of growth and therefore of 'need', so it was that groups of installations within the each of the broad sectors did not expect to have similar rates of emissions growth because of the somewhat different products that they produced and other intra-sector differences among firms. This process of differentiation was carried furthest in the UK where the originally proposed 14 sectors became 52.

The problem with the use of projections is that no projection will be accurate because of errors in expectation concerning important determinants of CO_2 emissions such as the rate of economic growth, relative energy prices (especially that between coal and natural gas), the ongoing rate of improvement in energy efficiency, and other structural transformations in the economy that will either increase or decrease CO_2 emissions. When the totals are at or close to the projected total, prediction errors will have a much greater effect on the unexpected tightness or slackness in the constraint and on allowance prices. At best, projections provided a range of estimates of BAU emissions and the choice of a total implicitly involved estimates of the probability of over-allocation. Agreeing on a central value or even a range would have been hard enough for any of the EU15, but it was even harder for the East European economies that are undergoing a fundamental structural transformation. From the standpoint of the Commission, the problem was one of avoiding a national total that had a high probability of creating surplus allowances. Although Brussels reduced a number of the proposed caps by significant amounts, the likelihood of some excess allowances was not eliminated, especially in Eastern Europe.

Notwithstanding these problems, projections did serve some useful purposes. Most generally, they provided a form of top-down discipline by constraining aggregated, bottom-up estimates of 'need'. As noted by Istvan Bart (one of the contributors), 'a reasonable emissions forecast for a sector is by definition lower than the sum of the safety points that would satisfy the expectations of all the sector's players.' Projections also served to channel the debate concerning totals into arguments about the reasonableness of underlying assumptions and consistency with projections used for other purposes. These helpful aspects of the use of projections were especially evident in the Czech Republic and Poland.

3.3 Central coordination is important

A distinguishing feature of the allocation process in the EU ETS is the highly decentralised manner in which it was done. This characteristic is what could be expected of a multilateral system in which the constituent members retain significant elements of national sovereignty, not to mention one in which the principle of subsidiarity is enshrined in principle and practice. Nevertheless, the role of the center was critical in arriving at the result that can be observed today.[7] Indeed, it is hard to imagine how twenty-five nations could have succeeded in such a multi-national enterprise without the central coordinating role played by the European Commission. Three aspects of this role are especially important.

The first and most visible role of the Commission was as agent for the whole in implementing a commonly agreed upon policy. As such, it found itself in the unenviable, and somewhat unexpected, role of being the enforcer of scarcity, as well as the agent insisting upon certain rules (such as no ex post adjustments) that promote an effective trading regime. The Commission could insist upon these conditions because the Emissions Trading Directive granted it the power to review and to reject individual NAPs, but this power had to be exercised judiciously. This delegation of considerable power to the central agent also allowed Member State governments, perhaps disingenuously, to shift the blame for unpopular decisions to an external authority that represented some greater good and thereby to make it easier for individual Member States to take unpopular decisions.

A less visible but probably equally important role of the Commission was as educator and facilitator of the decisions that Member States had to take. The degree of familiarity with emissions trading varied greatly among the Member States and for most a quick learning process was required. Studies funded by the Commission and guidance documents served to share the Commission's technical expertise in emissions trading and to inform Member States

[7] It can be argued that the Commission ensured that participation and optimal policy were jointly determined, as it has been suggested by recent research on climate policy (Cf. Buchner and Carraro 2004).

of the options available to them. A number of mechanisms were set up, such as Working Group 3, to facilitate the exchange of information at the technical level and to allow those charged with implementation in the Member States to share experiences among themselves. Had the Commission not taken this active role as educator and facilitator, it is doubtful that the EU ETS could have succeeded given the ambitious schedule for implementation and the inexperience of most Member States with this regulatory instrument. One other result is a degree of ‘soft’ harmonisation that is obscured by the not infrequent calls for still greater harmonisation of one provision or another.

A final aspect of role played by the Commission is the technical competence and political capability generally displayed in bringing the scheme to fruition. Technical competence in understanding what trading systems required was evident as early as the Green Paper which in March 2000 first publicly suggested that emissions trading might be one of the instruments to be included in the European Climate Change Programme (European Commission 2000), and it continued to be displayed in later proposals, guidance documents, and directives. A politically sensible approach was evident in the choice of an instrument that would not fall afoul of the Community’s unanimity rule (as had the earlier carbon tax proposal) but more generally in the minimalist approach in exercising its power to approve National Allocation Plans by focusing on only two issues, a total that was not overly generous and no ex post adjustments. These two conditions ensured some degree of scarcity and that trading would be necessary for compliance. Finally, low-key, back-channel, informal consultations were heavily used to avoid confrontations and to allow the process to move forward. For instance, conditional approval and subsequent, technical changes to the NAPs avoided outright rejection, thereby sparing Member States the unwelcome news coverage concerning the widely supported European endeavor to comply with the Kyoto Protocol. Yet, the Commission’s assessment process has also been criticised by some Member States as being too ‘high level’ and not involving enough technical expertise in the sense that decisions on the evaluation of the allocation plans have not always made by those who were familiar with the technical details of the different countries.

In a broader perspective, the Commission exercised the central coordination that the theoretical literature emphasises as necessary to correct for undersupplied goods and services in a decentralised market and to address distributional problems that can arise from uncoordinated, decentralised decisions. In this instance, the center provided a large part of the educational services needed to obtain a smooth and timely implementation and it acted to insure a reasonable degree of equity among Member States with respect to the burden that would be placed upon national industries included in the EU ETS. In this latter role, aspects of the Burden-sharing Agreement were effectively renegotiated to allow Member States with a greater deviation from the Path to Kyoto to adopt totals that were similar to those of other Member States not facing these problems. Finally, in their review of NAPs, the Commission also acted to ensure that individual provisions did not constitute unwarranted subsidy (*i.e.*, state aid).

4 The choices made

4.1 Benchmarking is little used

In no aspect of the allocation process for the EU ETS was the disparity between advocacy and practice greater than for benchmarking[8]. Although not always well defined, benchmarking refers to an allocation in which allowances are distributed according to some common emission rate multiplied usually by historical output. The emission rate is often one associated with best available technology, but it could also be an average emission rate for the sector. The common features is that installations having an emission rate higher than the standard will not receive more allowances, and those having an emission rate lower than the standard will not receive fewer allowances. As will be discussed in the next section, the basis for allocation was almost always recent emissions so that installations emitting more or less per unit of output received commensurately more or fewer allowances.

The failure to adopt benchmarking more widely was not because of a lack of trying. Many benchmarks were proposed; but, every time one was tried, the resulting deviations of allocations from recent emissions at the installation level were too great to gain wide acceptance. This points to what is the biggest problem in applying benchmarks: source heterogeneity. If all sources were more or less alike, benchmarking would be easy; but in practice installations differ greatly even within the same sector. Moreover, these differences lie not so much in energy or emissions efficiency, but in the specific output produced by the installation. (For instance, the energy and emissions associated with producing steel slab is not the same as what is required for finished rolls.) Thus, two facilities that may seem alike in producing similar quantities of output measured in some common denominator, such as tons,

[8] For a thorough discussion of the benchmarking allocation approach see Entec and NERA (2005).

may have very different emissions, not because one is producing more efficiently than the other, as is often implied in arguments for benchmarking, but because the products are different. An example of the extent to which output heterogeneity led to differentiation is provided by the Netherlands where 120 benchmarks were developed before the concept was abandoned.

Heterogeneity is not restricted to output; it can also affect inputs into a highly homogeneous product, such as electricity. A single, fuel-blind benchmark emission rate proved impossible to choose for existing facilities whenever the fuels used to generate electricity differed significantly in CO_2 content. Still other sources of heterogeneity exist as illustrated by the 26 separate benchmarks that were requested for the electric utility industry in Germany.

Heterogeneity in emission sources can be overcome if there is a widely accepted, pre-existing standard that can be applied. A good example is provided by the US SO_2 cap-and-trade program to which the EU ETS is often compared (cf. Ellerman *et al.* 2000 and Stavins 1998). Installations in the SO_2 program received an allocation that was benchmarked to 1.2 pounds of SO_2 per million Btu (approximately 500 grams/gigajoule) despite heterogeneity among affected sources that was comparable to that for CO_2 in the EU ETS. This emission rate is the same as the New Source Performance Standard (NSPS), adopted in the early 1970s as the best available control technology then available, and it has been applied since then with some modifications to all new generating units. By 1990, when the SO_2 program was adopted, the NSPS applied to about 60% of generation and it was an obvious benchmark. By comparison, nothing like this well-established standard for SO_2 emissions in the U.S. exists for CO_2 emissions in Europe or elsewhere. Thus, although plausible CO_2 benchmarks could be and were proposed, none had the institutional precedent and legal force that made adoption of the NSPS benchmark feasible in the US SO_2 trading program.

A final explanation for the absence of benchmarking is the problem of data availability. The informational requirements for a benchmarked allocation are more demanding than an allocation based on past emissions. CO_2 molecules are uniform and it was a lot easier to collect this common single data point across the great heterogeneity of affected facilities than it would have been to classify the facilities according to product (and sub-product) and decide upon a benchmark for each. When time was short and voluntary cooperation was required to produce the necessary data, this consideration became important. In addition, competitive considerations might also make firms less willing to reveal output or input data than emissions.

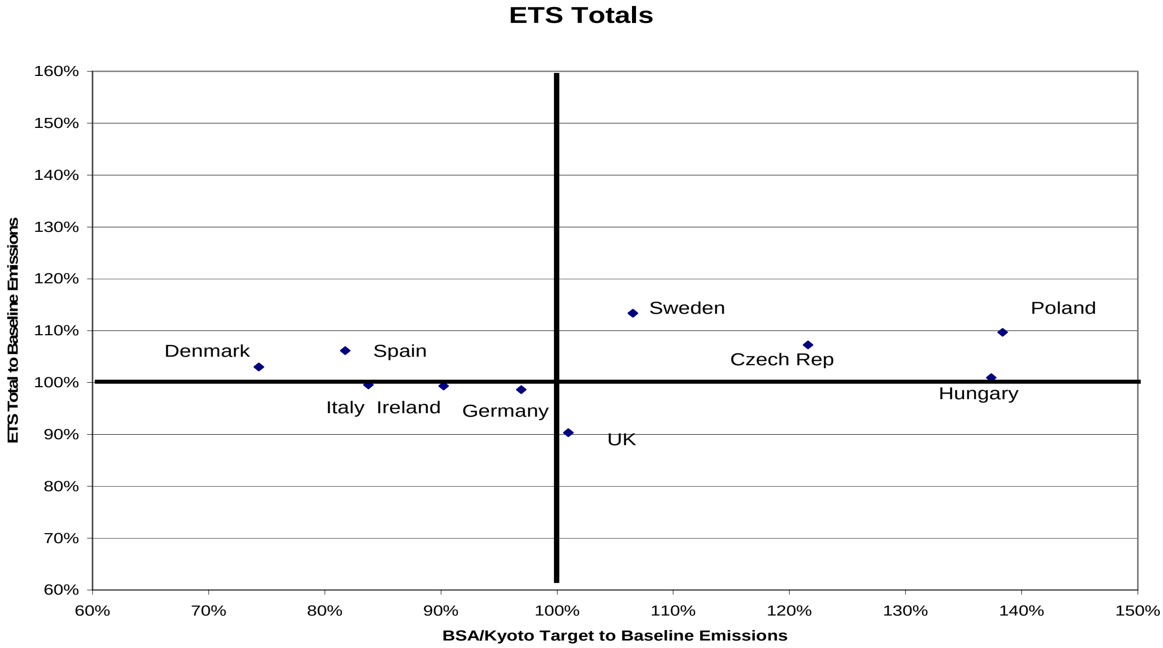
Despite the limited use of benchmarking, it was used in some cases and these cases illustrate the reasons for the limited use of benchmarking. The most common use is for allocations to new entrants for whom there is, by definition, no prior emissions history. Although the criteria applied for allocations from the new entrant reserves are often imprecise, the usual practice is to scale the allocation to capacity or expected output and to apply some benchmark generally reflecting best available technology, which is an obvious if ill-defined norm for new installations. Benchmarking is also applied to some existing installations. In both Denmark and Italy, benchmarks were used for the electricity sector. In both cases, the benchmark is the number of allowances remaining for the electric utility sector after allocation to the non-electric industrial sectors divided by baseline output of the electricity sector in Denmark and by expected output in Italy. In Denmark, the use of a benchmark for existing generation facilities was mandatory but it was made easier by the similarity of the generating fuel profile of the two main electric utilities and by the circumstance that the use of prior emissions would have punished facilities that reduced emissions under the earlier emissions trading program. In Spain, combined-cycle gas-fired units received a benchmarked allocation based on good practice, but these units were all recently built or under construction and therefore without complete baseline data. Also, certain industrial sectors received allocations proportional to capacity with an implicit benchmark of the average sector emission and utilisation rate because installation level data on emissions was lacking. In Poland and Italy, industrial sectors were given a choice and in each country several opted for benchmarking. In all of these cases, benchmarking was chosen either because historical data was not available, sources were homogeneous, or the sources in the affected industry could agree on a benchmark.

4.2 The main reference point for allocation is recent emissions

In the absence of an obvious or practical benchmark, recent emission levels became the basic reference point for the allocation process in the EU ETS. This was true both for the macro decision on the level of the cap and on the micro distribution of allowances to existing installations within individual Member States.

The macro level adherence to recent emissions is shown in Figure 1.

Figure 1 *The relation of recent emissions to the ETS allocation and the Kyoto target*



The horizontal axis represents the relationship between the Kyoto or BSA target (for the EU15) for national emissions and 2003 emissions. The spread among the ten Member States is considerable. Poland and Hungary have a target that is about 40% higher than recent emissions, while Denmark has a BSA target that is about 26% below recent emissions. The vertical axis represents the relation between total allowances and recent or baseline emissions for the trading sectors of each economy. Here the spread is much reduced. Poland, the Czech Republic, Sweden, Spain, and Denmark created totals that were from 3% to 13% above recent emissions, while Italy, Ireland, Germany, and Hungary were very close to recent emissions, and the UK was 10% below.[9]

[9] These relationships cannot be taken to indicate the relative stringency facing each of these countries. This depends upon expected growth which varies among the member states of the EU. Spain for instance is widely regarded as short in the EU ETS and it was so revealed in the 2005 emissions data (-6%), as were also the UK (-18%) and Ireland (-16%). Also, Denmark has highly variable annual emissions due to coal-fired electricity exports to the rest of Scandinavia, which depend on rainfall in Norway and Sweden. For instance, CO_2 emissions in 2002 and 2003 were 30.9 and 36.6 million tons, respectively. Denmark's ratio of 1.03 is calculated from the 1998-2003 average emissions for the ETS sectors.

The criteria for deciding total allocations by each Member State would suggest that countries for which the Kyoto/BSA target is greater than recent emissions could adopt a cap that would be somewhat higher, depending on projections of 2005-07 emissions. However, for countries with a Kyoto/BSA target below recent emission levels, as is the case for five of the countries in this sample of ten, an ETS total lower than recent emissions (although not necessarily lower than projected emissions) would seem to be implied. As can be readily seen, there is no relation between the Path to Kyoto and the ETS totals. All are clustered close to recent emissions. Significant gaps from the Path to Kyoto are to be bridged by the intended purchase of CDM and JI credits, as indicated in the NAPs of those Member States with Kyoto/BSA targets below recent emission levels.

While national and sector totals were typically based on recent or projected emissions, the distribution of the national total to installations could have been done on another basis such as capacity or past activity levels. This was done in a few instances, as noted in the section on benchmarking, but the nearly universal pattern was to allocate to installations on the basis of their share of baseline emissions. Where sector emissions were projected to increase, this could mean an allocation larger than baseline emissions, but it could as easily imply a smaller allocation often due to the subtraction of certain quantities from the national total for new entrants or special bonus provisions, such as for central heat and power, early action, or auctioning. Basing the micro-level distribution on emissions was also dictated by data limitations, especially in the industrial sector where output or input data was either not available or not easy to collect due to the heterogeneity of output. More generally, emission shares had the merit of recognising the heterogeneity of emissions sources whether the causes were differing products or earlier investments that had not anticipated a price being imposed on CO_2 emissions.

While recent emissions constituted the reference point for allocations; the baseline used to define recent emissions was not uniform among Member States. Data limitations dictated some choices, but variations in the definition of the baseline usually reflected other factors. Virtually all Member States chose an annual average of a multi-year baseline to avoid the idiosyncracies of any single year; however, that multi-year baseline ranged from three years in Germany and Spain to six years in the UK. Another variation was to adopt a drop-minimum rule, such as in the UK, to allow firms to eliminate an unrepresentative year from the baseline. In Spain, Denmark, or the Czech Republic, the standard baseline average could be set aside if the most recent year (or average of the most recent two years in Spain) were greater than the standard baseline, and the more recent year(s) used as the allocation baseline for that installation. The UK also adopted a set of Baseline, Commissioning, and Rationalisation Rules which had the effect of allowing further adjustments of the baseline for installations to reflect conditions that would more accurately reflect recent emissions. The end result of all these variations is that the baseline shares do not represent the actual shares of any single recent year or years, but shares of an artificial baseline consisting of what could be considered an appropriate average level of recent emissions. The extreme version of such an artificial baseline occurred in Hungary where the baseline was negotiated individually with all large emitters and with groups of small emitters to create an aggregate that comprised each installation's negotiated level of recent CO_2 emissions.

4.3 Shortage was allocated to electricity generation

Most of the Member States constituting the EU15 adopted a total allocation for the trading sectors that is less than predicted BAU emissions, although often slightly more than recent emissions. This total implied an expected shortage that had to be allocated somehow. One approach would have been to distribute the shortage equally among all sectors and sources. Instead a clear distinction was usually made between electricity generation and industrial sources and the shortage was allocated to the electricity sector. This approach was adopted by the UK, which was the first Member State to publish a draft NAP in January 2004 and many subsequent NAPs made the same choice when confronted with a expected shortage.

The reason for allocating the shortage to the electricity is two-fold. First, electricity generation did not face international, non-EU competition as did the products of many of the industrial installations. All governments were in a quandary concerning their climate change policy commitment and the feared competitive effects of that commitment. Second, power plants are commonly believed to have the ability to abate emissions at less cost than others, typically by switching to natural gas instead of the continued use of coal.

The main exceptions to the allocation of the shortage to electricity generation are Italy and Germany. Italy has little coal-fired generation so that the ability to switch to gas is limited and an expected significant turn-over of the electricity generating plant was a prominent feature of the preparation of the Italian NAP. In Germany, the distinction took another form, but the rationale was similar. Here the distinction was between combustion and process emissions.

Since electricity generation is entirely combustion and process emissions arise only from industrial processes, the difference concerns industrial combustion. In the German NAP, process emissions were given a preferential allocation equal to baseline emissions, while industrial combustion sources were treated like electricity generation in receiving the same compliance factor, or ratio of allowances to baseline emissions.

The general method of determining the shortage to be allocated to the electricity sector was: industrial sources would receive what they could reasonably be expected to need and the shortage would be allocated to the electricity sector by means of a uniform reduction from installation baselines. The method of determining what the industrial sectors would need was typically determined by sector-specific projections, but some countries, notably Denmark and Germany, gave industrial facilities (or process emissions in Germany) their historic baseline amounts without involving projections. Most other Member States allocated the remaining power sector allowances according to installation shares of recent sector emissions.

4.4 New entrant and closure rules are a common feature

Article 11(3) of the Emissions Trading Directive required Member States to take into account the need to provide access to allowances for new entrants, but it provided no specific guidance and did not direct free allocation. Nevertheless, all 25 Member States set up reserves to provide free allowances to new entrants and most require closed facilities to forfeit post-closure allowances. This feature is the more remarkable in that it is not found in other cap-and-trade systems (in the US; cf. Ellerman *et al.* 2003) where with few exceptions new entrants must purchase whatever allowances they may need and the owners of closed facilities are able to keep the allowances distributed to those facilities.

The motivation for these provisions is invariably explained by a desire not to be placed at a disadvantage in the competition for new investment and a complementary concern to avoid an incentive to shut down facilities in the Member State and to move production elsewhere. The argument concerning the closure provisions produced a particularly effective slogan in Germany where the absence of a closure rule was said to be equivalent to creating a 'shut-down premium.' In addition to these arguments based on employment concerns, comments are often heard that it doesn't seem fair to award allowances to incumbents and deny them to new entrants, or to continue the endowment of allowances to facilities where there is no ongoing need for them. New entrant provisions were also seen as required by pre-existing energy or industrial policy.

A number of observers have remarked on the distortionary effects of these provisions either as a subsidy to production or as biasing technology choices in a more CO_2 emitting manner. As noted in a number of the contributions appearing in Ellerman, Buchner and Carraro (2007), officials in the Member States and at the European Commission were well aware of these effects; but they were unable to resist the political demands that such provisions be included in NAPs. These political demands did not come from incumbents who favored retention of allowances upon closure and who, by definition, did not represent new entrants. A good example of the political importance of these provisions is provided by Ireland where one of the few technical recommendations over-ridden by the government was the one recommending against the provision of new entrant endowments and the forfeiture of allowances upon closure of a facility.

While new entrant provisions are common, their specific characteristics are not. The reserves established to provide free allowances to new entrants vary greatly in size. Among the ten countries (see footnote 3), new entrant reserves range from 6.5% of the national total in the UK and Italy to as little as 0.5% in Germany and Poland. Distribution is generally by a 'first-come-first serve' rule, but countries vary according to what happens if the reserve is exhausted. For most, late-comers will have to resort to the market but Italy and some other countries have stated that the government will purchase allowances on the market to provide for all new entrants. Provisions also differ if the new entrant reserve is not fully used. Most have stated an intention to sell the unclaimed surplus either by auction or on the market, generally in 2007, but Germany and Spain will annul remaining allowances. The criteria for determining the number of allowances to award to new entrants also differ considerably. As already noted, all employ some variation of a benchmark based on some definition of best practice or technology multiplied by expected production or by new capacity; however, these benchmarks can differ by fuel or technology used, especially in the electricity sector. For instance, the UK, Denmark, and Spain use a common benchmark for all fuels, while most other countries, notably Germany and Italy, differentiate by fuel. Among the ten countries, Sweden is unusual in allowing closed facilities to keep their allocations, at least for the rest of the allocation period, and Sweden also restricts access to the new entrant reserve to industrial and district heating facilities, implicitly excluding fossil-fuel fired generating units (of which none are planned to be built).

A final variation that is worth noting in this respect is the transfer rule that was pioneered by Germany and adopted by some other Member States. A transfer rule allows the owner of a closed facility to transfer the allowances of the closed facility to a new facility, which is thereby not eligible for allowances from the New Entrant Reserve. Transfer rules can be very complex. In Germany, for instance, the new facility must be put in operation within eighteen months before or after the closing of an existing facility. The transferred allowances are good for four years after which time the new facility receives allowances from the New Entrant Reserve. Also, the transfer is proportionate to the productive capacity of the closed facility. The key point in all of these transfer rules is that they operate only for new facilities within the Member State. Thus, while the 'shut-down premium' was not avoided for specific facilities, the social costs thereby incurred were offset by the benefits of the compensating new investment in the same Member State.

Given the wide-spread inclusion of special rules for new entrants, the general objective of avoiding investment disincentives through the allocation process was not accomplished. An endowment of allowances to new entrants reduces the cost of an investment and if it varies among Member States, a potential further distortion to the common economic market is introduced. The resulting differences have led to a lot of dispute (primarily from academic quarters), emphasising that harmonisation would seem especially appropriate for new facilities. Harmonisation could be accomplished by introducing harmonised provisions, a unified central EU new entrant reserve, or by prohibiting any new entrant allocations, in which case all new entrants would have to buy allowances in the unified market. The main argument for harmonising new entrant provisions is to avoid adding to the already existing differences in investment incentives across Member States for CO_2 emitting facilities, and the corresponding case for harmonised conditions is strong. Moreover, if an eventual harmonisation of all allocations, to existing and new installations alike, is desired, harmonising new entrant provisions is a first practicable step.

4.5 Auctioning is little used

One of the most striking features of the EU ETS allocation process is the extent to which auctioning was not chosen, despite the option provided by the Emissions Trading Directive to auction up to 5% of the Member State's total.[10] Only four Member States (Denmark, Ireland, Hungary, and Lithuania) decided to set aside any explicit amount for auctioning and only Denmark opted for the full five percent.[11] The total amount to be auctioned is 8.4 million tons annually out of a total EU ETS allocation of about 2.2 billion tons, or 0.13% of the total. This figure might be augmented by any unclaimed allowances in new entrant reserves.

Auctioning was strongly opposed by the owners of existing facilities in almost all Member States since the amount set aside meant fewer allowances for incumbents. The motivation for auctioning among the Member States choosing to auction varied. In Ireland, the motivation is explicitly budgetary: proceeds will fund the agency set up to administer the trading system. In both Denmark and Hungary, auction proceeds go to the general treasury. In Hungary, the finance minister was active in promoting auctioning but the initially proposed 5% was cut back to 2.5% under pressure from incumbents. Denmark is unusual also in that the electricity industry favors full or at least harmonised auctioning throughout the EU ETS in subsequent periods in order to reduce the competitive disadvantage the industry believes it suffers when allowances are distributed gratis. The reasoning is that the BSA obligation assumed by Denmark (-21% from 1990 emissions) implies fewer allowances per unit of output for Danish firms than for competing generators in other Member States. If allowances were auctioned, the differences in the distribution of free allowances and the consequent endowment effects would be wiped out.

5 Concluding remarks

5.1 A more general principle?

Entwined with these ten lessons and unifying themes are two more general issues that are raised by the European experience. To what extent is the allocation of emission rights for CO_2 different from that for other conventional pollutants, such as SO_2 or NO_x? Is there some more general consideration that influences the choices when emission rights are distributed? Or, to rephrase this second question more negatively, why are the many welfare-enhancing choices universally advocated by economists not chosen?

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[10] However, reviewing the experience with three main applications of tradable-permit systems—air-pollution control, water supply, and fisheries management—as well as some unique related programmes, Tietenberg (2003) also finds that 'grandfathered' rules tend to predominate despite an infinite number of possible distribution rules.

[11] The percentages for Hungary, Ireland, and Lithuania are 2.5%, 0.75% and 1.5%.

There are many ways in which CO_2 is different from SO_2 and NO_x, but the one that seems to matter for allocation is the perceived potential for abatement. Put simply, the perception is that, with few exceptions, CO_2 emissions cannot be reduced in relation to production other than by carbon capture and storage (CC&S), which is available only at costs that are higher than any society is now willing to bear. At lower costs, this perception maintains that the only way of reducing CO_2 emissions is to reduce output. This is a very different view of abatement potential from what characterises SO_2 and NO_x. In both cases, deep reduction technologies, achieving 90% or better removal efficiency are technically demonstrated and available at costs that do not imply significantly higher product prices. In addition, less expensive abatement methods effecting smaller reductions are available, such as switching to lower sulfur fuels or low NO_x firing. With this panoply of abatement options, the effect on production is expected to be slight and the only question is which options will be used to effect the significant emission reductions (>50%) mandated by the SO_2 or NO_x caps. For CO_2, the perception is quite different and the result is manifest in the unwillingness to adopt more than a gently constraining cap initially, to furnish most participants with as many allowances as they will likely need in order not to curtail production, and to allocate the modest shortage to the one sector, electricity, where some abatement is acknowledged as possible by switching from coal to natural gas.

Whether this perception of the abatement potential for CO_2 is correct is not the issue. There is much to suggest that it is not. If there is one lesson from the US experience with cap-and-trade systems, it is that unexpected forms of abatement appear when a price is imposed on emissions (cf. Ellerman *et al.* 2003). Moreover, it is commonly asserted in Europe that further gains in energy efficiency can be achieved a relatively low cost, in which case equal reductions in CO_2 emissions logically follow at low prices. While it is too early to know the degree of abatement that has occurred in Europe in response to the CO_2 prices, the question is whether the perception of limited abatement potential was accepted sufficiently to be politically important in determining allocation choices. That appears to have been the case.

There is hardly an economist who does not deplore the limited use of auctioning and the concomitant extensive use of free allocation in the EU ETS (as well as in other cap-and-trade systems)[12]. The choice in the EU ETS is the more puzzling in that the economic arguments

[12] Among others, Schmalensee *et al.* (2003) conclude that allowance auctions in the context of the US Sulfur Dioxide Emissions Trading Scheme seem to have facilitated both the price discovery process and the

for auctioning (i.e., the so-called ‘double dividend’) are highly applicable to the European Union. In brief, if allowances are auctioned, the revenues received can be used to reduce taxes on labor and capital thereby reducing the disincentive effect of such taxes on the supply of labor and capital, which would then result in greater output than would otherwise occur (and thus less loss of welfare from the environmental measure). Such ‘recycling’ of auction revenues would seem to have great appeal anywhere, but particularly in Europe where high social charges on labor are commonly seen as the cause of persistently high unemployment levels. That auctioning was so little chosen suggests that more is involved than arguments based on economic welfare.

The usual explanation is lobbying or a version of public choice theory whereby some perversion has entered the system. Yet this is a strangely incomplete explanation. While lobbying is no doubt present, the distinction between this form of advocacy in what are demonstrably democratic systems and other forms of pleading that are considered legitimate is never clearly made. In the case of the EU ETS, it must also be remembered that the decision to allocate at least 95% of the EUAs for free was taken by duly constituted political authorities, namely, the European Council of Ministers and the European Parliament. At the Member State level, industry was by necessity heavily involved in the process of allocation, but the government role is never described as one of awarding allowances to the highest bidder. As noted above, the government role was largely one of managing a process whereby competing claims could be reconciled, being the final arbiter, and imposing some top-down discipline on the process. Notably, concerns for fairness are at least as prominent as demands for more. Finally, lobbying notably fails to explain the phenomenon of the new entrant and closure provisions. Incumbents favored retaining allowances if facilities are closed and they generally did not advocate new entrant reserves, which implied fewer allowances for incumbents.

A different and perhaps broader perspective comes from political science where the argument is made that these distributions of private rights in public resources, whether air, grazing land, fisheries, etc., express social norms that often grant prior use a strong claim (Raymond 2003). The argument starts with the dual recognition that the rights to emit now being limited were previously freely exercised and that there will be continuing use after the

development of the allowance market. Cramton and Kerr (2002) provide an analysis and discussion of the positive characteristics related to auctioning.

constraint has been imposed. The question very quickly becomes whether the entitlement to the continuing rights should have any relation to the exercise of the implicit rights that existed prior to the constraint. A prior use norm implies that a strong relationship between the two is legitimate and appropriate.

This relationship is made stronger by difference in the regulatory obligation imposed on firms by the cap-and-trade form of regulation. In conventional, prescriptive environmental regulation, often pejoratively termed ‘command-and-control’, continuing use incurs no charge so long as the installation is deemed in compliance with the relevant ‘command’. In contrast, market-based approaches such as the EU ETS impose a charge on continuing use while also explicitly recognising some aggregate level of continuing emissions as allowable. Since those who freely exercised the implicit right before the constraint and those who will exercise the continuing right afterwards are very largely the same, imposition of the charge without some offsetting mechanism implies a drastic redistribution of those rights and one that would not occur with more conventional means of regulation. The simple way to solve the dilemma is to offset the liability imposed on continuing emissions with an endowment of assets conveying the rights to the newly created scarcity rent. Thus, the free allocation to regulated entities becomes the means by which the new form of liability is imposed. The compensating endowment may or may not be fully compensating, but whatever the balance, the incentive is clearly created to reduce emissions to the new aggregate limit as cost effectively as possible.

In this perspective, the newly regulated emissions are not so much a ‘bad’ as a heretofore fully authorised by-product of useful economic activity which is expected (and indeed fervently hoped) to continue. This view also helps explain the new entrant and closure provisions, which are not implied by a strict application of a prior use norm, and it suggests a modification of that norm. If useful economic activity is desired and CO_2 emissions are regarded as being to some extent unavoidable, then it is ongoing production that conveys a right to emit. The amounts provided to installations may differ according to vintage or industry, but the basic principle remains. This too seems to be a distinctive feature of CO_2 in contrast to SO_2 and NO_x, as well as other public resources, where new entrants are generally obliged to buy rights from those who implicitly exercised them prior to the imposition of the constraint.

5.2 Global implications

Notwithstanding the increasingly common and convenient reference to the European Union as a single entity, it is well to remember that the Union consists of twenty-five sovereign states, all of whom jealously guard national prerogative. Accordingly, it is appropriate to look at the EU ETS not just as an unusual example of common undertaking within the European Union, but also as an exercise in implementing a multi-national climate change regime in which not all participants are equally committed to taking effective action to restrict CO_2 emissions.

It is not surprising that EU Member States that are highly committed to meaningful action on climate change within the ambit of the Kyoto Protocol, such as Germany, the UK, Sweden, Denmark, and the Netherlands, should develop an international trading scheme, but the same cannot be said for a number of the other Member States. Eight of the ten accession states have national limits under the Kyoto Protocol, but for seven of these states those limits are slack so that nothing is required of them over the 2005-12 horizon by the Kyoto Protocol. Moreover, two of the accession states, Cyprus and Malta, are not Annex I countries and therefore do not have any caps under the Kyoto Protocol. Yet all of these states have adopted national caps and are participating in this multilateral trading scheme. The three contributions concerning Poland, Hungary, and the Czech Republic make it clear that none regard the rules as especially appropriate for their circumstances, and yet they belong.

A more interesting case is presented by Italy and Spain. These countries participated fully in the European Burden-Sharing Agreement and accepted targets that are in both cases considerably below likely BAU emissions. No serious measures appear to have been contemplated to ensure compliance with either the BSA targets apparently in the belief that the targets were aspirational goals to be used to justify measures that would be taken anyway, but certainly not the basis for imposing a significant price on a significant share of national emissions. In these countries, the EU ETS was not easily accepted, but in the end it was.

Given the highly different circumstances of the EU Member States and their equally varying commitments to adopting meaningful measures to restrict CO_2 emissions, one must ask: What caused the reluctant followers to adopt caps and to enter into the multinational trading regime? The short answer is that participation was a requirement of membership in the

European Union.[13] But this easy answer relying of legal formalities evades the more serious issue of what caused highly sovereign nations to accept a measure that they did not seek, would not otherwise have chosen, and that involved costs that they would have preferred to avoid. The answer would appear to be the broader benefits of participation in the European Union—not so much the European identity, however that may be defined, but the more concrete benefits of freer trade, access to larger markets, freer movement of labor and capital, and of other benefits (including aid) that come with becoming part of a broader community. As stated in the contribution concerning Hungary, accepting the EU ETS was 'just another obligation on the long march to the EU.' In Poland, there was strong industry opposition to accepting the European Commission's reduction of the total, but the Polish government was unwilling to challenge the Commission because of these broader interests. In these instances and others, the European idea served as the glue that both attracted reluctant participants and fastened them to undertakings that they would not otherwise have accepted. The European idea will not serve beyond Europe, but some similar combination of desirable community and practical advantage will need to be found.

The importance of the EU ETS extends well beyond its usefulness as a laboratory in which twenty-five experiments in the allocation of carbon rights can be studied or as an example of a multi-national endeavor that has successfully navigated the shoals of differing circumstance and motivation. It has cast the die concerning the nature of a future global climate regime if there is to be one. Europe's choice of emissions trading has created a fact on the ground that will be as difficult to ignore in the future as it is to imagine an effective global regime without the United States.

This influence on the nature of a future global climate regime is easiest to imagine in the event of failure of the EU ETS. Had the indispensable first step of allocation descended into a cacophony of conflicting interests and political mayhem, CO_2 trading as a national, not to mention global instrument of climate policy, would have been badly set back, perhaps irretrievably. The success of the European CO_2 trading experiment cannot yet be taken for granted, but the grounds for optimism are much greater than they were some years ago.[14]

[13] The issue of voluntary participation was actually discussed in various shades during the negotiations of the Emissions Trading Directive, and could be settled only through the introduction of the time-limited opt-out clause by installation, being subject to stringent conditions as well as Commission scrutiny.

[14] 'We believe that…having a Community-wide trading scheme in place by 2005 is a low-probability scenario.' (PointCarbon 2001).

Assuming that the whole experiment succeeds, and not just this first step of allocating carbon rights, the EU ETS will set the standard for a global regime and provide an unexpected but propitious example that others seeking effective measures to limit GHG emissions will find increasingly hard to resist.

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(lxxviii) This paper was presented at the Second International Conference on "Tourism and Sustainable Economic Development - Macro and Micro Economic Issues" jointly organised by CRENoS (Università di Cagliari and Sassari, Italy) and Fondazione Eni Enrico Mattei, Italy, and supported by the World Bank, Chia, Italy, 16-17 September 2005.
(lxxix) This paper was presented at the International Workshop on "Economic Theory and Experimental Economics" jointly organised by SET (Center for advanced Studies in Economic Theory, University of Milano-Bicocca) and Fondazione Eni Enrico Mattei, Italy, Milan, 20-23 November 2005. The Workshop was co-sponsored by CISEPS (Center for Interdisciplinary Studies in Economics and Social Sciences, University of Milan-Bicocca).
(lxxx) This paper was presented at the First EURODIV Conference "Understanding diversity: Mapping and measuring", held in Milan on 26-27 January 2006 and supported by the Marie Curie Series of Conferences "Cultural Diversity in Europe: a Series of Conferences.

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