Decentralization in the EU Emissions Trading Scheme and Lessons for Global Policy

Joseph Kruger, Wallace E. Oates, and William A. Pizer*

Introduction

The European Union (EU) has created not only the largest emissions trading scheme in the world, but also a system that has an innovative structure with some intriguing and problematic properties. In particular, the EU Emissions Trading Scheme (EU ETS) has its own decentralized character. In thinking about the decentralization issue, we can envision a range of design possibilities. At one extreme is a wholly centralized system in which the central environmental authority determines who will participate in the market, how many permits will be created, how these allowances will be distributed among the various emission sources (that is, installations being regulated), as well as all the rules for compliance and trading. At the other end of the spectrum is a completely decentralized system in which each country (or jurisdiction) runs its own system with no automatic links or connections to other jurisdictions. The EU ETS is midway along this spectrum, with the European Commission (EC) making certain basic decisions concerning the structure of, and participation in, the system, but member states deciding their national cap level, allocating the country's permits—also called allowances—to sources, creating institutions to monitor, report, and verify their emissions, and making choices about some structural features (such as auctions and banking). The EU ETS draws on the U.S. sulfur dioxide (SO_2) trading system for much of its inspiration, but relies much more heavily on decentralized decision-making for the allocation of emission allowances and for the monitoring and management of sources. Because the EU ETS links together emissions trading programs in so many separate countries, it also raises the broader issue of whether linking trading systems in different regions of the world in order to create a more global regime for trading carbon dioxide (CO_2) is ultimately feasible and desirable.

*The authors are Policy Director at the National Commission on Energy Policy, a professsor of economics at the University of Maryland and University Fellow at Resources for the Future, and Senior Fellow at Resources for the Future, respectively. Correspondence should be directed to pizer@rff.org. The authors acknowledge funding support from MISTRA, the Swedish Foundation for Strategic Environmental Research.

Review of Environmental Economics and Policy, volume 1, issue 1, winter 2007, pp. 112–133
doi: 10.1093/reep/rem009

Our purpose in this article is to examine the structure of the EU ETS, draw out its implications for the operation and particularly the efficiency of the EU ETS itself, and explore how the experience with the EU ETS might inform the design and functioning of any future global emissions trading system. We begin our analysis with a brief review of the structure of a centralized system of emissions trading, a description of the basic structure of the EU ETS, and then a discussion of the economic implications of decentralization and other key provisions of the EU ETS. Next, we discuss the basic analytics of linking emissions trading systems as well as the bigger question of whether linking is always desirable. We then turn to how these linking issues have played out in the context of the EU ETS. Finally, we look at the implications of the EU ETS for the establishment of a broader international emissions trading system.

The Decentralized Structure of the EU ETS

This section focuses on the decentralized structure of the EU ETS and its implications for economic efficiency. To provide a frame of reference or "benchmark" for our discussion, we begin with a description of the standard, more centralized emissions trading system and then point out the specific ways in which the EU ETS is different.

Standard Emissions Trading System: A Benchmark

In the conventional system of emissions trading, an environmental authority imposes a limit or "cap" on total emissions and then issues permits for units of emissions, where the total number of permits equals the cap. In establishing the permit market, the authority makes three fundamental decisions. First, it defines who will participate in the market: the sources of emissions that can buy and sell permits. This effectively determines demand in the permit market, since the demand curve is simply the horizontal summation of the demand curves of the individual sources. Second, when the authority specifies the cap, it sets the supply in the market because this determines the number of permits that will be available.[1] Third, it sets the market in motion by actually allocating the permits among the sources. This can be done in either of two fundamentally different ways: an auction of the permits or a free distribution of the permits among sources or other entities, often by a formula based on historic emissions or output. The authority may also employ some combination of these two methods.

The resulting market equilibrium will determine the actual pattern of emissions among the sources. A cost-minimizing source will purchase (or sell) permits until its marginal abatement cost (MAC) equals the market price of a permit. This implies that, in equilibrium, the MAC of all sources will be equal, which satisfies the necessary condition for minimizing the aggregate abatement cost of realizing the cap. In addition, in the absence of transactions costs, this outcome does not depend upon how the permits are initially allocated among sources, because the actual emissions of the various sources are independent of the initial distribution. Thus, in principle, the emissions trading system generates the least-cost pattern

[1]This does not include offsets—credits generated outside the trading system that can be used to meet allowance requirements. We return to this issue later.

of emissions among sources. The system also provides a continuing incentive for research and development (R&D) in abatement technology, since, by finding more effective and less costly ways to reduce emissions, sources can reduce the number of permits they need, thereby increasing their profits.

It is important to note one additional property of the "standard" system: the specific allocation of permits typically has no effect on the output prices of polluting firms. Under free distribution, initial allocations constitute a one-time transfer of wealth to the sources. Each source receives an asset whose value is equal to the number of permits in its allocation multiplied by the price of a permit. As noted above, a cost-minimizing source then buys or sells permits until its MAC equals the price of permits. So, in this general setting, the initial allocation of permits has no effect on either marginal abatement costs or the marginal cost of output.[2]

In addition to these fundamental decisions, there are a myriad of other choices facing the environmental authority as it creates an emissions market. Two choices are particularly relevant in discussions of decentralization and linking: those concerning flexibility mechanisms over time and compliance. While emissions trading minimizes costs among sources during a particular period of time, costs can still be low in one period and high in another. A banking mechanism allows sources collectively to reduce emissions below the overall cap when costs are low, accumulate excess allowances, and then use them to increase emissions above the overall cap during a future period when costs are high. Although shifting emissions over time has consequences for some pollutants, the exact timing of CO_2 emissions is of little importance in terms of their environmental impact. What matters is the accumulated volume of emissions over a longer period of time (years or even decades). It thus makes economic sense to provide flexibility in the timing of emissions.

While banking allows sources initially facing low costs to over-comply and save allowances for future periods of high costs, it does not address the possibility of high costs in the initial period followed by lower costs. One mechanism for dealing with this possibility is a "safety valve." A safety valve refers to a specified price for emissions that is well above expectations (known as a "trigger price") that comes into effect when the market price for permits threatens to become excessively high, signaling a time of inordinately high abatement costs. Sources have the option of simply paying the trigger price for any emissions in excess of their allowances, instead of bidding and raising the price of permits to yet higher levels. The trigger price thus effectively sets a ceiling on marginal abatement costs, thereby allowing sources to maintain their emissions during times when controlling emission is extraordinarily costly. The economics literature has long made a compelling case for such a device, and, as Newell and Pizer (2003) have shown, a sensible safety valve offers large potential savings in the climate change context (such a mechanism has been discussed in the context of actual policies in the United States and Canada; see Government of Canada, 2002; U.S. Senate

[2]This assumes a setting of competitive markets with marginal-cost pricing. In fact, markets in energy-generating sectors are often regulated markets, where the form of regulation effectively results in some kind of average-cost pricing. In such a regulated setting, the way in which permits are allocated matters for output and pricing decisions. Under an auction of permits, for example, the cost of permits tends to be passed through in the form of higher energy prices.

2005). Borrowing is another alternative mechanism, as well as options combining various borrowing and safety valve features (Doniger, Herzog, and Lashof 2006).

A final set of design choices involves compliance. An environmental authority must decide how sources will monitor, report, and verify their emissions in order to ensure that emissions do not exceed the cap. Such features are often taken for granted in the United States, with its strong legal traditions and watchdog organizations. These features create the foundation for trust, confidence, and fairness in a trading system, but they are less certain in countries with weaker institutions or with different traditions.

The U.S. system for trading SO_2 allowances is one example that ties together all the features of the standard emissions-trading model. This "cap-and-trade" system establishes a cap on overall national emissions for trading sources and a mechanism for allocating the permits that relies primarily on historic levels of heat input and an emissions performance standard. The program allows unrestricted banking of allowances and uses continuous emissions monitors (particularly coal-fired units) to ensure compliance. This has been a carefully studied "grand experiment" in environmental policy-making. As noted by Burtraw and Palmer (2004), it has been highly successful in achieving its goal of reducing emissions at relatively low cost; it has become a model for other cap-and-trade programs.[3]

The Basic Structure of the EU ETS[4]

The EU ETS differs in some important ways from the standard model. Most significantly, as noted above, it adopts a more decentralized structure of decision-making that leaves much of the authority for key decisions to the member states. This, as we shall see, has some important implications for the way the system operates and its expected efficiency.

The starting point for the EU system is an overall cap on total emissions from all sectors of the economy in all 25 member states that is equal to the EU commitment under the Kyoto Protocol. Given this overall cap, the central EU authority has specified the sectors of the economy—the "trading sectors"—that will initially participate in the EU ETS. This encompasses four broad sectors: iron and steel, certain mineral industries (including the cement industry), energy production (including electric power facilities and refining), and pulp and paper. It is estimated that this includes over 12,000 installations that account for about 46 percent of CO_2 emissions in the EU. Because half of EU emissions remain outside the trading program, the EU's Kyoto cap necessarily will be met by a combination of efforts by sources in the trading sectors and by controls on sources in the nontrading sectors.

Within the EU-wide Kyoto target, each member state has its own national emissions target as determined under the EU burden-sharing agreement, which defines each member state's emissions reduction obligation. Each country is required to develop a National Allocation Plan (NAP), which, among other matters, addresses the national emissions target in two steps. First, it allocates the country's total burden-sharing target between the trading

[3] For an excellent review of the experience with trading programs, see Stavins (2003). For a comprehensive treatment of the principles and practice of emissions trading, see the new edition of Tietenberg's classic work (2006). Stavins (2003) and Ellerman et al. (2000) discuss how U.S. trading programs have benefited from the temporal flexibility provided by banking provisions.

[4] For a more detailed description and assessment of the EU ETS, see Kruger and Pizer (2004). An EC Green Paper (2000) also provides some useful background on the design of the system

and nontrading sectors. Second, it specifies how the permits in the trading sector will be distributed among the individual sources. The decision about how much of the target to allocate to the trading sectors also determines residually the stringency of a country's emissions controls on its nontrading sectors.

The EU ETS is being introduced in phases. The first phase (2005–2007) is a kind of "warm-up" phase, during which there is an opportunity to develop experience with the program and see how it needs to be modified in later periods. The second phase (2008–2112) coincides with the period when the EU must meet its Kyoto commitment. The EU then envisions subsequent 5-year (or possibly longer) phases.

Economic Implications of Decentralization

To understand the essential character of the EU ETS and its economic implications, we must first determine the source of demand and supply in the allowance market. In our benchmark case, the central authority determines both demand and supply by specifying the participants in the market and setting the overall cap on emissions. Under the EU ETS, the central authority—the EC—still determines the demand for allowances by specifying the sectors that will participate in the market (although countries can opt to include additional sectors). By knowing who is in the market, we can directly determine, in principle, the demand curve for allowances in the EU. The aggregate demand curve for allowances is again simply the horizontal sum of the demand curves of all the sources in the trading sectors across the member states.

The determination of supply in the trading sector, however, differs from the benchmark case. The member states individually determine what fraction of their national emissions budget they will allocate to the trading sectors. Thus, each country is effectively creating a certain number of allowances, and the aggregate supply of allowances is the sum of these allocations over all the member states. This results in a rather curious system of tradable emissions permits in which the demand curve is centrally determined at the EU level, but the supply curve is determined jointly by the decisions of the member states.[5] A number of other typically centralized choices have also been decentralized in the EU ETS.

Trading vs. Nontrading Sectors

This structure of decentralized supply and centralized demand decisions has two implications that are worth noting. First, it is difficult for any member state to predict the market price of allowances as they set their own NAP, since one would have to know all the other NAPs in advance. For example, were the other member states to devote a relatively small share of their national allowances to the trading sector, the supply of allowances in the market would be comparatively low and their price high, regardless of what one particular member state chose to do. In turn, this uncertainty about the market price of allowances means that member states will have a hard time in efficiently balancing the level of effort between their trading sectors and nontrading sources. While each member state very clearly sets

[5]The EC does have some indirect control over supply through its approval of member state NAPs, as well as decisions about the use of offsets.

the level of effort required by nontrading sources (by default, the difference between their Kyoto target and their allocation to the trading sectors), the actual effort and emissions from a country's trading sector will depend upon its behavior in the allowance market (i.e., the number of allowances it buys and sells). This leads to the basic conclusion that allowing a decentralized division of allowances between the trading and nontrading sectors will not, in general, result in a cost-minimizing pattern of emissions between trading and nontrading sectors because member states do not possess the information necessary to predict the market price of allowances and set the nontrading sources' level of effort accordingly.

This uncertainty about the market price of allowances could be resolved by centralizing at the EU level the division between trading and nontrading sectors for each member state, leaving the member state to allocate its budget among the respective sources within its borders. This would create a system similar in spirit to the nitrogen oxide (NOx) trading program in the United States in which each participating state distributes its fixed NOx "budget" among the sources within that state (Burtraw and Evans 2004). Is this a better idea from an efficiency perspective? The answer is not clear: The centralized EU authority would need to know the costs of each member state's nontrading sources in order to divide efficiently the national target between trading and nontrading sectors in each country. Yet, it is unlikely that the centralized EU authority could assemble such information. This is really part of a more general problem that arises because complete information about costs in all countries and all sectors is unlikely to exist with a single decision-making authority. That makes it hard to reach any general conclusion as to whether it is best (from an economic efficiency perspective) to delegate the decision on dividing allowances between trading and nontrading sectors to the member states or to keep it centralized (Bohringer and Lange 2005). Specifically, what is required is a general equilibrium analysis to compare the efficiency properties of alternative regimes (Parry and Williams 1999).

Equity and Fairness

From a political–economy perspective, the rationale for letting member states make these allocation decisions seems to stem largely from the widespread concern in Europe about "competitiveness" and unfair subsidies being given to particular sources or groups of sources. But does it matter if sources in some countries receive fewer allowances than identical sources in other countries? Since the free distribution of allowances is a one-time wealth transfer to sources, it does raise concerns about equity and fairness. Although in principle the allocation of allowances to a source has no impact on its abatement or production decisions, it can affect its liquidity, including, for example, the need to resort to capital markets for funds. Moreover, there can be some impact on output and prices in regulated sectors. However, over the longer haul, these initial wealth transfers will diminish in significance and may have little impact on the profitability of sources. But regardless of the true impact, this issue looms large in European thinking. Indeed, the EC is required to review and approve each country's NAP, in part to ensure that there are no elements of unfair competition or subsidies (though, in practice, competitiveness does not appear to have played a large role in the EC's criticisms of Phase I NAPs).

Offset Provisions

The EU ETS also offers the possibility of effectively expanding the supply of allowances by obtaining offsets for some EU emissions from outside Europe. More specifically, the EU ETS allows the use of Joint Implementation (JI) and the Clean Development Mechanism (CDM), the same offset provisions available to member states under the Kyoto Protocol.[6] This holds out the possibility of equalizing marginal costs across trading and nontrading sources if the same pool of offsets feeds both demands. It already seems likely that offsets will play an important role for the nontrading sectors, as there is a widespread recognition that member states have been overly generous in their allocations of allowances to sources in their trading sectors. Without offsets, this generous allocation to the trading sectors implies that very stringent controls will be required on sources in the nontrading sectors if member states are to meet their Kyoto commitments. In fact, it may be that the only way the EU will be able to meet its Kyoto cap is by acquiring emissions offsets through the CDM or JI mechanisms, or through purchasing excess allowances from Russia or Ukraine. But a heavy reliance on offsets from sources outside Europe raises some serious political issues, especially since it is seen by many as being contrary to the spirit of the EU's leadership in the global effort to mitigate climate change.

Auctioning, Banking, and Compliance Provisions

In addition to decentralizing the choice of cap and allocation, the EU ETS gives member states a number of other important responsibilities. Although the bulk of allowances are to be distributed free of charge, there is a provision in the EU ETS that allows individual member states to auction up to 5 percent of their allowances in the first phase and up to 10 percent in the second phase. And, while there is a common requirement that banking be allowed after 2008, member states are free to choose whether and how to allow banking between the 2005–2007 period and the 2008–2012 period. Finally, member states are given considerable latitude to establish domestic compliance procedures, including monitoring, reporting and verification—procedures that are increasingly important as the EU is expanded to countries with weaker institutions and different traditions. Member states have no latitude to introduce something like a safety valve; instead, the EC provides stiff penalties for noncompliance.

The economics literature has made a strong efficiency argument in favor of banking (noted above) and auctioning rather than a free distribution of permits.[7] However, the potential role for both mechanisms in individual member states may be small, either because decentralization creates fears of placing one country's industries at a competitive

[6]A detailed discussion of the CDM's history, status and prospects is presented in Lecocq and Ambrosi (2007), which appears in this volume.

[7]There exists a large literature that explores the efficiency implications of different forms of market incentives in a setting with preexisting distortions in the economy. Parry, Williams, and Goulder (1999), for example, find that auctions of tradeable CO_2 permits promise significantly larger efficiency gains than programs which grandfather permits, largely because the revenues from the auctions can be used to reduce rates on existing distorting taxes. For a large collection of articles on this issue of "the double dividend," see Goulder (2002).

disadvantage relative to industry in other member states or because of more general distributional concerns. In the case of banking, a member state that allows banking will have a lower cap in the 2008–2012 period because any banked allowances must be subtracted from that country's 2008–2012 Kyoto target. As with an auction, this reduces the pool of free allowances, potentially harming that country's industry relative to industry in member states without auctions or banking. Thus the very freedom given to each member state with regard to auctions and banking may actually discourage their use. On the other hand, there may simply be pressure to maintain greater flexibility for free allocations, regardless of decentralization.

A Discussion of Linking Issues

With the growing number of emissions trading systems at national, regional, and even corporate levels, there has been increased interest in the feasibility of linking distinct programs. This section provides a brief overview of the implementation issues associated with linking issues, describes the basic analytics of linking, and discusses whether linking separate domestic emissions trading programs is always desirable.

Overview of Linking Issues

In theory, linking distinct emissions trading systems will increase efficiency by taking advantage of diverse marginal abatement costs of firms in the larger linked system. In practice, however, there are implementation challenges associated with linking. Several studies have examined the extent to which there is a need for consistency in some of the key design elements of trading programs in linked systems, particularly concerning monitoring and enforcement, allowance distribution, and target type and stringency (for example, Haites and Mullins 2001; Blythe and Bossi 2004; Baron and Philibert 2005 and Baron and Bygrave 2002). Overall, most studies have concluded that it is technically feasible to link different trading systems, but that reconciling differences in design elements may require additional administrative procedures, which may increase administrative costs and complexity.

There are also potential fairness issues raised by linking programs, and these may be more difficult to resolve. Haites and Mullins (2001) note that some participants in a linked system could be worse off than they are in separate systems. For example, when two programs with emission targets of different stringency are linked, prices in the combined program will be higher for one of the programs and lower for the other than they would have been without linking. Victor, House, and Joy (2005) and McKibbin et al. (1999) also note that there may be large capital flows associated with emissions trading when countries with different greenhouse gas obligations are linked. These transfers may be politically controversial if it is perceived that countries are shouldering different burdens for emissions reductions.

Basic Analytics of Linking

We turn now to a discussion of the basic analytics of linking to illustrate what actually happens when countries link their emissions trading programs. In particular, we look at the effects of linking when countries have different permit prices before linking or when there are differences in the basic architecture of their prelinking trading systems.[8]

Effects of Linking with Differences in Prelinking Prices

Let us start with the question of what happens when two countries with significant differences in their prelinking permit prices link their trading systems. The simple answer is that the price is equalized. Sources in the country with the higher permit price purchase permits from sources in the country where the price of permits is lower until the prices (and hence MACs) are equalized across the two countries with total emissions (i.e., the sum of emissions in the two countries) remaining the same.[9]

While this results in a reduction in overall abatement costs, it does not mean that everyone is better off (Haites and Mullins 2001). Specifically, buyers in the previously low-price country find that they must pay more for additional allowances, while sellers in the higher price country are paid less. But this is really no different from the basic process of opening up trade more generally. The introduction of international trade typically involves winners and losers as a result of the distributional effects from changing prices.

Linking in the Presence of a Safety Valve and/or Banking

Earlier we described two mechanisms for introducing some cost-saving flexibility into the temporal pattern of emission: a safety valve and banking. Banking tends to put a lower bound on prices in each period, as sources see future value in saved allowances and will therefore choose to hold on to them rather than selling them at an unusually low price or using them when relatively inexpensive abatement options remain available. In contrast, a safety valve defines an upper bound on prices in each period that is equal to the specified trigger price. We can see in Figure 1 the effects of these mechanisms on the supply of allowances. In the absence of these mechanisms, the supply of allowances is simply equal to the number of permits issued; in Figure 1, this is represented by a vertical line at the indicated cap level. In an autarkic (i.e., nonlinked) setting of a single country, the introduction of banking and a safety valve would tend to create a price floor at P_e (equal to the expected value of banked allowances) and a price ceiling at P_t, the trigger price (at which point the supply of allowances becomes infinitely elastic). Thus, the supply of allowances has the shape of a step function P_eABD.

Linking raises the intriguing question of what happens to the supply of allowances in the system as a whole when some countries have these mechanisms in place and others do not. Suppose, for example, that country B (without banking or a safety valve) links its trading system to that of country A (which has both banking and a safety valve). In this case, country A will effectively export its banking and safety valve mechanisms to the new consolidated

8Additional information on linking systems with different architectures can be found in Marcu and Pizer (2003).

9The assumption that total emissions remain the same with linking applies only to emissions covered by the trading program. Emissions outside the trading program, either in nontraded sectors within countries A and B, or in other countries, could change in ways that raise or lower global emissions.

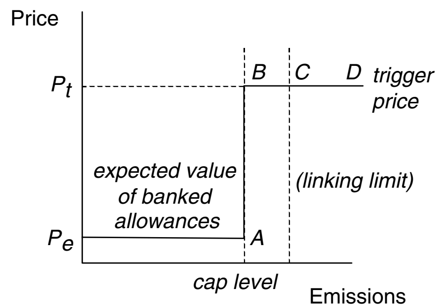

Figure 1. Effect of banking and a safety valve on supply of allowances.

permit market. If the common permit price tips above the trigger price in A, sources in A would pay the trigger price to the government of A for permits that they could then sell at a profit to sources in B. Conversely, if the common permit price dips below the expected value of banked allowances, sources in A will tend to buy permits from sources in B and bank them for future use.

However, it is important to recognize that there may be limits to the exporting of a safety valve, depending on its implementation. Assuming the firms in country A are given a choice between turning in an allowance or paying the trigger price, the exporting process must cease when sources in A have sold all their permits to sources in B and have nothing left to sell. This implies that the potential increase in the supply of allowances for the trading system outside of country A is limited to the supply of permits in country A. Now using Figure 1 to describe the shape of the supply schedule for the linked countries beyond A, the horizontal stretch of the supply curve of allowances at the trigger price can only equal the length of segment *BC* (the supply of allowances in A), at which point the supply curve again becomes vertical (the vertical line at point *C*). In the EU context, for example, this suggests that if Luxembourg were to have a safety valve, it would only have a very small effect on the potential supply of allowances in the EU ETS outside of Luxembourg (i.e., the horizontal segment BC in Figure 1 would be quite short). In contrast, if Germany were to have a safety valve, the increase in the EU supply of allowances at the trigger price would be quite substantial.

A further important consideration is how linking will affect overall emissions now and in the future. If linking changes the use of a safety valve mechanism, this will tend to have a lasting effect on overall emissions, with the direction and magnitude depending on how much the use of the safety valve increases or decreases. Any change will also affect the revenue accruing to the government of country A from payments of the trigger price. In contrast, changes in the use of a banking mechanism will affect current period emission but the change will be offset in a future period.

Linking with a Carbon Tax System

While the safety valve highlights one type of an explicitly price-based program, one might also consider the possibility of linking a completely price-based program, such as a carbon tax system, to a trading program. Suppose country A now has a carbon tax rather than a trading system with a safety valve. It is relatively easy to imagine country A stipulating that firms can present permits purchased from country B in lieu of a tax payment (perhaps offsets are already used in this way in country A). This will work to equalize marginal costs if country A initially has a higher price when linking occurs. Sources in A will buy permits from sources in B until MACs are equalized at the tax level in A. It is also possible for country A to adjust its tax in response to the price level in country B, with the consequence that the overall emissions level will change as country A changes its tax rate (such a proposal was put forward in New Zealand in 2002; see Government of New Zealand 2005). As in the safety valve case, linking will tend to change the revenue accruing to the government with the carbon tax (country A).

However, it is hard to imagine how linking could work to equalize prices if country A initially has a lower price than country B and does not adjust its tax rate. Since a carbon tax does not provide a currency to trade in, it is not clear how firms in country B could use the lower price in country A to any sort of advantage. This contrasts with the safety valve case, where the permits being sold by the government in country A could be exported to firms in country B and used to increase country B's effective cap.

Linking with Indexed Caps

Yet another possibility arises when country A indexes its cap to output and country B does not. Such an indexed system in country A might look like the proposed Canadian Large Final Emitters (LFE) program, where the cap is adjusted based on output growth. It might also look like the lead phase-down in the United States, where a regulation was written in terms of a rate standard (grams of lead per gallon of gasoline), but sources were allowed to trade. As Fischer (2003) points out, if output in country A is affected by the linking of the systems, this will (by definition) affect the aggregate cap level. Generally, linking raises output in both countries because trading increases efficiency and therefore the productive use of resources. This would tend to raise output and therefore the cap in country A. In some situations, however, where the price in country A is initially lower than in country B, the output level could fall and lower the cap in country A. Other than this effect on the aggregate cap, however, the story is similar to the simple linking story.

Whether or Not to Link

We turn now to the question of whether, in theory, it is always a good idea for countries to link emissions trading programs. The economic underpinnings of linking are seemingly unassailable: once targets are chosen in a given set of countries, trading enhances economic efficiency and reduces overall costs. However, we have already noted two possible weaknesses in this argument. First, there is a tacit, mutual acknowledgement of each others' targets in a linked system, an acknowledgement that might not be desired if one country sees another as having too weak a target. Because emissions targets inevitably will be revised over time, countries necessarily have to think about how their decisions now will affect other countries' decisions in the future. From this perspective, choosing to link one's trading program to another country's weaker regime will likely hinge on whether such action will further encourage a weaker regime or, arguably, give one greater leverage to extract more stringent targets in the future.

Second, although when viewed as a whole, trading leads to overall gains, certain groups do lose. Specifically, as emphasized above, buyers in the low-price country and sellers in the high-price country are made worse off when trading between the countries occurs. Like the case of expanding free trade in general, concerns about adverse impacts on particular groups can thwart progress even though the aggregate gains are well understood.

There is yet a third wrinkle in the argument for linking, and it has to do with the possible effects of linking on technology development. When we broaden the scope of discussion from the EU to the entire world, we have to confront the fact that current mitigation is not the goal. Most countries recognize that even as we reduce emissions now, an equally if not more important goal is to encourage technology development that will allow expanded mitigation opportunities in the future. For example, near-term mitigation consists entirely of energy conservation and fuel switching from coal to gas (given that current, zero-emissions sources produce at capacity). Over the medium term, additional, zero-emissions nuclear and renewable power generation can be constructed. But over the longer term, if we want to stabilize concentrations and therefore halt the flow of greenhouse gases into the atmosphere, all countries have to move towards zero-emissions power generation and transportation, a task unlikely to be met by the set of current, commercially available technologies.

Technology development can be approached in a variety of ways, including market mechanisms that put a price on emissions, but also through direct technology policies (Fischer and Newell 2004). In fact, because of dual market failures—the failure of firms to internalize the externality of pollution damage and the inability of innovators to capture all the rents from their inventions—we actually need two policies to achieve a first-best solution. Even without a first-best solution, however, countries are likely to use different approaches to the dual challenge of mitigation now and technology development for the future. Some countries may seek a high emissions price partly to spur technology development; other countries may prefer to pursue lower prices until more technologies are available, leaning more on direct technology incentives.

The bottom line is that a country seeking to maintain a high emissions price to encourage new technologies may not be happy about a linking mechanism that lowers that price. The problem in this case is that emissions trading is really serving two purposes—mitigation and providing incentives for technology development. Achieving one more cheaply at the expense of the other may not be desirable. These divergent goals are somewhat easy to reconcile within a region, such as the EU, but may be more difficult to reconcile across regions, such as between the United States and the EU, where the balance of mitigation and technology policy has differed. Recent proposals in the United States, for example, have emphasized a linked program where a trading system with a modest emissions price finances a more aggressive technology policy (NCEP 2004; U.S. Senate 2005).

The EU ETS Experience with Linking

This section discusses how the linking issues discussed in the previous section have played out in the specific case of the EU ETS. We examine how variations or inconsistencies in key design elements of member states' domestic programs, such as provisions for emissions monitoring, reporting and verification (MRV), allowance distribution, auctions and banking, and stringency of emissions targets have affected linking under the EU ETS. We also describe potential and actual steps being taken to address these issues.

Monitoring, Reporting, and Verification Provisions

The MRV provisions in the EU ETS give considerable flexibility to both installations and to member states. There are different "tiers" of methodologies that have different degrees of assumed accuracy. Firms propose installation-specific methodologies to the relevant authority in each member state. Installations are assumed to use the top tiers, but they may petition to use lower tiered methods with lower assumed accuracy if they show that a methodology is impractical or cannot be achieved at reasonable cost. Each member state has the autonomy to grant waivers from use of the top tier methods (European Commission 2004). Member State authorities may require companies to use third-party verifiers if the government does not have the capacity to verify hundreds of emissions reports, and each member state has the authority to set up its own certification procedure for qualified verifiers. It is worth noting that CO_2 monitoring guidelines for most sources in the EU system are relatively straightforward since emissions estimates are based on fuel use, rather than continuous emissions monitors (CEMs), as is done under the U.S. SO_2 and NO_x trading programs.

There has yet to be a thorough evaluation of the degree of consistency among the MRV approaches being used in individual member states. However, early analysis shows that there could be several important differences, including variation in inspection frequency and procedures, differences in overall enforcement rigor, and inconsistent application of the "tiers" or other aspects of the EC's monitoring guidance (Kruger and Engenhoffer 2006). In addition, there may be differences in the stringency of accreditation procedures for verifiers.

In the short run, member state discretion on the interpretation of monitoring guidelines and the certification of third-party verifiers may undermine some of the consistency that is necessary for an effective monitoring and compliance regime. On the other hand, there are a number of activities underway, both by member states as well as trade associations, to harmonize member state procedures and to encourage common accreditation standards for third-party verifiers (Kruger and Engenhofer 2006). For example, one effort would develop standardized electronic emissions reporting protocols (Kruijd 2006). Furthermore, initial variation in the application of these guidelines will likely diminish over time.

Perhaps more serious are the broader differences in legal systems, enforcement cultures, and administrative capabilities among the twenty-five member states. Will civil and criminal penalties be enforced consistently? Are adequate resources devoted to compliance and enforcement aspects of the program? An uneven approach to enforcement among member states could create unfair competitive advantages for firms in member states with weaker enforcement regimes. Clearly, there are no quick solutions to these issues, as they mirror the broader variation in regulatory institutions and practices throughout the EU.

Allowance Distribution Rules

Several analyses have documented significant differences in the allowance allocation methodologies of member states (Betz, Eichhammer, and Schleich 2004; Zetterberg et al.

2004; DEHSt 2005). For example, some studies have looked at the impact on innovation and investment incentives of different aspects of allocation rules (Mattes et al. 2005; Schleich and Betz 2005) and have found that these rules can affect technology choices and investment decisions. Ahman et al. (2006) and Betz, Eichhammer, and Schleich (2004) examine the impacts of different facility closure and new entrant policies. They find that when member states have policies that require confiscation of allowances after facilities close, they effectively create a subsidy for continued operation of older facilities and therefore a disincentive to build new, cleaner facilities.[10] They further show that different formulas for new entrants can create different incentives for investments across member states. Ellerman (forthcoming) notes that although these provisions increase output and reduce output prices, their impact on allowance prices is ambiguous.

There are a number of proposals to address some of the inconsistencies in allocation methods. For example, some have advocated "benchmarking," whereby allowances would be allocated according to performance standards that are based on emissions per unit of production (Mistra 2005). Of course, choosing EU-wide benchmarks would be extremely difficult and would raise a variety of equity issues. Others have advocated an increased use of auctions across the EU (Hepburn et al. 2006).

As noted earlier, member states may auction up to 5 percent of their allocations in Phase I and up to 10 percent in Phase II. In Phase I, only four EU member states (Denmark 5 percent, Hungary 2.5 percent, Ireland 0.75 percent and Lithuania 1.5 percent) chose to use an auction (Betz, Eichhammer, and Schleich 2004). There has been a modest increase in the number of member states who plan to auction permits in Phase II. An early analysis of 18 Phase II NAPs shows seven of the member states using an auction provision, with the percentage of allowances auctioned ranging from 0.5 percent in Ireland and the Flanders region of Belgium to 7 percent in the U.K. (Rogge, Schleich, and Betz 2006)[11] Neither Germany nor France, two of Europe's largest economies, has chosen to use an auction mechanism. However, as this article goes to press, there are reports that an auction is still under consideration in Germany (Point Carbon 2007).

In the short term, the highly political nature of allowance distribution and the diverse political pressures in different member states make it unlikely that there will be completely uniform allowance distribution methods across Europe. In fact, for the second phase of the EU ETS, the EC has largely avoided any attempt to harmonize sectoral allowance distribution methods, although it has encouraged simpler and more transparent methods (European Commission 2005). However, to the extent that certain allocation formulas are perceived as inequitable or providing subsidies for national industries, there may be an attempt by the EC to impose more uniformity in the long term.

[10] Stavins (2005) finds similar disincentives in vintage differentiated conventional regulations such as the U.S. New Source Review Program.

[11] The seven member states using an auction in Phase II are Ireland, the U.K., the Flanders region of Belgium, Lithuania, Luxembourg, Poland, and the Netherlands. In addition, Hungary has indicated in a draft NAP that it will auction allowances.

Banking

Prices for a linked system may also be influenced by decisions on banking in individual member states. Although the EU allows each member state to decide whether it will allow banking between the first two periods, most member states ruled out this option. This was largely to avoid building up a volume of banked allowances that would make it more difficult to meet the Kyoto target.[12] Only Poland and France included limited banking provisions in their Phase I NAPs. In theory, the ability to bank allowances through any member state's banking provisions could have had a significant impact on Phase I prices by letting prices rise to reflect future expected prices (Ellerman and Parsons 2006). However, it now appears that the French and Polish provisions will not have a significant impact on Phase I prices because they significantly restrict the number of allowances that may be banked. In November 2006, the EC issued a ruling that any banked Phase I allowances must be deducted entirely from a member state's trading sector NAP, rather than shared between the trading and nontrading sector obligations, further limiting these provisions.

Stringency of Emissions Targets

There has been little analysis of how domestic emissions targets vary in stringency. For example, we know of only one analysis (de Muizon 2006) that estimates the hypothetical autarkic allowance price for each member state based on its allocation to the trading sector. However, there have been analyses that compare targets selected by member states in various other ways. Baron and Philibert (2005) find large differences in volumes allocated to identical sectors across Europe, with allocated emissions for the electricity sector ranging from 30.9 percent above the baseline emissions period used for allocation in Finland to 21.5 percent below the baseline emissions period in the United Kingdom.[13] Others have found variations among member states in their overall reductions from business-as-usual (BAU) emissions. For example, Grubb, Azar, and Persson (2005) found that emissions targets proposed for the trading sectors in Phase 1 NAPs varied widely, with a few member states attempting absolute reductions, most proposing growth targets just below a hypothetical BAU projection, and a few with targets above BAU projections.

Such variations in targets among member states are not surprising, nor even necessarily undesirable. Member states face different marginal costs, both inside and outside the trading sectors, which could lead them to set different targets. Moreover, the EU ETS targets are required to reflect the overall national Kyoto (EU burden-sharing) targets, and these targets differ substantially by country in their relation to BAU projections.

Perhaps more problematic is that a lack of consistency in target setting assumptions and methodologies has made it difficult to evaluate the adequacy of targets. The emissions projections contained in member states' NAPs have been developed using a variety of techniques and assumption (Grubb and Neuhoff forthcoming). The EC can require

[12]There is no mechanism under the Kyoto Protocol to increase a country's 2008–2012 emissions target in exchange for early reductions prior to 2008, which is what national governments in the EU would be doing if they allowed banking.

[13]Note that allocation baseline periods may include different years in different member states.

member states to reduce their targets in the trading sector, but there has been a lack of reliable information for evaluating the stringency of a target. The EC has tried to address inconsistent assumptions and projection techniques in its revised guidance for Phase II NAPs (European Commission 2005).[14]

Uncertainty about the actual stringency of member state targets has also affected allowance prices, as became clear at the end of the first compliance year of the EU ETS, when the allowance market was surprised by lower than expected emissions and the resulting surpluses of allowances in many member states.[15] As individual countries began to report compliance results and verified emissions were lower than expected, the market price of allowances dropped dramatically. Ultimately, six countries, Austria, Ireland, Spain, the United Kingdom, Italy and Greece, had annual emissions greater than annual allocations.[16] Ellerman and Buchner (2006) note that although there was likely some over-allocation during Phase I of the EU ETS, even greater over-allocation was avoided by reductions in Phase I caps imposed by the European Commission on a number of member states. Without these cuts, the amount of over-allocation might have been almost twice as large.

The price volatility in the EU ETS also illustrates a fundamental risk of a linked emissions trading system. Put simply, decisions about the stringency of emissions targets in one country can affect the allowance prices faced in other countries. In addition to the sharp price drop that occurred when 2005 emissions were announced, there were other, though much less severe, examples of this phenomenon. For example, when Germany submitted its proposed NAP to the European Commission at the end of March 2004, allowances prices dropped. This was reportedly due to a larger-than-expected German allocation (Point Carbon 2004). Similarly, an announcement that the total allocation in the proposed German Phase II NAP would be reduced by 17 million tons led to an increase in the Phase II allowance price in November 2006 (Point Carbon 2006a). Pooling these risks across a larger market can, in general, be a good thing. However, an individual country's perspective depends on whether it perceives itself to be on the giving or receiving side of these events, and particularly whether events turn out to be more strategic than random.

The variation in stringency of member state targets has also had an impact on capital flows between firms in different member states and raised concerns about inequitable sharing of the burden of emissions reductions. Not surprisingly, some of the most negative reaction to the first year of the EU ETS has come from some sources in the United Kingdom, which had the most stringent target in the EU ETS and was a large net purchaser of emissions allowances. One study estimates that in 2005, U.K. companies had to purchase allowances from companies in other member states at a cost of £ 475M (Open Europe 2006). On the other hand, the ability to borrow within a compliance period (e.g., from 2006 to 2005) has attenuated some of these effects (Point Carbon 2006b). Thus far, the issue of capital

[14]Despite efforts to promote consistency, a recent analysis of Phase II NAPs finds that assumptions, data quality, and methodologies continue to differ in member state projections of BAU. See Rathmann et al. 2006.

[15]A plausible alternative explanation is offered by Ellerman and Buchner (2006), who note that lower than expected emissions may have been the result of greater than expected abatement.

[16]Note that estimates of allocations vs. reported emissions are preliminary and do not necessarily include allocations from new entrant reserves (see Ellerman and Buchner 2006).

flows between member states has not been a major issue in Europe, and the levels of capital flows involved have been small. However, the issue could grow in importance if there is a perception in subsequent phases of the EU ETS that variation in target stringency is accentuating winners and losers at the member state or firm level.

Ultimately, the European Commission may decide to exert more control over individual member state targets in order to enforce more uniformity of effort and to support a price that is consistent with EU-wide policy. In fact, as this article goes to press, the Commission is ordering even more aggressive cuts in member state caps for Phase II than it did for Phase I. More generally, concerns about inconsistencies between member state programs could lead to an increasingly stronger role for the Commission and a somewhat more centralized EU ETS.[17] However, it is unclear whether, in the long term, this method of reducing individual member state targets will be considered a transparent, effective and equitable approach for imposing the desired level of target stringency.

Implications of the EU ETS for Global Linking

The recent experience with the EU ETS provides a useful lens through which to look at the issue of establishing a broader international emissions trading regime. We would expect all of the EU ETS experience described above concerning heterogeneous MRV, allowance distribution, stringency of emissions targets, and architecture to apply also at the global level. However, at a global level, the variation in enforcement capacity, allocation choices, and stringency will be even greater than within the EU ETS, only without the cohesion and authority of the European Commission.

Variations in Institutional Capacity for MRV

The EU ETS experience is particularly relevant regarding divergent institutions for measuring, reporting, and verification. The issue faced by the EU ETS concerning differences in cultures of enforcement and administrative capacity among member states is only a fraction of what might exist if one were to compare the EU as a whole to, say, Russia or China. While this is less likely to be an issue among countries or regions such as the EU, Japan, and the United States, by most accounts, it is emissions trading with developing countries that is necessary to lower costs substantially. Studies of the Kyoto Protocol, for example, found the most dramatic effect in lowering costs to occur when trading was expanded to include developing countries (Weyant and Hill 1999). Concerns about institutional capacity have been raised for some time, even as international climate negotiations and scholarship have focused on the idea that developing countries and others with potentially weaker legal and economic institutions ought to embrace market mechanisms (Bell and Russell 2002). If it turns out that institutions in these countries cannot support emissions trading, it will be important to consider other, presumably less efficient, mechanisms to get at these cheap mitigation opportunities, including continued project-based approaches such

[17]The Commission is also considering a number of steps to make allowance distribution, monitoring and verification, and other aspects of program design more harmonized between member states after 2012. See European Commission (2006).

as the CDM, broader crediting programs for sectoral efforts or regulatory reform (such as efficiency standards or removing harmful subsidies), and government-to-government efforts on major energy deals.

Variation in Allowance Distribution

The variation in approaches to allowance allocation is also likely to be larger when we look outside the EU. Proposals in the United States, for example, have suggested larger auctions (RGGI 2005) and allocation to businesses at different points in the fossil-fuel chain (U.S. Senate 2005). However, these differences are probably less important in a discussion of linking and decentralization at the global level. Part of the reason allocation is high on the list of concerns within the EU is that the EU is a single market with fluid factors of production and movement of goods. The United States often faces the same pressures with regard to harmonization of environmental regulations and other rules across states (Oates 2002). At the global level, however, the existing differences among regulations and laws in different regions, as well as obstacles to the fluid movement of capital and labor, mean that there is likely to be less pressure concerning allowance allocation. Allocation concerns that figure prominently in the EU, such as new entrant allocations and effects on new plant locations, are simply dwarfed at the global level, where there are more major concerns regarding plant location across regions rather than within.

The single area where allocation might draw significant attention on the global level would be if such choices had an impact on the price of traded goods (and therefore existing producers in other countries). As noted earlier, allocation generally does not affect output prices unless output markets are regulated (e.g., electricity) or unless the allocation is updated in response to changes in output or input. If some countries choose to use updated allocations in sectors with tradable goods, other countries might see this as an unfair trade practice. However, this concern would exist irrespective of linking or coordination of climate policy.

Variation in Stringency and Architecture of Emissions Targets

Perhaps the most significant concern about the EU ETS that would be relevant at the global level is the variation in the levels of stringency and architecture of emissions targets in different regions. There are at least two reasons why there might be a problem with linking programs with different levels of stringency (Pizer 2006). First, as noted in our above discussion about the EU ETS, the flow of allowances among countries (e.g., into the UK), coupled with the existing perception that some countries have adopted more aggressive targets in their NAPs, has made some member states question the fairness of NAPs. Second, there is a concern that capital flows associated with large net sales of allowances across borders will itself be an adverse consequence.

Imagine, now, the much greater possible variation in stringency that is likely to exist at the global level. Developing countries, were they ever to create trading programs and join an international regime, would likely start out near their BAU levels (Frankel 2006). Such approaches are arguably fair, based on a variety of notions of national responsibility, but will they hold up in light of large trade flows? Perhaps of more concern would be programs in other industrialized countries, like the United States, that might be less stringent (measured by the allowance price prior to linking) relative to the EU. Would Europe open the EU ETS to a U.S. market where the net effect would be that Europeans pay U.S. companies for allowances whose relative abundance in the United States was a political decision? Would such a linkage change the dynamics for future target-setting in these countries? All of these concerns are only amplified when national programs use different architectures, such as indexed targets or a safety valve. It seems virtually impossible to imagine a country without a safety valve agreeing to link to a country with a safety valve with a low trigger price, as emissions will likely rise with a consequent increase in revenue to the government with the safety valve.

A related concern is whether the designers of national emissions trading programs have tried to set the price in their programs (explicitly or implicitly) to meet certain domestic needs and/or constraints, versus setting the cap without regard to price. Because linking programs means equalizing permit prices, the new price might not meet those needs or constraints. What might those needs or constraints be? One is the distribution of costs within a country. Even if linking makes a country better off as a whole, some may lose. Another issue arises if one country is seeking higher permit prices to drive technology development, and linking lowers the price.

Conclusions

The EU ETS essentially links the domestic trading programs of 25 countries, which, although subject to some common standards and oversight from the EC, still have considerable autonomy. This experiment with linked systems provides a useful laboratory for considering the political, economic and administrative challenges that would be faced by a global trading system, which will likely be even more decentralized than the EU ETS but with less oversight. An important, early observation is that linking makes it increasingly difficult to achieve an efficient balance of the burden across trading and nontrading sources (though access of both sources to a common offset market may alleviate some of this difficulty).

The current Kyoto Protocol has the appearance of something similar to the EU ETS, with provisions of the Protocol resulting in some degree of harmonization over national and (in the case of the EU) regional policies and the ability of governments to trade national commitments. For example, parties to the Kyoto Protocol have a centralized system for approving project-based credits in developing countries through the CDM Executive Board and common reporting standards.

Yet, looking at the policies pursued by different Kyoto signatories—namely the absence of any national trading programs outside of Europe—it is clear that the degree of supranational authority is much lower than in the EU ETS. Moreover, imagining a future regime that includes the United States suggests an international system with even less centralized authority than Kyoto.

In sum, the model of decentralization in the EU ETS has broken new ground in our experience with emissions trading regimes across multiple jurisdictions. It is providing new evidence on how different approaches to enforcement and monitoring, allocation, and even effort and stringency, can be encompassed in a single trading program. This experience is particularly valuable as we think about how a global regime might evolve.

The differences between the EU experience and the global context suggest that the challenges of a global system are likely to be even more formidable. On the enforcement and institution side, this suggests that broad-based emissions trading within developing countries may not be a realistic goal in the near term, and other avenues for engagement and trade need to be explored. Allocation is likely to be less of an issue, however, because the mobility of capital and labor is lower globally than within the EU. On the other hand, concerns over different choices about target stringency and effort are likely to loom large. Increasing efforts by the EC to tighten and effectively centralize member states' allocations has no global analogy. A global regime will also have to confront differences in architecture. The EU ETS has followed the traditional approach of an absolute cap. But other countries have discussed the possibility of price-based mechanisms and indexed caps, as well as a wide range of nonmarket mechanisms (e.g., standards and technology mandates). All of this may mean that in the short term, other national programs will not link to the EU ETS (assuming mandatory programs arise in other nations).

This raises the possibility of an alternative to linking: price harmonization. Countries could set their domestic policies in ways that recognize and respond to the efforts in other countries in an effort to harmonize marginal costs. New Zealand showed some interest in this approach when it proposed a carbon tax linked in value to the average permit price in the EU ETS. It is also somewhat remarkable that other domestic proposals suggest similar price levels. For example, the trading price in the EU ETS has been around €15 ($18) per ton of CO_2, while the proposed safety valve in Canada was C$15 ($13). In one U.S. proposal the projected price was $7, and the level of a proposed Japanese tax was ¥2500–3000 ($6–7). While not identical, these prices are much closer to each other than estimates of autarkic prices in response to the Kyoto Protocol. The experience in Europe, as well as signals about evolving policies in other countries, suggests that a natural tendency towards price harmonization may be inevitable.

References

Ahman, M., D. Burtraw, J. Kruger, and L. Zetterberg. 2006. A ten year rule to guide the allocation of EU emission allowances. *Energy Policy* 35: 1718–30.

Baron, R., and S. Bygrave. 2002. *Towards International Emissions Trading, Design Implications for Linkages*. Paris: OECD and IEA Information Paper.

Baron, R., and C. Philibert. 2005. *Act Locally, Trade Globally: Emissions Trading for Climate*. Paris: International Energy Agency.

Bell, R. G., and C. Russell. 2002. Environmental policy for developing countries. *Issues in Science and Technology* 18 no. 3: 63–70.

Betz, R., W. Eichhammer, and J. Schleich. 2004. Designing National Allocation Plans for EU emissions trading—A first analysis of the outcomes. *Energy & Environment* 15: 375–425.

Blythe, W., and M. Bossi. 2004. *Linking Non-EU Emissions Trading Schemes with the EU Emissions Trading Scheme*. Paris: OECD, COM/ENV/EPOC/IEA/SLT/(2004)6.

Bohringer, C., and A. Lange. 2005. Mission impossible!? On the harmonization of National Allocation Plans Under the EU trading directive. *Journal of Regulatory Economics* 27: 81–94.

Burtraw, D., and D. Evans. 2004. NOx emissions in the United States: a potpourri of policies. In *Choosing Environmental Policy: Comparing Instruments and Outcomes in the United States and Europe*, eds. W. Harrington, et al. 133–157. Washington, DC: Resources for the Future.

Burtraw, D., and K. Palmer. 2004. SO2 Cap-and-Trade program in the United States: A "Living Legend" of market effectiveness. In *Choosing Environmental Policy: Comparing Instruments and Outcomes in the United States and Europe*, ed. W. Harrington et al., 41–66. Washington, DC: Resources for the Future.

de Muizon, G. 2006. Articuler les marches de permis d'émission avec les autres instruments dans les politiques de lutte contre le changement climatique. PhD thesis, University of Paris Nanterre, Nanterre, France.

DEHSt. 2005. "Implementation of the Emissions Trading in the EU: National Allocation Plans of all EU States," German Emissions Trading Authority (DEHSt) at the Federal Environmental Agency (UBA), Berlin, November 2005.

Doniger, D. D., A. V. Herzog, and D. A. Lashof. 2006. Climate change: An ambitious, centrist approach to global warming legislation. *Science* 314: 764–65.

Ellerman, A. D. Forthcoming. New entrant and closure provisions: How do they distort? *Energy Journal*, Forthcoming.

Ellerman, A. D., and B. Buchner. 2006. Over-allocation or abatement? A preliminary analysis of the EU ETS based on the 2005 emissions data Fondazione Eni Enrico Mattei (FEEM), Nota Di Lavoro 139.2006, November.

Ellerman, A. D., and J. Parsons. 2006. Shortage, inter-period pricing, and banking. *Tendances Carbone* (July). www.powernext.fr/modules/PwnDl/download/files/eng/English_Tendances_carbone_n5.pdf.

Ellerman, A. D., P. Joskow, R. Schmalensee, J. P. Montero, and E. Bailey. 2000. *Markets for Clean Air, The U.S. Acid Rain Program*: Cambridge University Press, Cambridge, UK, European Commission, 2006. "Building a global carbon market—Report pursuant to Article 30 of Directive 2003/87/EC Brussels, COM(2006)676 final, 13.11.2006.

European Commission. 2004. Commission Decision establishing guidelines for the monitoring and reporting of greenhouse gas emissions pursuant to Directive 2003/87/EC of the European Parliament and of the Council, C(2004) 130 Final, Brussels, 29/01/2004.

European Commission. 2005. Further guidance on allocation plans for the 2008 to 2012 trading period of the EU Emissions Trading Scheme, COM (2005) 703 final, Brussels, 22.12.2005.

European Commission. 2006. Building a global carbon market, Report pursuant to Article 30 of directive 2003/87EC, COM (2006)~676 final, Brussels, 13.11.2006.

Fischer, C. 2003. Combining rate-based and cap-and-trade emissions policies. *Climate Policy* 3 S2: S89–S109.

Fischer, C. and R. Newell. 2004. *Environmental and Technology Policies for Climate Change and Renewable Energy*. Washington, DC: Resources for the Future.

Frankel, J. 2006. Formulas for quantitative emission targets. In *Architectures for Agreement: Addressing Global Climate Change in the Post Kyoto World*, ed. J. Aldy and R. Stavins. Cambridge: Cambridge University Press.

Goulder, L. 2002. *Environmental Policy Making in Economies with Prior Tax Distortions*. Cheltenham: Edward Elgar.

Government of Canada. 2002. Climate change plan for Canada, November, 2002. http://www.climatechange.gc.ca/plan_for_canada/plan/pdf/full_version.pdf.

Government of New Zealand. 2005. Review of climate change policies. http://www.climatechange.govt.nz/resources/reports/policy-review-05/policy-review-05.pdf.

Grubb, M., and K. Neuhoff. Forthcoming. Allocation and competitiveness in the EU Emissions Trading System. Policy Overview. *Climate Policy*, 6: 7–30.

Grubb, M., C. Azar, and U. M. Persson. 2005. Allowance allocation in the European emissions trading scheme: a commentary. *Climate Policy* 5: 127–136.

Haites, E., and F. Mullins. 2001. *Linking Domestic and Industry Greenhouse Gas Trading Systems*. Report for EPRI, IEA, and IETA. Toronto, Canada: Margaree Consultants, Inc.

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

Kruger, J., and C. Engenhofer. 2006. Confidence through compliance in emissions trading markets. *Sustainable Development Law and Policy* VI 2: 2–13.

Kruger, J., and W. Pizer. 2004. Greenhouse gas trading in Europe: The grand new policy experiment. *Environment* 46: 8–23.

Kruijd, J. 2006. Information technology enhances consistency, transparency, and efficiency in the EU ETS. Paper for emissions trading workshop on February 24, 2006, organized by INECE and IETA, PriceWaterhouseCoopers. http://inece.org/emissions/ITworkshop/IT workshop_backgroundpaper.pdf.

Lecocq, Frank, and P. Ambrosi. 2007. The Clean Development Mechanism: History, status, and prospects. *Review of Environmental Economics and Policy* 1 no. 1: 134–151.

Marcu, Andrei, and W. A. Pizer. 2003. Defining and trading emission targets. *Climate Policy* 3 S2: S3–S6.

Matthes, Felix, V. Graichen, and J. Repenning. 2005. The environmental effectiveness and economic efficiency of the European Union Emissions Trading Scheme: Structural aspects of allocation. Öko-Institut report to the World Wildlife Federation (WWF). http://assets.panda.org/downloads/okoinstitutetal_2005_euetsstructuralanalysisfinal_3.pdf.

McKibbin, W. J., M. Ross, R. Shackleton, and P. Wilcoxen. 1999. Emissions trading, capital flows and the Kyoto Protocol. *Energy Journal*, 20: Special Issue, 287–334.

Mistra's Climate Policy Research Program (Mistra). 2005. Allocation in the EU emissions trading system, lessons learned, a report from the clipore and ETIC Policy Forum Workshop in Gothenburg Sweden, June 17, 2005. Downloaded at: www.mistra-research.se/download/18.10c0ef21072c21c2ce80004362/.

National Commission on Energy Policy (NCEP). 2004. *Ending the Energy Stalemate: A Bipartisan Strategy to Meet America's Energy Challenges*. Washington: NCEP.

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

Oates, W. E. 2002. A reconsideration of environmental federalism. In *Recent Advances in Environmental Economics*, eds. J. List and A. de Zeeuw, 1–32. Cheltenham: Edward Elgar.

Open Europe. 2006. The high price of hot air: Why the EU Emissions Trading Scheme is an environmental and economic failure, Open Europe, accessed at http://www.openeurope.org.uk/research/ets.pdf.

Parry, I., and R. Williams. 1999. A second-best evaluation of eight policy instruments to reduce carbon emissions. *Resource and Energy Economics* 21: 347–73.

Parry, I., R. Williams, and L. Goulder. 1999. When can carbon abatement policies increase welfare? The fundamental role of distorted factor markets. *Journal of Environmental Economics and Management* 37: 52–84.

Pizer, W. A. 2006. The evolution of a global climate change agreement. *American Economic Review, Papers and Proceedings* 96 2: 26–30.

Point Carbon. 2004. Five of fifteen comply with NAP deadline. *Carbon Market Europe* 4 2: 3.

———. 2006a. *Carbon Market Daily* 2 233: 1.

———. 2006b. *Carbon Market Daily* 2 213: 2.

———. 2007. Germany accepts EC cap, auctioning likely. *Carbon Market Europe* 6, no. 6 (February 9): 6.

Rathmann, M, G. Reece, D. Phylipsen, and M. Voogt. 2006. Initial assessment of national allocation plans for phase II of the EU emissions trading scheme, Ecofys, November.

Regional Greenhouse Gas Initiative (RGGI). 2005. Memorandum of understanding, Accessed at http://www.rggi.org/docs/mou_12_20_05.pdf.

Rogge, K., J. Schleich, and R. Betz. 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 Institute Systems and Innovation Research, and Centre for Energy and Environmental Markets (CEEM) at the University of New South Wales (UNSW).

Schleich, J., and R. Betz. 2005. Incentives for energy efficiency and innovation in the European emission trading system. In *Proceedings of the 2005 ECEEE Summer Study—What Works and Who Delivers*? *Mandelieu*, France, 1495.

Stavins, R. N. 2003. Experience with market-based environmental policy instruments. In *Handbook of Environmental Economics*, ed. K. Mäler and J. Vincent, vol. I, chapter 9, 355–435. Amsterdam: Elsevier Science.

———. 2005. *The Effect of Vintage-Differentiated Environmental Regulations*, Discussion Paper 05–12. Washington, DC: Resources for the Future (March).

United States Senate. 2005. Climate and Economy Insurance Act of 2005. 109, S Amdt 868.

Victor, D. C., J. C. House, and S. Joy. 2005. A Madisonian approach to climate policy. *Science* 309: 1820–21.

Weyant, J. P., and J. Hill. 1999. The costs of the Kyoto Protocol: A multi-model evaluation, introduction and overview. *Energy Journal* 20 (Special Issue): vii–xliv.

Zetterberg, L., K. Nilsson, M. Ahman, A. S. Kumlin, and L. Birgirsdotter. 2004. Analysis of National Allocation Plans of the EU ETS. Stockholm, Sweden: IVL Swedish Environmental Research Institute Ltd., IVL Report B-1591.