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REVIEW OF MARKET-BASED INCENTIVES FOR CONSIDERATION OF APPLICATIONS IN CALIFORNIA

PIER FINAL REPORT

Prepared For:
California Energy Commission
Public Interest Energy Research Program

Prepared By:
University of California, Berkeley

March 2005
CEC-500-2005-025

California Climate Change Center
Report Series Number 2005-012

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Please cite this report as follows:

Farrell, Alexander E. February 2005. *Review of Market-based Incentives for Consideration of Applications in California*. University of California, Berkeley for the California Energy Commission, PIER Energy-Related Environmental Research. CEC-500-2005-12.

Preface

The Public Interest Energy Research (PIER) Program supports public interest energy research and development that will help improve the quality of life in California by bringing environmentally safe, affordable, and reliable energy services and products to the marketplace.

The PIER Program, managed by the California Energy Commission (Energy Commission), annually awards up to $62 million to conduct the most promising public interest energy research by partnering with Research, Development, and Demonstration (RD&D) organizations, including individuals, businesses, utilities, and public or private research institutions.

PIER funding efforts are focused on the following RD&D program areas:

- Buildings End-Use Energy Efficiency
- Energy-Related Environmental Research
- Energy Systems Integration
- Environmentally Preferred Advanced Generation
- Industrial/Agricultural/Water End-Use Energy Efficiency
- Renewable Energy Technologies

The California Climate Change Center (CCCC) is sponsored by the PIER program and coordinated by its Energy-Related Environmental Research area. The Center is managed by the California Energy Commission, Scripps Institution of Oceanography at the University of California at San Diego, and the University of California at Berkeley. The Scripps Institution of Oceanography conducts and administers research on climate change detection, analysis, and modeling; and the University of California at Berkeley conducts and administers research on economic analyses and policy issues. The Center also supports the Global Climate Change Grant Program, which offers competitive solicitations for climate research.

The California Climate Change Center Report Series details ongoing Center-sponsored research. As interim project results, these reports receive minimal editing, and the information contained in these reports may change; authors should be contacted for the most recent project results. By providing ready access to this timely research, the Center seeks to inform the public and expand dissemination of climate change information; thereby leveraging collaborative efforts and increasing the benefits of this research to California's citizens, environment, and economy.

The work described in this report was conducted under the Preliminary Economic Analyses of Climate Change Impacts and Adaption, and GHG Mitigation contract, contract number 500-02-004, WA MR-006, by Alexander E. Farrell, at the University of California, Berkeley.

For more information on the PIER Program, please visit the Energy Commission's Web site www.energy.ca.gov/pier/ or contract the Energy Commission at (916) 654-4628.

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Abstract

Review of Market-based Incentives for Consideration of Applications in California reviews the use of market-based instruments in environmental regulation, in order to understand how they might be used in California as part of a program to control greenhouse gas (GHG) emissions as part of the response to climate change. Both theoretical studies and specific examples are reviewed. The types of market-based instruments include subsidies, taxes, and several varieties of emission trading such as baseline and credit, cap-and-trade, and open market system. Emphasis is placed on emission trading, examples of which include the U.S. sulfur dioxide program, a multi-state program to control nitrogen oxides, California's RECLAIM program, and two national-level carbon dioxide (CO_2) programs from Europe. Key lessons for the possible implementation of emission trading in California include the scope and stringency of such a program, choices about if and how allowances will be auctioned or distributed, the need for an adequate monitoring and enforcement infrastructure, and how an emission trading program for GHGs would interact with other government policies, such as renewable portfolio standards and electricity industry regulation.

Keywords: emission trading, GHGs, climate change, California, cap-and-trade

1. Introduction

There is broad scientific consensus that rising concentrations of greenhouse gases (GHGs) in the atmosphere have already caused perceptible changes in climate and will lead to further climate change in the future (Intergovernmental Panel on Climate Change 2001). The impact of climate change in California may be significant for water resources, agriculture, and ecosystems (Shaw 2002; Roos 2003; Hayhoe et al. 2004). Avoiding the most serious climate change impacts will be challenging; it is estimated that deep (greater than 50%), near-term and sustained cuts in global GHG emissions will be required to stabilize atmospheric concentrations at a level that averts dangerous climate impacts (Wigley et al. 1996; O'Neill and Oppenheimer 2002). This will require significant changes from current practices, especially in the way that energy is consumed throughout the global economy and in the direction of innovative activity (Azar and Dowlatabadi 1999; Hoffert et al. 2002). Such changes will not come about without some form of government action, because avoiding dangerous climate change (like most environmental protection) is a public good (and so under-provided by markets), and because innovation designed to achieve public goods tends to require government action (Arrow et al. 1995; Norberg-Bohm 1999).

In response to these challenges, a variety of efforts are currently underway at the local, state, national, and international levels to attempt to mitigate GHG emissions and to stimulate the technological innovation that will be necessary to avoid dangerous climate change. Among these are the consideration of using market-based incentives (MBIs) to regulate GHG emissions, of which the main forms are a cap on emissions combined with emissions trading (cap-and-trade, C/T), or a tax on emissions.

The use of MBI in environmental policy has grown rapidly over the last several decades, because several features make MBI attractive relative to more traditional command and control (CAC) regulation (Stavins 2003). Most obviously, MBI can help reduce the cost of environmental protection, which they do by providing significant flexibility to regulated organizations (typically firms). Market-based incentives give firms incentives for environmental performance, rather than specifying emission rates or technical standards as CAC regulations do. The basic logic behind MBIs is that firms differ in how much it costs to control emissions, so if a high-cost company can pay a low-cost company to control emissions on its behalf, the net result is a savings. This approach assumes emissions from all sources are equivalent (i.e., uniform mixing); when this is not the case, additional rules can be added.

Importantly, the use of MBIs can make environmental protection look more like an ordinary business issue to managers and can allow them to apply risk management tools (often in the form of financial derivatives). In addition, MBIs can be (but are not always) easier to implement since government often does need to use intrusive inspections or detailed permitting processes that require considerable industry knowledge, which may be hard for government to obtain. However, careful (and expensive) monitoring may be needed, both at the plant level and in the marketplace to ensure MBIs perform as expected. Finally, MBIs may provide significant incentives for innovation, both managerially and technologically. However, the size and shape of these benefits vary greatly with the details of MBI design and implementation. This issue is discussed below, after the basic concepts are introduced.

An issue that sometimes arises with MBIs, including both taxes and C/T, is the possibility that the harmful effects of the emissions vary in time or space. In this case, when and where emissions occur matters. For instance, a ton of smog-forming emissions may have a greater impact in the summer than in the winter. However, many MBI programs treat all emissions the same, or place emissions into broad categories (e.g., summer and winter). Hence, it is more difficult to use MBIs to regulate flow pollutants (e.g., air pollution) than stock pollutants (e.g., GHGs). In this respect, GHGs are particularly well-suited to the market-based approach, because they are uniformly mixed and do not directly cause acute, short-term, or localized impacts. Thus, more flexibility can be afforded to the location and timing of emissions reductions, which makes it easier to implement MBIs.

This paper reviews key examples of the recent experience with emission trading in order to provide insights into how to apply this approach in California. The paper has four sections that follow this introduction. Section 2 presents the key concepts associated with emission trading. Section 3 presents some of the key results from analytic economic research on emission trading. Section 4 is the longest, it discusses seven examples of emissions trading programs. Section 5 discusses what lessons for California can be gained from both the analytic research and prior experience.

2. Concepts

Three essential types of MBIs can be identified: (1) direct taxes and subsidies, (2) baseline and credit systems, and (3) cap-and-trade systems. Each of these is discussed in turn, as well as a hybrid approach called *Open Market Trading.*

2.1 Taxes and Subsidies

The simplest MBI is a tax on emissions, which will give firms incentives to reduce emissions and to innovate to find cheaper ways to do so. Subsidies can work similarly, but, of course, give positive incentives rather than negative. By making polluters pay for the damage they produce, and thus internalizing externalities, emission taxes can increase economic efficiency. However, an emission tax generates a higher cost burden for firms than a CAC regulation that induces the same level of abatement. This is because, with the tax, the firm not only faces the cost of abatement but, in addition, it pays a charge on each not abated, unlike with CAC regulation. For this reason, emission taxes have been generally unpopular in the United States, although they have been used in several cases in Europe.

Taxes and subsidies are a very common energy policy, however (National Renewable Energy Laboratory 1995). For instance, a tax on gasoline funds a significant portion of highway construction. Subsidy programs have also been used to promote renewable energy. A recent study examined the use of state funds in eight states and found them to be fairly effective (Bolinger et al. 2004). In general, this study found that most states focused on utility-scale projects, setting aside or obligating about $344 million in construction or operational support for 163 projects totaling 2,288 MW. California has been the leader in this effort, spending about $193 million of that total for 1,285 MW. Wind has received most of this funding. Interestingly, this study finds that states are using innovative incentive-based policies to support these projects, including real-time production incentives (~85% of the total), price insurance, and debt financing. Outright grants make up only about 7% of the total.

However, subsidies may not always be a good idea. The national ethanol subsidy, for instance, appears to be quite inefficient (Duke and Kammen 1999).

2.2 Baseline and Credit Systems

The first type of emission trading system is based on facility-specific baselines and provides for the opportunity for facilities to operate above or below their baseline by using credits. Baseline and credit systems can be further differentiated into systems in which the credits are considered permanent (commonly called Emission Reduction Credits, or ERCs) or one-time (commonly called Discrete Emission Reduction, or DER, credits).

Emission Reduction Credits

In places with poor air quality and a CAC regulatory framework, new "stationary" sources (like a new power plant) or stationary sources undergoing expansions may be required to offset new emissions. The instruments used for this purpose are *Emission Reduction Credits* (ERCs), which rely on permanent reductions in emissions from specific sources compared to a baseline. These "project-based" approaches occur when a polluting facility goes beyond regulatory requirements in pollution control (overcontrol), permanently slows output, or shuts down. Emission credits are typically denominated in *rates* (e.g., tons per day) and are usually permanent, meaning that they allow for emissions at the given rate in perpetuity.[1] However, the firm trying to create credits must get government approval, which can be slow and raise transaction costs. Typically, ERC systems are add-ons to preexisting CAC regulations that provide flexibility and reduce costs.

In the United States, ERC systems have had limited success, but they are not used much elsewhere (Stavins 1997; Solomon and Gorman 1998). They have played a role in programs for federal air pollution control and are used by a few states. In California, the experience with ERCs has been mixed. The particularly difficult air quality challenges facing the state have led to more stringent rules governing ERCs for air pollutants than elsewhere, resulting in higher transaction costs and a less efficient outcome. One problem is obtaining agreement between regulators, environmental advocates, and industry about the size of emission reductions, because it is often impossible to measure emissions directly and because of disputes over appropriate historical baselines. Firms must prove that the ERCs are "surplus" or "additional"—that is, that the actions that create the ERCs would not have taken place anyway (Stavins 1997).

An important feature of ERC systems is that they are voluntary (although the underlying regulations are not) which, in practice, has meant that the incentives to create allowances and put them on the market have been weak (Hahn and Hester 1989b). Firms generally do not go out of their way to create ERCs, because they do not see themselves as being in the ERC business and prefer to invest capital in their core activities. Firms often want to keep ERCs to support possible expansions of their own facilities in the future rather than selling them into a market (where potential competitors might buy them). In addition, there is the concern that by voluntarily creating ERCs, they will provide regulators with additional information that emission control costs in that industry or process are lower than might be otherwise believed, leading regulators to mandate such emission reductions (Dudek and Palmisano 1988). Unfortunately, these problems can limit the number of available emission credits, making it hard for new entrants to buy credits to support new businesses. Additionally, the limited number of programs, the terms and conditions associated with ERCs, and the relatively small markets for them have hindered the development of risk management tools (e.g., futures and options) for them, limiting

[1] In this paper, *ton* refers to an English ton, or 2000 pounds. In cases where European values are given, ranges were originally provided, e.g., 2–4 metric tons. In these cases, the 10% difference between English and metric tons are ignored (e.g., the values are given as 2–4 tons, not 2.2–4.4 tons). The additional digits seemed to add complexity without much added value, especially because a range was given to start with.

their utility. One of the reactions to this problem has been for local governments to obtain ERCs as part of the process approving their creation, which can be used to foster growth (and job creation) by giving them to new companies entering the area.

Discrete Emission Reductions

Discrete Emission Reductions are created by reducing emissions relative to a baseline and are project-based. However, DERs are temporary and can only be used once. Discrete Emission Reductions can be generated by installation of pollution-control equipment, installation of control equipment with a higher-than-required efficiency or prior to a compliance date for additional control, or a process change. In the United States, several states have rules that allow DERs, but they have different mechanisms for granting credits. For example, some states certify DERs, while others allow for self-certification with third-party verification. A similar term, verified emission reductions (VERs), is sometimes used to describe one-time reductions in greenhouse gas (GHG) emissions, but their future is uncertain, because VERs are created *before* the rules for emission trading are in place.

2.3 Cap-And-Trade Systems (C/T)

The best-known MBI is the *emission allowance,* used in cap-and-trade (C/T) systems, which, as the name implies, create a permanent limit on emissions, a key virtue for environmental advocates. In a C/T system, the government defines the regulated sources and the total amount of pollution that can be emitted during a set period—the “cap.” Typically, the cap is set in mass units (e.g., tons), is lower than historical emissions, and declines over time. The government creates allowances equal to the size of the cap and then distributes them to the regulated sources—a process called *allocation.* All C/T existing systems allocate allowances based on historical emissions (i.e., grandfathering), which produces no revenues for the government and is problematic for new entrants, as discussed below.

The government then requires regulated facilities to surrender emission allowances equal to the emissions of the facilities on a periodic basis. It will also set standards for emissions monitoring and establish rules for how allowances may be used and enforcement measures. These are crucial choices, not just details. One particularly important decision is if and how allowances can be saved (or “banked”) from one period to another.

Because the allocation to each firm is smaller than its previous emissions, regulated firms have four basic options: (1) control emissions to exactly match their allocation; (2) undercontrol and buy allowances to cover their emissions; (3) overcontrol and then sell their excess; or (4) overcontrol and bank allowances for use in future years (when even fewer allowances will be allocated). The reason companies might buy or sell allowances is that facilities will have different emission control costs, or they might change operations so that they need more (or fewer) allowances. Firms with higher costs could save money by undercontrolling and buying allowances from those with lower costs, which could make money by overcontrolling and selling allowances.

Government regulates the trading of emissions allowances differently in various C/T systems. The government usually acts as the accountant for C/T systems by establishing a registry for participants. Usually, participants must report the size of transactions and the names of buyer and seller. This can be facilitated by creating a serial number for each allowance. However, there is often no requirement that market participants disclose the price at which a sale was made, nor any requirement that they inform government of the trade in a timely manner. This lack of information can limit the transparency of the market, as participants may delay reporting trades in order to conceal strategic information. Brokerage and consulting firms complete the picture by providing services to market participants, including small markets in derivative commodities, and by increasing transparency by providing information about the markets. Simplicity in market design and competition among brokers has tended to keep transaction costs low (up to a few percent of allowance prices) in emission allowance markets.

Several key features of C/T systems are worth noting. First, a cap on total emissions means that as an economy grows, new emissions-control technologies or emission-free production processes will be needed. Some observers worry that a fixed emission cap is a limit to economic growth. While there have been modeling studies on this issue, there appears to be no analysis of the effect of existing emission trading programs on economic growth (Hoffert et al. 1998; Energy Information Administration 2001; Ono 2002). However, all existing programs to date may simply be too small to have a noticeable effect on growth, so such a study may not be feasible at this time. Significant GHG emission reductions might have far larger effects. Second the problem of getting allowances to new entrants can be minimized if an active, liquid market develops or if a small number of allowances are set aside by the government for this purpose. Third, the standardization of C/T allowances allows for larger emissions markets and permit brokers to offer derivative securities based on them. This has proved important since the ability to use derivative securities like options and futures greatly enhances the flexibility a firm has in planning its operations.

2.4 Open Market Trading

There have been attempts to allow DERs to be used in C/T systems—a concept called "Open Market Trading" (OMTR) (Ayres 1994; U.S. Environmental Protection Agency 1995). Advocates of this approach typically look to create ERCs in the mobile source sector (by buying and scrapping old vehicles, or paying for upgrades to cleaner vehicles) to sell to stationary sources . Although the open market trading approach has been attempted several times, it has usually failed, often due to disagreements over credit certification requirements (National Healthy Air License Exchange 1995; Goffman 1997; EPA - Office of the Inspector General 2002). For instance, a prominent OMTR program in New Jersey collapsed in late 2002 after years of development. It appears that the costs of adequate monitoring and verification for the use of DERs in C/T systems may be high enough to largely eliminate the value of OMTR programs.

3. Analytic Studies

A considerable amount of research using analytical economic techniques has been conducted on MBIs over the last several decades. Because of the limited number of examples available for study relative to the possible variations and combinations of MBIs, this research is a valuable addition to the experience discussed below.

The argument that emission taxes and CAC regulation are equivalent in their economic impact, except for the cost difference noted above, applies only in the case where the regulator has complete certainty regarding each firm's response to the tax. This can happen, for example, if the regulator has full knowledge of every firm's abatement cost function, and there is no temporal variability in firm operating conditions that could lead to an unanticipated variation in the firm's response to the tax. These are ideal conditions, however, that are unlikely to apply in practice.

The issue of uncertainty, or variability, was first raised by Weitzman (1974), who analyzed the choice between price-based instruments (e.g., taxes) and quantity-based (e.g., allowances) in a one-period model under the conditions of uncertainty and/or variability. As Weitzman pointed out, the uncertainty/variability induces an asymmetry between what is accomplished with taxes versus a quantity control. With the latter, the regulator knows for sure what the effect will be on emissions. With a tax, by contrast, because the firm's response to the tax is not known by the regulator with any certainty, there can be no certainty regarding the effect on emissions—presumably the tax will lead the firm to reduce its emissions, but the regulator cannot know ahead of time just how large the reduction will be. Therefore, if certainty regarding the achievement of specific target for emission reduction (i.e., precision of control) is important, the quantity control works better than the tax. On the other hand, there should be some consideration of efficiency when setting the target for emission reductions.. If it turns out to be more or less costly for the firm to reduce emissions than the regulator had thought, then whatever tax the regulator sets and, in the case of quantity control, whatever target emission reduction he sets, will turn out to have been mistaken and inefficient.

However, there is an asymmetry between the inefficiency associated with a tax and that associated with a quantity control. Suppose the firm's emission reduction cost is higher than the regulator bargained for? In the case of quantity control, this means that the regulator calls for a degree of emission reduction that is, in fact, uneconomically large. In the case of a tax, this means that the regulator sets a tax that is too low. But here there is a beneficial feedback effect: since the firm's emission reduction cost is so high, it responds to the tax with less emission reduction than if its cost had been what the regulator expected. Thus, the firm's responsiveness to the tax which makes its emission reductions endogenous, unlike a quantity control, partly offsets the inefficiency stemming from the regulator's incorrect cost information. There is a similar beneficial feedback if the firm's emission reduction cost is lower than the regulator anticipated. The upshot is that the tax has the advantage over a quantity control with respect to the inefficiency associated with incorrect cost information.

The foregoing provides some intuition for the formal result obtained by Weitzman (1974), which is that the choice between tax and quantity control turns upon the slopes of the marginal changes in emission reduction cost and benefit functions. The tax is preferred (i.e., has great net social benefit) than the quantity control when the marginal benefit curve is flatter (i.e., less steep) than the marginal benefit curve. If the marginal benefit curve is relatively more steep, then the quantity control is preferred. The condition that makes the marginal benefit curve steep is essentially that there are threshold-like phenomena creating a sharp difference between the benefits of one level of emission reduction and that of a slightly larger level. The more there are such threshold effects, the greater the relative weight one would expect to be placed on precision of control, which favors the use of quantity control. If such threshold effects are relatively minor, then avoiding inefficiency in regulation is more important, which favors the tax approach.

A feature of Weitzman's analysis is that it is purely partial equilibrium. It focuses narrowly on the costs and benefits of emission reduction in a particular industry, it ignores other sectors of the economy and other inputs to production, and it is silent on the disposition of the revenues from a tax. These issues have come into focus in the more recent literature on the design of emissions taxes. At the same time, the new literature has ignored the issue of uncertainty that was originally raised by Weitzman: it assumes that the cost function for emission reduction is known to everybody, including the regulator (and the modeler). The recent literature grew out of the notion that the revenues from GHG taxes (or, more commonly, taxes on carbon dioxide emissions, often simple called “carbon taxes”) might be significant, and that they could be used to reduce the effect of other, distorting, policies (Pearce 1991; Repetto et al. 1992). This idea was named the “double dividend” that could be realized by reducing emissions and the costs of preexisting tax distortions at the same time. The term “revenue recycling” is also used.

Subsequently, a second set of analyses, pointed out another potential interaction between emissions taxes and the broader tax system, working in the opposite direction to the double-dividend effect (Parry 1995; Bovenberg and Goulder 1996). The key here is an interaction effect between environmental policy and other regulatory and tax policies, such as how income or payroll taxes might affect the choice of environmental policy. This effect comes about because energy prices rise as a result of carbon taxes, and since energy is an input in most production sectors, this leads to a slight reduction in real household wages and labor supply. These reductions, called the “tax-interaction effect,” generally lower net social welfare more than the revenue-recycling effect raises it. This finding is generally consistent with research on taxation more broadly, which shows that the use of narrow (e.g., carbon) taxes tend to make economies less efficient to a greater degree than broader (e.g., labor) taxes (Parry 2003).

Similar research into the quantity-based MBIs (e.g., emission allowances) showed more significant results, and in particular *grandfathering* (the allocation of allowances for free, based on historical emissions) was found to be problematic (Goulder et al. 1997; Parry 1997). The reason is fairly straightforward, if allowances are grandfathered, energy prices will still rise and the tax interaction effect will still operate, but the government will be forgoing potential gains from the revenue recycling effect. Thus, there is a potentially strong case for preferring auctioned allowance systems or tax systems, and an equally important emphasis on assumptions about the use of tax revenues. That is, the advantages of a carbon tax are dependent on the use of the revenue to reduce pre-existing tax distortions; other uses (such as to close budget deficits or expand government expenditures) may not have equally beneficial effects. Deficit reduction, for instance, might reduce future debt payments, so distorting taxes in the future might be lower. Expansion of government services might or might not improve social welfare, depending on whether or not the expansion exceeds the benefits that are forgone by not reducing distortionary taxes.

In a recent paper, Parry (2003) has noted that further complications could enhance these effects, including the existence of pre-existing tax deductions, pre-existing energy taxes, and capital market interactions. Tax deductions, which include, for instance, deductions for mortgage income and for employer-provided medical insurance, tend to enhance the revenue recycling effect and have little effect on the tax interaction effect. Energy taxes, such as the per-gallon tax on gasoline, act like distortionary taxes in the labor market but probably have a smaller net effect. Taxes on capital (e.g., dividend and capital gains taxes) also probably have a smaller effect than taxes in the labor market, but may tend to bias downward estimates of control costs.

In addition, there is some suggestion that auctions will tend to stimulate greater innovation and may lead to more efficient investments in technology (Milliman and Prince 1989; Kerr and Newell 2003; Popp 2003). Real-world complexities, however, such as multiple distortionary taxes and policies, monopoly power, and differences among regulated firms complicate the issue, making the optimal choice less clear (Babiker et al. 2003; Fischer et al. 2003). The same is true for the specific method used to allocate grandfathered allowances (those given away for free to pre-existing units), whether based on heat input, generation output, or historic emissions. It appears that output-based allocations can be more efficient, though this depends on specific circumstances (Babiker, Metcalf et al. 2003).

Revenues associated with either a tax or C/T program for GHGs could have significant impacts on the distribution of household income due to two other effects. First, by creating emission allowances, government creates a new asset, which has value and can be taxed. To whom that value is assigned and how it is taxed are inevitable policy choices that face governments that use MBIs. In grandfathering schemes, the value (e.g., "rents") of the allowances will ultimately accrue to shareholders. This creates the most significant effect; because stock ownership is highly skewed to the top income quintile (which owns about 60% of all shares), well-off households can actually end up better off after the imposition of a C/T policy than before, while other households are worse off and the economy as a whole is less efficient. Second, lower-income households tend to spend a higher fraction of their incomes on energy and energy-intensive goods, increasing the burden on them directly. Thus, they tend to bear the largest burden of emissions control policy, as a percentage of income.

The empirical evidence presently available relates entirely to the national economy, rather than to a regional economy such as California. One estimate of the impact of a grandfathered-based approach to a nationwide emission reductions of 15% suggested annual real income of households in the lowest quintile would decrease by about $500 (6%), but could go *up* by as much as $1,500 (1.5%) for families in the highest quintile (Dinan and Rogers 2002). This study also showed that policies that included auctioned allowances could produce the same results, or they could yield a progressive change in distribution, depending on how the auction revenues were used. A policy of auctioned allowances and decreased corporate taxes would be the most regressive, with quantitative results similar to those given above. A policy of auctioned allowances and decreased payroll taxes is somewhat less regressive, although the impact on the lowest-income quintile is about the same. However, if the auction revenues are used for a lump-sum tax rebate distributed equally to each household, annual real income of households in the lowest quintile would go up by about $300 (3.5%), while families in the highest quintile would see a decrease of almost $1,700 (1.6%). Unfortunately, this study finds a trade-off between equity and efficiency, policy choice with the most progressive income effects is among the most expensive overall and the least expensive are quite regressive. This study also finds that international trading tends to reduce the price of allowances, which reduces in regressivity of any policy choice.

A more recent study at the national level developed a more powerful analytic model with which to address this question and found generally similar results, as well as developed some new insights (Parry 2004). This study estimated the costs for reducing CO_2 emissions nationwide by 10% with various policy instruments with various revenue recycling methods. The absolute costs were lower by about an order of magnitude, but many of the patterns were similar. Thus, grandfathered permits could impose annual costs as high as $56 (0.6%) on the lowest-income quintile households, while annual income for the highest-income quintile goes up by as much as $120 (0.2%). The use of an emissions tax with lump-sum recycling leads to a progressive outcome: annual incomes for the lowest-income quintile is estimated to go up by $110 (1%), while it decreases for the top quintile by $190 (0.3%). Moreover, this study finds that conflict between efficiency and equity may be avoided if revenues from either an emission tax or allowance auctions are recycled through broad income tax reductions, enhancing both goals.

This study also examines the magnitude of the equity effects under various levels of stringency for emissions control (Parry 2004). The key result is that the regressive effects are greater under very low levels of control, because the rent-creation effect dominates the effects due to higher energy prices. Other research has suggested that in the presence of other taxes (e.g., on labor and income), and at high levels of mitigation, MBIs may lose some of their advantages over CAC regulation (Goulder et al. 1999; Fischer, Parry et al. 2003). This effect comes about because the efficiency advantages of MBIs derive from the flexibility they provide to firms, and as emission reductions approach 100%, the potential for flexibility is increasingly limited. This suggests that optimal policies may depend on the level of emission control.

It is important to note that the same effects apply to subsidies, such as those for renewable energy. Thus, while subsidies do increase the need for government expenditures, a general equilibrium analysis has shown that if they lower energy production costs and product prices, they will tend to reduce the relative price of consumption goods and will increase real household wages and labor supply (Parry 1998). This effect partly offsets the increase in the need for distorting tax revenues to pay for the subsidy, which leads to the conclusion that the optimal level of such subsidies is greater than zero but not as large as it would be in the absence of pre-existing distortions.

Analytical research strongly suggests that allowance systems are much more likely to achieve environmental goals and be cost-effective than ERC systems, and the emerging evidence from practical experience tends to support this claim (Dewees 2001; Tietenberg 2003). The principal theoretical reason for this difference is that under ERC programs, there is no firm linkage between emissions and credit price, so increasing activity levels yield increased emissions and therefore increased environmental damage. In general, therefore, ERC prices will be below the efficient value. On a practical level, ERC programs generally create more opportunities for transaction costs, which tend to reduce the efficiency of any policy. Evidence to date suggests that transaction costs for ERC programs are often quite substantial (Liroff 1986; Dudek and Palmisano 1988; Palmisano and McKeage 1992; Foster and Hahn 1995; Israels et al. 2002) And, as illustrated in the examples below, C/T systems facilitate the creation of derivative products (e.g., futures), which can help firms manage the risks associated with environmental regulation.

An interesting study suggests that while a tax on emissions from mobile sources is impractical, a fuel (gasoline) tax that varied with type of fuel, engine size, and emission control technology, or a vehicle tax that varied with mileage, or some combination, might be achieve efficient outcomes (Fullerton and West 2002).

4. Examples

Several more important examples of emission trading are discussed in this section, with particular emphasis given to two programs. The first of these is the federal Acid Rain Program to control SO_2 emissions from power plants, which is notable because it is the first large C/T program put in place and has been well-studied. The second is the Ozone Transport Commission's (OTC) NO_x Budget program, a multi-state C/T effort that included industrial sources (not just electricity producers) and was later expanded to a larger-scale federal program. The other programs that will be discussed include the long-standing federal ERC programs for air pollution, California's RECLAIM program, and two GHG programs in Europe—one in Denmark and the other in the United Kingdom.

4.1 The Federal ERC Programs

The original introduction of MBIs was the result of a U.S. Environmental Protection Agency (EPA) requirement that new facilities wishing to locate in areas with unacceptable air quality “offset” their emissions. The only way to do this was to obtain offsets from existing firms. This suddenly turned the legacy of emitting pollution into an asset that could be sold to firms entering or expanding in an area. This was late expanded to include a number of similar provisions called “bubbling,” “netting,” and “banking” that improves the efficiency of CAC regulations.

State air pollution agencies typically manage these systems with guidance and oversight from the EPA. Environmental groups have had an important role in ensuring the environmental integrity of these programs. Pollutants in these programs include volatile organic compounds (VOCs), nitrogen oxides (NO_x), sulfur dioxide (SO_2), and others. Price and availability for ERCs vary greatly by pollutant, location (ERCs cannot be transferred between different urban areas) and sometimes by other specific conditions attached to ERCs. While these programs are numerous, they are somewhat cumbersome and the incentives for the generation and sale of credits are low, so most of their utilization has been to facilitate trades within individual companies, not between different firms (Hahn and Hester 1989b; Dewees 2001).

Nonetheless, these approaches have saved billions of dollars in compliance costs with few, if any, negative effect on environmental quality (Hahn and Hester 1989a). The program likely reduced emissions below the levels required by the CAC regulations, since many transactions required a 10%–20% emission reduction for each trade. An important aspect of these programs was that they often brought to light flaws with the centralized CAC approach, that had been previously hidden (Dudek and Palmisano 1988 pp. 231, 236-9).

These flaws are problems that both regulators and regulated firms seem willing to ignore in a CAC framework, but which they are not willing to ignore in an emission trading framework. The most significant challenge is probably how to set appropriate baselines—the counterfactual emission rates that would have occurred except for the action that resulted in the creation of an ERC. There is no objective way to establish baselines, and since in an ERC program higher baselines mean the firm receives more valuable ERCs, firms may have a strong incentive to argue for a higher baseline than might really be deserved. For instance, if a firm plans to change a production process in a way that would incidentally lower emissions, they would have the incentive to attempt to seek ERCs before doing so by arguing that their baseline should be the higher emissions from the current configuration, most likely without revealing their planned change. There is likely no way for regulators to know what the firm’s plans are, and thus to judge what an appropriate baseline should be. This sort of problem does not arise in CAC systems. Similar problems arise in the determination of operating hours, which matter little in CAC systems but are very important in ERC systems. The same is true for the treatment of plant shutdowns.

These problems have serious policy implications. Dudek and Palmisano characterize ERC programs as “the harbinger of bad news” for the CAC programs they have been attached to by focusing on their shortcomings and “sloppiness” (1988 pp. 236-7). In the case of the EPA’s bubbling, netting, and banking rule, the result was 46 pages of detailed regulations in the Federal Register, which tended to reduce the flexibility and usefulness of the rule (Liroff 1986). Of course, what looks like a loss of flexibility to one party might look like simply an assurance of environmental integrity to another.

Perhaps the most important lesson for California from this example is that ERC programs are very challenging to implement. Because ERC programs are essentially additions (or modifications) to CAC regulations, if a CAC framework does not already exist (as in the case of GHGs), perhaps it would be best not to implement one as a means to eventually implement emission trading. Moving directly to a cap-and-trade approach might be better.

4.2 U. S. Acid Rain Program (Title IV)

The best-known C/T system is the EPA's Acid Rain Program for SO_2 emissions from coal-fired power plants. Key features include: the strict monitoring provisions for both SO_2 and NO_x; the national scope of the program; the relatively deep cuts in emissions (50% over about ten years); completely unrestricted trading and banking; a small auction program in the early years of the program; and, most importantly, its success. Importantly, facilities regulated by the Acid Rain Program must still comply with health-based CAC SO_2 regulations that prevent hotspots from developing, although these restrictions have not affected the market. The Acid Rain program used a system of grandfathered allowance allocation in which pre-existing units received allowances based on historical emissions free of charge, with some modifications (Joskow and Schmalensee 1998). This approach creates no government revenues.

The Acid Rain Program has been a success in several ways. First, substantial emission reductions have occurred, as shown in Figures 1 and 2. Emissions of regulated sources have declined substantially since their peak in the early 1980s. In part this is due to the availability of low-sulfur coal (from the Powder River Basin) across much of the country (Ellerman and Montero 1998; Ellerman et al. 2000). However, this process was sped up and extended by the Acid Rain Program, as emission reductions continued to occur in spite of increasing coal use. From 1990-2002, SO_2 emissions declined by about one-third, while coal-fired generation increased by more than 20%. During the less-stringent Phase I (1995–1999) regulated sources overcontrolled and banked more than a year's worth of emission allowances, and began to draw them down in more stringent Phase II, following a relatively efficient path (Ellerman 2003).

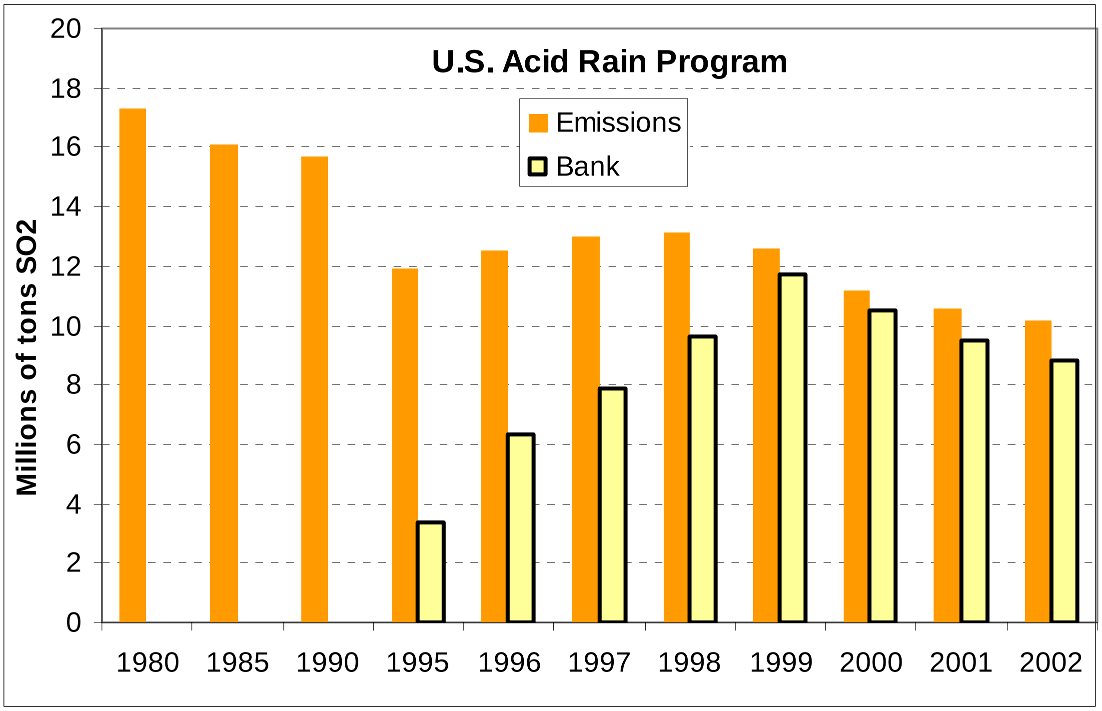

Figure 1. SO_2 emissions from sources regulated by the Acid Rain Program

Second, the program has greatly reduced the cost of SO_2 control compared to command-and-control polices. In the first five years, emissions trading reduced compliance costs by about one-third to half, estimates of the savings range from $350 million to $1,400 million (Ellerman, Joskow et al. 2000). Allowance prices have ranged from $66/ton to about $200/ton (nominal). This is not to say that the cost of SO_2 control has been cheap. In 1995 annual costs were about $726 million, and capital costs for scrubbers in Phase I alone are estimated at $3.5 billion. However, most of these savings are not due to trading of allowances per se, but from the flexibility in compliance that allowed firms to find their own least-cost approach.

An important development was that Midwestern power plants that were designed to burn high-sulfur local coal were adapted to burn low-sulfur Western fuel (from the Powder River Basin) just as it was becoming cheaper due to railroad deregulation. During the 1980s, while possible acid rain policies were being debated, the almost universal assumption was that coal-fired power plants in the Midwest, the most important sources of SO_2, would find it cheapest to control emissions by continuing to burn high-sulfur Midwest coal and use flue gas desulfurization units (or "scrubbers") (Schmalensee et al. 1998). This line of thinking arose partly due to technical reasons (it was thought that Western low-sulfur coals were incompatible with Midwestern boilers), partly economic (transportation costs had previously precluded Western coals from being competitive in the Midwest), and partly political (Midwestern legislators sometimes attempted to protect Midwestern coal mining jobs by mandating the use of high-sulfur, in-state fuels). (McCarthy 1992).

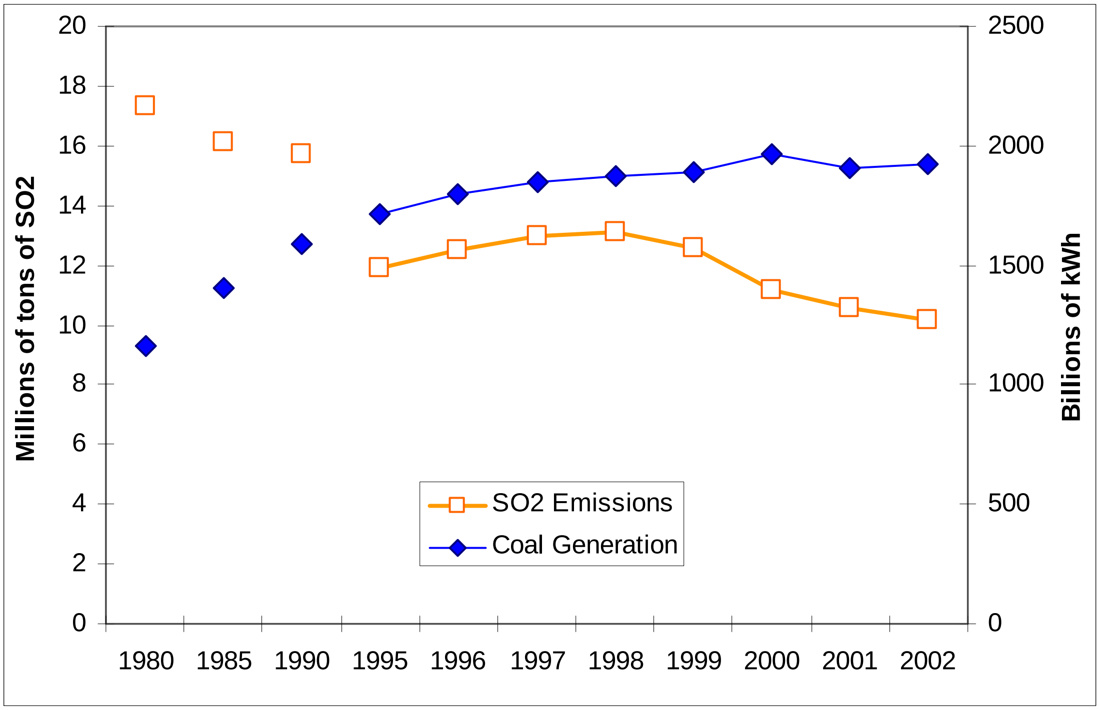

Figure 2. SO_2 emissions (left) and electricity generation from coal (right)

Following this line of thinking, some power plants ordered scrubbers in the early 1990s. Others had installed scrubbers already, in response to state emission control regulations. But as Western low-sulfur coal began to emerge as a better option and as the courts began to overturn protectionist state legislation, some scrubbers were canceled (Kolstad 1990; Smock 1991; Greenberger 1992; Kuehn 1993). Nonetheless, the erroneous expectations about the need for scrubbers led to more scrubbing capacity than was needed for compliance (Ellerman and Montero 1998). The growing realization of this result drove allowance prices down sharply, from around \$350/ton in the first trades in 1992, to under \$100 by late 1995—well below prior predictions (see Figure 2).

Third, the SO_2 market has been a success, as seen in Figure 3. Although this market is not overseen by financial regulators, the Securities and Exchange Commission, or the Commodity and Futures Trade Commission, prices in this market are relatively reliable. There are up to several dozen trades each day, resulting in from 20,000 to 100,000 allowances trading hands each week. Several different organizations monitor the market closely, some of which publish regular (daily, or monthly) reports. All vintages of allowances are priced the same, because there are no restrictions on banking, which helps smooth the operation of the market.

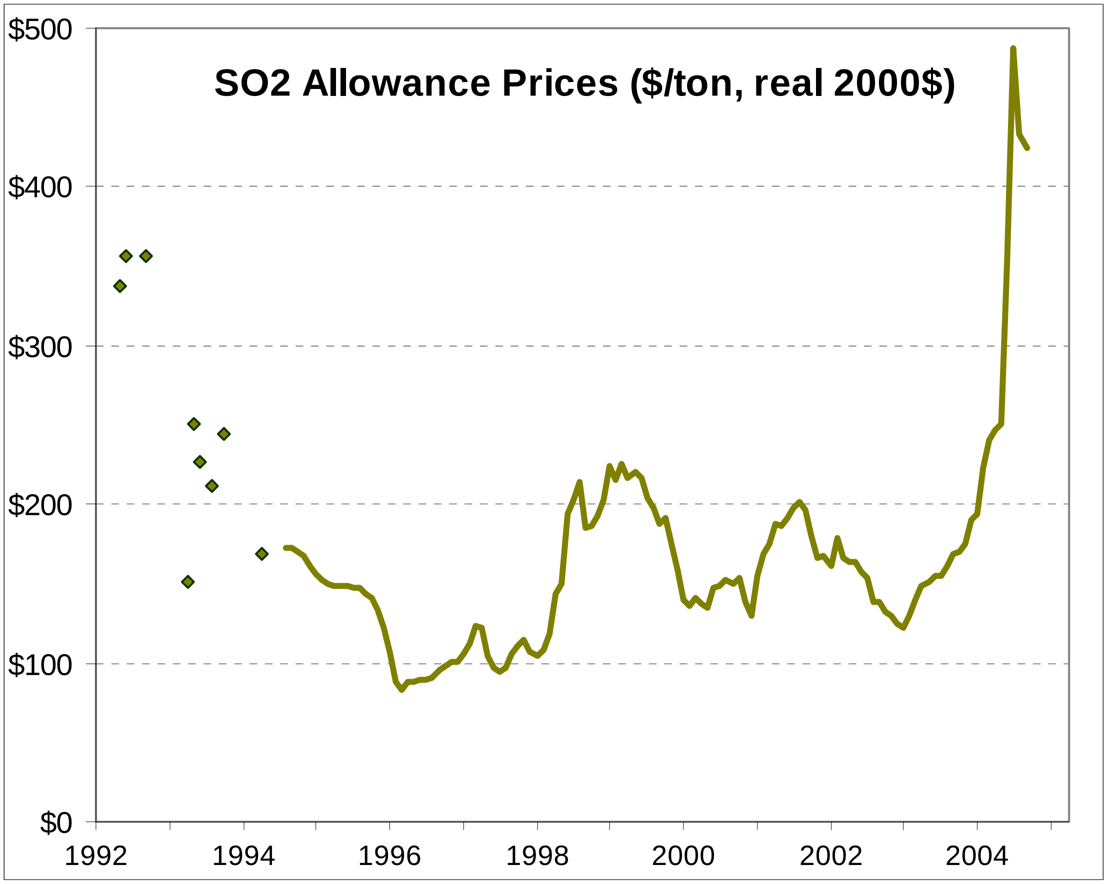

Figure 3. Prices for SO_2 emissions allowances (real 2000 $)

An important feature of this market was auctions of set-aside allowances starting in 1992 and 1993, several years before the first compliance year. Although the design of these early auctions market has been criticized, they were extremely valuable, because they helped with price discovery in an untested and highly uncertain market. The revenue from these auctions was returned to the regulated firms in proportion to their allowance allocation. These auctions also allowed for new entrants and the public to participate in the market. (Several school and public interest groups bought allowances in order to retire them.) This system also had an attractive program for early emission reductions, while some state programs encouraged early investment in emission control equipment (scrubbers), which made some allowances excess and created natural sellers. All of this helped the industry build up a bank of allowances before the first compliance year, reducing regulatory and price uncertainty.

Several important findings about the operation of allowance markets and industries regulated by a C/T system have emerged from the Acid Rain program. First, market participation and compliance strategies have evolved, from an autarkic approach in which firms trade allowances among their own units and bank their own allowances, towards a greater and greater reliance on the market (Swift 2001). Second, allowance prices have shown considerable volatility, and the lower bound of emissions prices have been shown to be equal to the marginal cost of operating emission control devices (scrubbers) (Ellerman and Montero 1998). Third, the relatively few units that did install scrubbers increased their utilization and lowered their emissions beyond the original design specifications as a result of the incentives offered by the allowance market

(Taylor et al. 2003). This is an example of learning-by-doing, which can help reduce the cost of emission control.

However, there is no guarantee that the surprisingly low prices (and therefore the implied low cost of emission control) observed for about the first decade of the market (1994–2003) will be repeated in other emission markets. Although allowance prices were quite low for a long time, this is the result of the unexpected ability to switch readily and cheaply to a much cleaner fuel. The potential for such an outcome seems unlikely in the case of GHGs in California because switching to fuels with less carbon than the existing fuel mix seems unlikely without a significant cost premium.

4.3 Ozone Transport Commission NO_X Budget

The first cap-and-trade system devised by independent political entities (U.S. states) rather than imposed by a central government (as in the case of the Acid Rain program), was the OTC NO_x Budget Program (Farrell 2000; OTC and EPA 2003). It operated successfully between 1999 and 2002, after which it was subsumed by the federal NO_x SIP Call program administered by the EPA. This section begins with a basic overview of the OTC NO_X Budget Program, covering regulated units, the development of a "model rule," emissions monitoring, and accounting issues. It then reviews the key parameters of program performance, including the following: environmental outcomes, leakage, economic outcomes, industrial source participation, opt-in and set-aside provisions, and state-by-state allocation.

The OTC NO_x Budget applied to electrical generating units 15 megawatts (MW) or larger and similar-sized industrial facilities (for example, process boilers and refineries) with a heat input rate of 250 mmBtu per hour or more. The NO_x Budget involved over 900 electric generating units (EGUs) and over 120 industrial units. More than 100 different facility owners were affected. Although a substantial number of facilities were covered, these large stationary sources represented less than half of total NO_x emissions in the region. The NO_x Budget program used a system of grandfathered allowance allocation in which pre-existing units received allowances based on historical emissions free of charge, with some modifications. This approach creates no government revenues.

The emissions control period was defined as the five months from May through September, known as the "ozone season." From 1999 to 2002, the NO_x emissions cap for the ozone season was set at 219,000 tons, representing a 25% decrease in emissions compared to 1995 levels, and a 54% decrease compared to 1990 base year emissions.

While the NO_x Budget used a cap-and-trade approach similar to the Acid Rain Program for SO_2, it was not a centrally organized system. Rather, it was the result of coordinated laws and rules in each state in the OTC. The federal EPA did provide important assistance, though, such as tracking emissions and allowances. To facilitate regional control of NO_x, the OTC states chose to create a "model rule," which could be modified to fit specific circumstances in each state prior to state adoption (Northeast States for Coordinated Air Use Management / Mid-Atlantic Regional Air Management Association 1995). The details of the model rule were crucial, since they needed to be defined enough to yield a consistent regulatory program and flexible enough to suit the interests and local politics of each jurisdiction.

Importantly, the OTC states had already developed a history of cooperative work on air quality management, partly due to the debates on acid rain in the 1980s. This created a network of cooperative organizations, personal relationships, technical competencies, and, most of all, trust. The development of the model rule began in late 1994 when the level and timing of emission reductions was set. The technical/political negotiation process that followed was designed to determine whether emission trading would be used as part of a multi-phase approach. In early 1996 the model rule was published and the OTC states began to develop their own rules, which were all final by the end of 1998, thus allowing the NO_x Budget to start in May 1999. Regulated sources felt this was rather short-fused, because the rules were not in place in several states until less than a year before the start of the first ozone season, while engineering, procurement, and construction of NO_x control technologies can take several years. Regulators disagreed with this assessment, noting that if the emission trading program had not come together, very similar command-and-control regulations would have been the default. Further, much of the delay in developing state rules was due to the need to address the concerns of regulated industries, which then protested about the uncertainty created by this delay.

As originally agreed in the model rule and as implemented in the various state rules, the OTC NO_x Budget was to continue after 2002. However, the geographic scale of NO_x transport and ozone pollution turned out to be larger than that of the OTC states. In the late 1990s and early 2000s, the EPA issued the NO_x SIP Call and won a number of legal challenges, the effect of which was to extend to a total of 22 states the level of emission reductions that would have been achieved in 2003 by the OTC NO_x Budget. Thus, a state-based cap-and-trade program evolved into a federal program with similar features.

An important feature of the OTC NO_x Budget is that it relied on a pre-existing mechanism for monitoring and accounting that had been developed by the federal EPA for the Acid Rain program. Requirements were in place for continuous emissions monitors (CEMS) for NO_x emissions from EGUs over 25 MW capacity, which meant that the only new monitoring rules were for smaller EGUs and industrial sources.

After some debate about accounting rules for NO_x allowances, the states decided to use a serialized approach. That is, each allowance created by a state would receive a unique serial number that the EPA would assign. The other options—individual state serial numbers and unserialized approaches—were rejected. State-by-state serial numbers were thought to be too complicated, and unserialized approaches would have required more oversight. The EPA was chosen to be the accountant rather than choosing a private firm, because this left a key part of an environmental regulatory program in the hands of government. Further, no important advantage in competing private accounting systems was found. Other than the accounting role played by the EPA, there is no market oversight in the OTC NO_x Budget program that would be comparable to, for example, the role that the Securities & Exchange Commission plays for the New York Stock Exchange or that the Commodities & Futures Trading Commission plays for the Chicago Mercantile Exchange. Private emissions traders and brokers have been able to operate successfully in this environment.

Environmental Outcomes

Figure 4 and Table 1 illustrate the most important accomplishment of the OTC NO_x Budget program: the reduction in seasonal emissions. Figure 4 shows total allocation, or the total number of allowances provided to regulated sources each year, compared to the total emissions during the May–September period when the program was in force. For historical reference, Figure 4 also shows emissions in 1990 and 1995. Significant reductions were achieved between 1990 and 1998 due to implementation of regulatory standards known as “Reasonably Available Control

Technology" (RACT). RACT generally involved the installation of a control technology called a "low- NO_x burner," sometimes in combination with overfire air.

Figure 4 shows that emissions were always below the number of allowances that were allocated. In fact, on average, regulated sources overcontrolled by 13.5%. Over 110,000 tons of NO_x that could have been emitted into the atmosphere were either delayed or avoided as banked tons were either used or expired, respectively. By 2002, emissions were 34% lower than in 1995—the period when RACT technology standards were being implemented (EPA, personal communication). For the four years that the OTC NO_x Budget was in place, emissions were 62% below 1990 base year emissions.

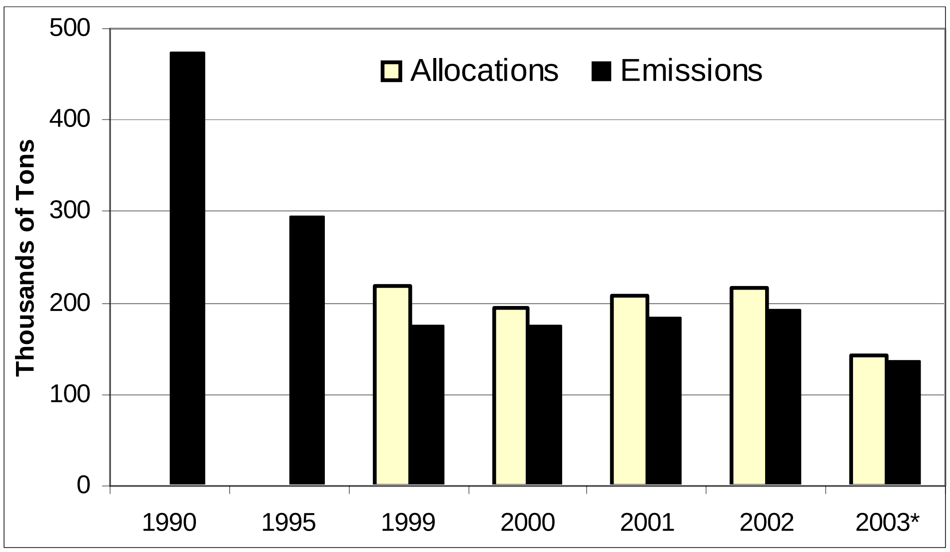

Figure 4. Allocations and emissions for sources in the OTC NO_x Budget
*In 2003, the OTC NO_X Budget program was subsumed by the NO_X SIP Call

Table 1. OTC NO_x Budget Program allowances vs. emissions

	1999	2000	2001	2002
Total Allocation (tons)	219,438	195,398	207,756	217,175
Total Emissions (tons)	174,843	174,492	183,283	193,393
Unused allowances (tons)	44,595	20,906	24,473	23,782
% Unused allowances	20%	11%	12%	11%

Source: U.S. EPA, OTC NO_x Budget Program Compliance Reports, 2000–2003

Looking at the year-to-year changes in Figure 4, two factors deserve mention. First, the drop in allocations between 1999 and 2000 is largely explained by the discontinuation of early reduction credits. These are credits which were given to firms that lowered their emissions prior to the start of the regulatory program. They often work somewhat like ERCs, where performance is measured against a baseline. However, an important distinction is often made—while there may be no limits to the potential number of ERCs that can be created, and they are not linked to any specific cap, early reduction credits can be designed to have the opposite properties. In the case of the NO_x Budget, as in other examples, a small, fixed number of allowances from state allowance totals were identified as early reduction credits. Thus over a period of several years, the cap on emissions remained effective, although some allowances were available somewhat earlier because of the early reduction credit program (this is not important for stock pollutants like GHGs). In addition, the distribution of allowances among regulated firms changed, so that firms that undertook early emission reductions ended up with more allowances than they would have otherwise. This last feature of early reduction credits may be important for GHGs, because the potential redistribution of allowances among various firms may change the interest and support for such programs.

Second, the rise in allocations in 2001 and 2002 is largely the result of the addition of sources in Maryland being regulated under the NO_x Budget for the first time. Nonetheless, the overall environmental outcome is clear: substantially reduced emissions of NO_x. In 2003, a more stringent allocation took effect with the transition to the NO_x SIP Call, yet the OTC states adapted smoothly. Emissions continued to fall, dropping by almost 30% from 2002 values.

Figure 5 presents a more detailed analysis of environmental performance for participating EGUs within the Pennsylvania-New Jersey-Maryland (PJM), New York, and New England power pools, specifically from 1998 to 2001. This includes all power plants in New England, New Jersey, New York, Delaware, and some of those in Pennsylvania. While this is not the entire universe of sources in the OTC NO_x Budget, it is the largest set for which straightforward comparisons can be made. Using plant-level data for 1998–2001, various measures of performance were compared. The data has been normalized in Figure 5 to allow all the relevant values to be shown. Five measures of environmental performance are illustrated along with the amount of electricity generated during the relevant ozone periods. 1998 is used as the basis of comparison, thus taking into account all of the emissions reductions that were achieved between 1990 and 1998 due to RACT implementation.

Moving from left to right in Figure 5, the NO_x emissions from EGUs declined in each of the first three years of the OTC NO_x Budget. The decline from the 1998 emission levels of the NO_x RACT program averages over 25%. The next four sets of measures are emissions rates, either emissions per hour or emissions per megawatt-hour (MWh) of net electricity generated and provided to the grid. These measures provide different perspectives on how the NO_x Budget performed. Generally, the data shows good performance. The average emission rates decline each year, and both average and peak emission rates per MWh of generation decline substantially, by more than 50% from 1998 levels on average for 1999–2001. Note that the large declines in emissions per generation are driven by the combined effects of decreasing emissions and increasing generation. This is a good example of one of the chief benefits of cap-and-trade regulation: the cap helps maintain environmental protection goals while still allowing for increased economic activity. If command-and-control approaches had been used, emissions would have likely have first declined in 1999 when the new regulation had taken effect and then gone up as generation increased.

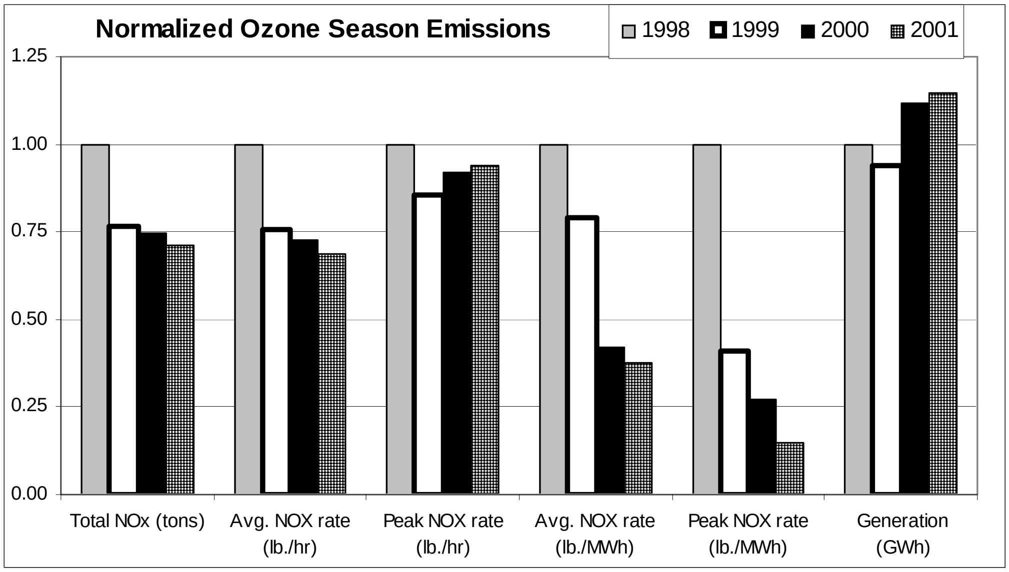

Figure 5. Measures of environmental performance of the OTC NO_X Budget*

*Normalized; includes representative EGUs only

This, in fact, is the pattern seen in the peak emission rate (NO_x lbs/hour) recorded over any single hour during the ozone season. This performance measure first declines by about 15% from 1998 to 1999 and then rises again, although never rising back to 1998 levels. This pattern is of concern, because smog is an episodic problem in the eastern United States, where the highest concentrations and greatest health risk occurs on a small number of short periods in the range of 2–5 days. These "ozone episodes" tend to happen on hotter days, which is also when electrical generation tends to peak. However, the higher peak emission rates shown in Figure 5 may not be a health concern for two reasons. First, depending on the emissions profile of the peak generating units in a given state, these peak rates do not always occur during ozone episodes, although they usually do. Second, the emission peak rates are short-term phenomena lasting only an hour or two, although the importance of the short duration may be limited because of multi-day lifetime of NO_x in the atmosphere and the potential for regional transport. Thus, total EGU NO_x emissions during ozone episodes in 1999–2001 were less than those during ozone episodes in 1998, and average emission rates were generally lower as well.

Economic Outcomes

Three types of economic outcomes of the OTC NO_x Budget are evaluated here: (1) the allowance market, which reflects the marginal costs of emissions control, (2) electricity markets in the OTC region, and (3) the overall performance of the economies of the OTC states.

The OTC NO_x Budget had several distinctive features in relation to the development of its allowance market. First, there were three phases of the program. Phase 1 adopted the existing federal RACT performance standard, while Phases 2 and 3 evolved into progressively stringent emission trading. Second, the OTC NO_x Budget had no methods for early price discovery before it went into effect. In contrast, the Acid Rain program for SO_2 had a set of auctions several years before the start of the regulatory period. Although these auctions were criticized for not providing the most accurate and useful price information, they were informative to market participants and facilitated start-up of the program. Third, banked allowances in the OTC NO_x Budget were slightly discounted because of a regulatory provision known as “flow control,” which was designed to prevent spikes in emissions that would exacerbate ozone formation. The details of flow control are not discussed here, except to note that flow control created some additional uncertainty and made banked allowances less valuable than current or future year allowances.

The price of NO_x allowances were forecast by the EPA, various consultants, and other researchers. Generally, the costs in Phase 2 were expected in the range of \$1,200–\$2,400, and costs in Phase 3 in the range of \$2,500–\$3,500 (STAPPA/ALAPCO 1994; ICF Resources 1995; Dorris et al. 1999; Farrell et al. 1999). Specific forecasts depended on highly variable factors such as the relative prices of gas and coal, but were bounded to some extent by relatively well-understood economics of NO_x control engineering and technologies. By the time that NO_x trading started in 1999, the RACT implementation phase had already captured a significant amount of “low hanging fruit,” or low-cost emissions reductions, specifically through the use of low-NO_x burners, sometimes in combination with overfire air. These control options were relatively cheap. For large coal-fired utility boilers, costs were in the range of \$100–\$400 per ton of NO_x reduced (EPRI 1999), while costs for industrial boiler retrofits were under \$800 per ton (Amar and Staudt 2000). Once RACT implementation was complete, though, the remaining technology options for achieving additional reductions, such as selective catalytic reduction (SCR), were more expensive. The advent of trading was intended to provide flexibility to help alleviate the higher projected costs associated with these technologies.

Figure 6 shows the prices for NO_x allowances in the OTC NO_x Budget market over the period 1998 to early 2004. Although there was significant price volatility at the outset in 1999, most NO_x allowances sold for prices well below the forecasts. Since 2003, the market functioned within the NO_x SIP Call. The effects of the changes in program status are visible. Note that trading occurs year-round, despite the fact that NO_x allowances are only required to cover emissions during the May to September ozone season.

A small amount of emission trading began in early 1998 as some regulated sources came to believe that the program would go ahead and that they could take advantage of the opportunity to either lower costs or perhaps even generate revenue through allowance transactions. Trades began at about the level that most forecasts had predicted, approximately \$1,500/ton of NO_x. During the middle of 1998, it became clear that most OTC states would in fact implement the NO_x Budget in 1999. By the end of 1998 and the beginning of 1999, average monthly allowance prices had risen to over \$5,000/ton, far above the cost of control for any regulated sources. Market participants thought that the market was short, meaning that regulated firms might not have installed enough emissions control equipment in aggregate to meet the cap.

As this realization occurred near the end of 1998, there was insufficient time to install control equipment for the upcoming ozone season. It also seems that some participants in the NO_x market were surprised to find that the experience of the SO_2 market (low prices and an abundant supply of allowances) was not repeated. These factors added up to a tight allowance market with insufficient supply of allowances relative to demand. Allowance prices naturally rose. Importantly, only a few economically significant trades, meaning trades between different firms, occurred during this period of high prices.

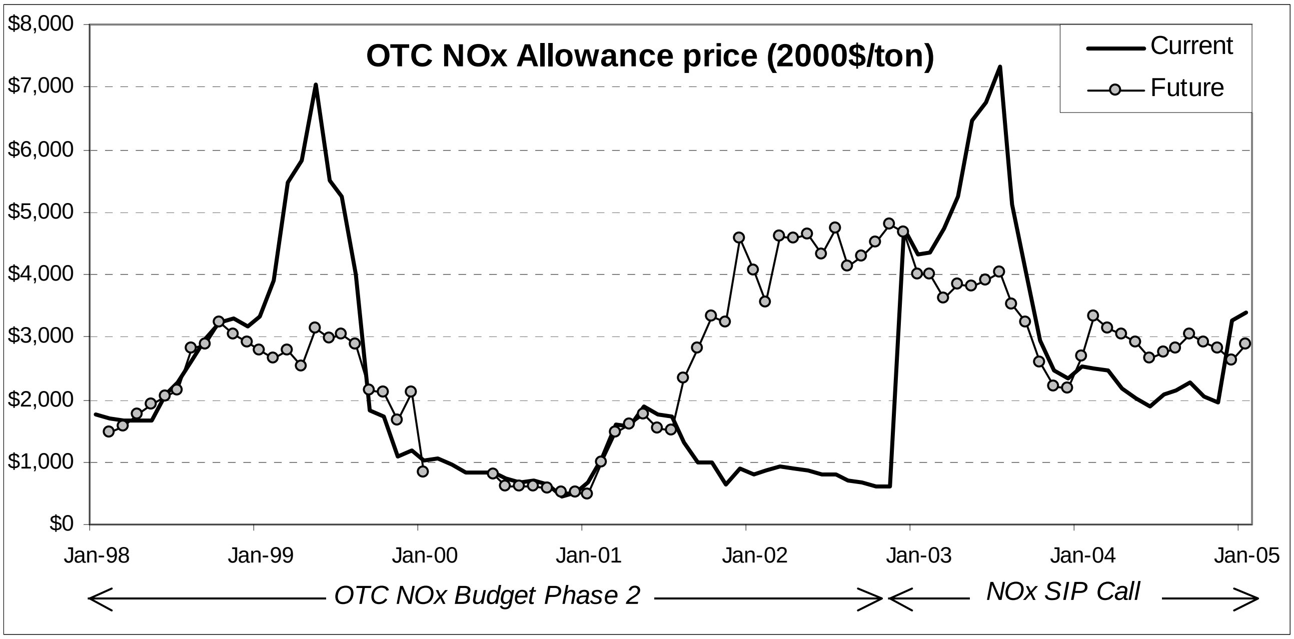

Figure 6. Market prices, OTC NO_X Budget Program (real 2000$ dollars)

Prices stayed high for several months, but fell back to the predicted range by July 1999, and fell to about $1,000/ton by the end of the year. Several factors accounted for this fall. First, early reduction allowances began to enter the market, starting with New Hampshire allowances in early April, followed by large distributions to New York and New Jersey. This dramatically expanded allowance supply. Second, several firms found that, given incentives, they could install additional emissions control equipment in a more timely fashion. Unexpected and much faster-than-normal installations of controls on plants in Massachusetts, New Hampshire, New Jersey, and Pennsylvania reduced demand. Third, a Maryland lawsuit reduced demand further by taking power plants that were expected to be net buyers of allowances out of the market, albeit temporarily.

The response of state governments and regulated sources during the early price spike illustrates the advantageous features of a cap-and-trade system. Regulators stood by the market system despite the high prices. They did not seek to impose price caps, or safety valves, that would set a ceiling on allowance prices, effectively allowing the plants to exceed their emissions allowances in exchange for paying a fixed penalty (see, e.g., Pizer 1999). Also, firms used the market to work out their difficulties rather than seek legal challenges in court. In a command-and-control system, uncertainty and delays would potentially cause industry to seek such “regulatory relief,” which in turn would reduce the environmental effectiveness of the program. While a few firms experienced higher costs, others gained a windfall, although overall the effect was relatively small. Most important, the OTC NO_x Budget provided powerful economic signals that prudent management of NO_x emissions could reduce compliance costs.

Subsequently, the OTC NO_x market matured. Emission trading became more common and price volatility declined. For the remainder of the period that the OTC NO_x Budget was in force (2000–2002), prices averaged somewhat below $1,500/ton. In addition, a difference in prices for banked and current year allowances developed to reflect the flow control restrictions on using banked allowances. Thus, the market was able to adapt readily to a complex regulatory issue.

Most of the allowances sold in 1999–2002 for Phase 2 of the program traded at prices below the range that had been forecasted prior to implementation.

By the middle of 2001, regulated sources were already looking ahead to the more stringent cap that would be in place in 2003, the Phase 3 period. Although it was expected (and later proved to be true), that Phase 3 would be replaced by the NO_x SIP Call, regulators in the OTC states had begun to issue rules for how banked allowances from the OTC NO_x Budget would be converted for use in for NO_x SIP Call compliance. Again, the market adapted readily.

However, a similar pattern of industry lawsuits and other delaying maneuvers also emerged from some of the firms that were newly regulated under the NO_x SIP Call. The resulting pattern of expectations and court-issued complications to the original regulations led to uncertainty that drove prices up again—this time over $7,000/ton. Again, this was higher than predicted. In addition, uncertainties about the performance of new emission control technologies added to the desire of some regulated sources to purchase allowances. By early 2003, allowances in the NO_x SIP Call market itself had started to trade at relatively high prices, about $4,000/ton, for similar reasons. However, these prices quickly fell to the $2,000–$3,000 range, which is at the low end or below the range predicted prior to implementation.

Using average values for the 2000 ozone season, NO_x emission allowances were priced at $0.40/MWh, while electricity prices averaged $42/MWh and peaked at over $1,500/MW in at least one market. Given these prices, it is likely that power plant operators would favor reliability in generating electricity over making slight changes to the emissions control equipment to optimize NO_x control costs. Interviews with plant operators and environmental managers of power companies indicated this was the case.[2] Emphasis was placed on generating electricity to meet contract requirements and/or spot market demands, not on trying to come as close to emission limits as possible. The structure of contracts in electricity markets would tend to reinforce this effect, since they punish both over- and under-generation relative to the amount promised in day-ahead markets. Interviews with corporate and plant managers and with other market participants supported these arguments.

It is not possible to observe the average cost of controlling NO_x emissions, since this data is proprietary to each firm. However, long-run market allowance prices provide a proxy for long-run marginal control costs, which tend to approach average costs. Using emissions for 1999–2001, a high estimate of allowance prices ($1,500/ton), and 1998 base year emissions, a total cost for emission controls beyond the NO_x RACT program can be estimated. This estimate is subject to numerous uncertainties, and is probably accurate only within a factor of two or three, but it provides us some idea of the cost of the OTC NO_x Budget program. Using this approach, the total cost of the OTC NO_x Budget program, beyond the NO_x RACT phase, is about $60 million from 1999-2001, or approximately $0.15/MWh. This can be contrasted to average wholesale cost of electricity at approximately $35 per MWh over the same period. Thus, the cost of the OTC NO_x Budget program is less than half of one percent of the cost of electricity, which is the same order of magnitude of other market-based emission control programs. The percentage would be even smaller if it were calculated on a retail basis, or what the consumer pays.

The impacts of environmental regulation on overall economic performance can also be estimated. Using the values calculated above and data from the U.S. Department of Commerce, it

[2] These interviews were conducted by phone and in person with power plant managers, power company environmental managers, and emission traders in the summer and fall of 2003.

is clear that the economic impact of the OTC NO_x Budget is negligible compared to the Gross State Product (GSP) of the OTC states, representing less than 0.0005%. It is difficult to see how anything so minute had any significant economic impact on overall activity, growth, or jobs. There is no evidence that the OTC NO_x Budget slowed the state economies in which it was applied. For the years that inflation-adjusted GSP data is available (1987–2001), the OTC states show a lower average annual growth rate of 2.55% before the OTC NO_x Budget was implemented, relative to non-OTC states (3.46%). However for the first three years of the OTC NO_x Budget program, this pattern reversed. OTC states averaged a higher annual growth rate (3.30%) than did non-OTC states (3.15%). This does not prove that added environmental regulation leads to higher growth. The relatively high growth rate in the OTC states does suggest that the higher cost of electricity production was overwhelmed by countervailing factors in the state economies.

4.4 California RECLAIM

The last U.S. example is California's Regional Clean Air Incentives Market (RECLAIM), a C/T program for SO_2 and NO_x emissions from industrial sources such as power plants, refineries, and metal fabricators (Mueller 1995; Lents and Leyden 1996; Klier et al. 1997; Thompson 2000; Israels, Grow et al. 2002). RECLAIM does not permit banking, since government regulators felt this would compromise its environmental integrity. This proved to be a significant problem for RECLAIM, emphasizing the benefits of allowing banking. Fortunately, banking is relatively unimportant for stock pollutants like GHGs, making it easier to impose effective and efficient emission trading programs.

When RECLAIM was first implemented in 1994, the cap was generous, allowing for an increase in emissions over historical levels for many sources, but it declined steadily each year, aiming at an overall reduction of about 75% by 2003. For the first several years, the RECLAIM market functioned quite well, with readily available allowances at low prices. However, emissions in 1993–1998 did not decline nearly as fast as the cap, due to a failure of many (but not all) participants to install emission control equipment. Although the state regulatory agency amply warned participants of a looming problem, many firms were unwilling to take appropriate actions because of a failure to consider future emission allowance markets and its belief that the government would bail them out in case of serious problems.

The result was a breakdown of the market and in response a temporary abandonment of the MBI approach by the state government. By early 2000 it had become clear to even the most shortsighted that emissions would exceed allocations, which was a problem, because RECLAIM had no banking provision, and prices for NO_x allowances skyrocketed to over $40,000/ton. Electricity companies, which were making record profits at the time, could afford these prices, but other companies in the RECLAIM market could not. Thus, the RECLAIM cap was broken, and several firms were significantly out of compliance and paid record fines. This is the first time a C/T system had failed in this way. Facing significant political pressure, the state regulatory agency decided essentially to go back to a CAC approach for electric power plants by requiring them to submit compliance plans. This is particularly important in that most cost savings in C/T systems come from the ability to innovate in compliance strategy, not from buying or selling allowances. In addition, state regulators separated power companies from the rest of the RECLAIM market and subjected them to a high tax for emissions not covered by allowances. For other participants, RECLAIM proceeds as before and allowance prices have moderated.

Several key lessons emerge from the RECLAIM experience. First, because they force firms to gather more information and make more decisions, MBIs may be *more* difficult for firms to understand and manage than CAC programs, even if they have lower costs. This is especially true for smaller companies, some of whom may even have increases in monitoring costs required by RECLAIM that were greater than the savings in control costs. Second, in some cases the optimal strategy may be non-compliance, placing more emphasis on the design of penalties. Third, emission markets are no different from others; they are volatile (especially when it is not possible to store the commodity, like electricity).

4.5 Denmark CO_2 Program

The first emission trading program to address climate change was a C/T system for CO_2 emissions adopted by Denmark in 1999 to achieve a national GHG emission reduction target of 5% in 2000 (relative to 1990) and 20% by 2005. The Danish program covers the domestic electricity sector (electricity imports and exports are treated separately), which is made up of eight firms, although two account for more than 90% of all emissions. Allocations were made on a modified historical basis and are not serialized. Banking is limited and there is some uncertainty about the validity of allowances beyond 2003. This program includes a tax of about $5.5/ton for emissions that are not covered by an allowance, which means the integrity of the cap is not guaranteed. It is possible to use verified emissions reductions (VERs) as well as credits created through provisions of the Kyoto Protocol (discussed below). Monitoring is accomplished through an analysis of fuel consumption—in-stack monitors are not required.

The Danish power sector met its CO_2 cap for the last several years. The C/T program was adopted in order to regulate the last major unregulated set of CO_2 emitters so that Denmark could reach its domestic CO_2 goals cost-effectively while also gaining experience with emission trading. In the first two years of the program, about two dozen trades were made in the approximate range of $2–$4/ton. This has resulted in over half a million allowances changing hands. Some of these trades have been swaps of Danish allowances for VER credits, and one trade of Danish allowances for U.K. allowances (discussed below), the first instance of a trade of two government-backed MBIs. However, the Danish system will soon be superseded by a European Union (EU) system.

The Danish experience illustrates that C/T systems can be used in conjunction with other policies, in this case differentiated by sector. It also illustrates that smaller C/T systems can be integrated into larger programs. And it has the distinction of being the first C/T system for GHGs.

4.6 United Kingdom Climate Change Levy and Emission Trading System

The first economy-wide GHG control policy was announced in November 2000 by the United Kingdom, and it contained a combination of MBIs, including taxes, subsidies, and ERCs. This program is designed both to achieve significant emission reductions and to provide U.K. organizations experience in emission trading, not least so that the City of London might become a leader in this activity.

The first part of this policy is the Climate Change Levy on fossil fuel energy supply for industrial users that came into effect in April 2001 and would typically add 15%–20% to the cost of to the firm. This has had moderately increased the demand for energy efficiency, and the revenue has been recycled into a fund that subsidizes consulting and capital purchases to reduce energy use.

Moreover, it provides an important basis for voluntary participation (including credit generation) in the ERC program, possibly overcoming one of the key problems associated with past ERC programs. Through these two mechanisms, the revenues collected by the Climate Change Levy, and its resulting impact on the British economy have been limited.

The U.K. Emission Trading System (ETS) is an ERC program open to reductions in all GHGs (measured in CO_2-equivalent, or CO_2e). To join the program, firms must enter into a Climate Change Levy Agreement (CCLA) in which they voluntarily accept an emissions cap in return for an 80% reduction in their Climate Change Levy until 2013. Companies that adopt such a target also earn the right to use ETS credits to meet their CCLA targets and sell allowances generated by exceeding their target. By the end of 2002, over three dozen trade organizations had designed model CCLAs for their members and over six thousand companies had signed CCLAs. CCLA firms that do not achieve the promised reductions are taxed on the excess at about $44/ton of CO_2e, measured as CO_2.

The last component, Direct Entry, is a $310M subsidy that the U.K. government made available through an auction for voluntary actions by eligible firms to reduce GHG emissions in 2002–2006 from a 1998–2000 baseline. This auction is designed to obtain the maximum emission reduction and provide some price discovery by allowing eligible companies to bid a fixed amount of emission reduction for a given price, and selecting (through repeated rounds) the price that yielded the maximum emission reduction given the government's budget. Electricity and heat production were not eligible for this program, except for combined heat and power, which was allowed in. This approach made bids for reductions of non-CO_2 GHGs more likely and created a natural set of buyers (energy companies) and sellers (manufacturers and end users) for ETS credits. The rules for the ETS include standards for certifying ERCs through third-party verifiers. This auction was held via the Internet over two days in March 2002, and resulted in 34 organizations (of 38 bidders) winning subsidies at the level of about $22/ton-$CO_2$e. Over half of the emissions will be non-CO_2e GHGs. However, there is some speculation that the U.K. government paid more for these reductions than was necessary.

In addition to these two methods, organizations can join the ETS through more traditional means, by earning ERCs from a specific project that meets all the necessary monitoring and verification requirements of the ETS and simply by buying or selling credits. Over time overseas-sourced ERCs (similar to those in anticipated in the Kyoto Protocol, as discussed below) may be allowed, but the U.K. government is going slowly to reduce uncertainty in the early stages of the market. There has been considerable interest in the U.K. ETS. By the end of 2002, over 400 companies had opened accounts on the UK registry and about one million credits had been exchanged in several hundred individual transactions. Prices on this market are in the range of $5–$10/ton-CO_2e, measured as CO_2.

5. Lessons for California

From these cases, observations and lessons for California can be drawn for two broad areas: (1) the design of a GHG emissions trading system, and (2) coordination with other policies.

5.1 The Design of a GHG Emissions Trading System

In contrast to many other pollutants, GHG emissions are more dispersed across the major sectors of the economy, and a greater fraction is associated with large stationary sources. In California, non-CO_2 GHGs make up a larger fraction of all GHG emissions than for the nation as a whole

(Brandt et al. 2004). This is significant because an emissions trading system generally benefits from a set of participating sources that is large and diverse. A broad set of participating sources expands the pool of opportunities to reduce emissions. This increases the likelihood of discovering low cost reductions, often achieved through innovation, which in turn drives down overall program costs for the regulated facilities and for society. The case of the OTC NO_x Budget supports the axiom that broad participation is good. Although EGUs were the prime source for regulation and the single largest stationary source of NO_x emissions, the program extended to other industrial sources, such as refineries and chemical plants. These facilities reduce emissions sharply and were net sellers to the EGUs, indicating that their marginal cost of reducing NO_x emissions was lower and therefore served to lower overall program costs.

Thus, California may want to give priority to integrating a broad set of sources and sectors into a GHG emissions trading system. Steep reductions in GHG emissions are needed to avert dangerous climate change, and at the same time there is concern over the cost of achieving these reductions. In addition, "end of pipe" solutions to reducing GHG emissions are unlikely to be widely used in the near term. The installation of end-of-pipe "scrubbers" was not as important a compliance option for the SO_2 and NO_x emissions trading programs as some had expected, largely due to the ability of engineers to extend the capabilities of other, cheaper technologies. Comparable GHG technology, known as "carbon capture and storage," is currently limited to uses such as enhanced oil recovery, and the technology is expensive. It will not gain wide application unless emissions caps are set so stringently that all cheaper reduction options are first exhausted. As a result, there is a sharp need for a GHG emissions trading system that can exploit diverse technological solutions, and this is supported by covering a broad array of emissions sources and gases (i.e., non-CO_2 gases).

Given that a reduction in GHG emissions of 50% or more is likely necessary to avoid dangerous interference with the climate system, climate policy implies a wholly different energy supply system than the one that exists today (Hoffert, Caldeira et al. 1998; Nakicenovic et al. 1998). This includes different energy resources, such as wind power, and greater efficiency in both distribution and end-use of energy. The pace of this change may be spread across a longer period of time than the control of regional ozone (perhaps four decades or more), but the scale of the challenge is also much greater, as are the barriers to policy development. In broad terms, effective climate policy entails the stimulation of energy technology innovation, changes in investment patterns in the energy industry, global participation, and broad cooperation throughout the public and private sectors, including factions historically opposed to climate policy (Baer et al. 2000; Morgan 2000; Hoffert, Caldeira et al. 2002; McCright and Dunlap 2003).

Taking these factors into consideration, a GHG cap-and-trade system need not start with a cap as stringent as the OTC NO_x Budget, though near-term reductions and a steep ratcheting down of the cap over time would be necessary. Adopting a cap, even if it is not particularly stringent at the outset, is a major step in creating incentives for innovation and investment to meet the long-term goals of climate policy (Nordhaus and Danish 2003). A pre-determined schedule of long-term, downward adjustments to the cap may be useful, though, to avoid entrenchment in the status quo (i.e., a situation where regulated sources become invested in the initial target level and thus become strongly opposed to more stringent targets later on).

Allowance allocation

In an output-based model, the allowances are granted based on actual production, such as megawatt-hours in the case of the power sector. In this way, allowances can be allocated to any producer. In the case of some renewable power plants, such as a wind plant, it has no GHG emissions, so the plant does not need allowances for compliance and can therefore sell all of its allocation into the market. This raises direct revenue, which can then be reinvested into new or expanded generation. As a result, the output-based allocation method has the potential to accelerate market penetration of clean generation technology, a vital step towards achieving significant GHG reductions in the long term.

Another allocation option is based on historic emissions. Because GHG emissions come from a variety of sources and sectors and involve a variety of gases, it may be useful for regulators to focus on emissions during allocation rather than parameters that are specific to EGUs, such as heat input. An emissions-based approach could facilitate multi-sector allocation, either at the outset of program implementation or over the long term, by allowing for inter-sector assessment of fairness and equity. However, emissions-based allocation may create somewhat of a "windfall" for polluters. One possible compromise is to start with a system that is mostly grandfathered allocations (free allocation to pre-existing sources) with a small percentage of auctioned allowances that gradually shifts to an all-auctioned approach. Another would be to allocate only enough allowances to keep existing sources financially whole, and to auction the rest. Research suggests this is possible at a relatively low level of allocation (Morgenstern 2002). The use of auction revenues could strongly affect the net social welfare implications of an emission trading program, analytic research has produced results that very strongly suggest that using a grandfathered approach would tend to make the economy less efficient, but that appropriate use of auction revenues to offset distorting taxes (e.g., labor taxes) can reverse this problem and make an emission trading program efficiency-enhancing.

Another important consideration is that examination of the emission trading programs discussed above, and others, is that errors in the estimates of historical emissions is often brought to light. This is due to the new importance assigned to estimates of historical emissions in their role as the basis for allocation of valuable allowances. (Dudek and Palmisano 1988; Tietenberg 2003). In addition, policy goals can change over time, leading to changes (often decreases, or accelerations in decreases) in allowed emissions. Thus, emission trading programs will often feature evolving caps, not fixed ones.

Opt in

An opt in provision has potential to expand a GHG trading system, to introduce low cost emissions reductions, and to provide innovation in reduction technologies and practices. In practice, however, these provisions have either been little used or exploited by firms that would have seen their emissions decline anyway under a "business as usual" scenario, a problem known as "adverse selection." In the OTC NO_x Budget, the opt in provision was virtually unused, mainly because the costs of opting in, including monitoring and reporting costs, tended to be larger for many sources than the potential value of the NO_x allowances. This was especially true for smaller sources. If designers of cap-and-trade systems want to encourage opt in, greater attention needs to be paid to the costs of participation. In the Acid Rain Program for SO_2 trading, many opt-ins occurred, but these were mostly due to adverse selection, and while they had little effect on reducing allowance prices, they led to higher emissions than would have occurred without the provision (Montero 1999).

Related to opt in is the concept of an auction by which non-regulated sources bid for a share of government funds in exchange for an amount of emissions reductions. Winning bidders join a GHG trading system and are effectively paid for meeting the level of emissions reductions that was pledged. This approach was used to launch the United Kingdom GHG emissions trading system in 2001. It was coupled with a regulatory incentive that provided relief from a carbon tax for companies participating in the trading program. For U.S. states, it is unclear whether they have the resources or the will to use public funds and taxation to create such incentives.

Recently the Russian President and Duma (Parliament) have indicated they would ratify the Kyoto Protocol, which raises the issue of how U.S. states could participate in the international trading provisions. This question cannot be answered definitively at this point, because the Kyoto Protocol and subsequent agreements at various Conference of Parties (COP) meetings have not addressed the issue of sub-national participants. The Parties to the Kyoto Protocol could in theory develop provisions for participation in the Kyoto Protocol, but such participation would likely require states to agree to provisions very similar to those of the Kyoto Protocol. Because the United States is not a Party to the Kyoto Protocol, it could have no role in proposing or advocating for such special provisions, except through other countries. California would be in a similar position. In addition, international treaties and law generally presume actions by sovereign governments, not sub-national bodies, so creating a mechanism for allowing California or other states to participate would be timely, if in fact it could be achieved at all.

Infrastructure: emissions monitoring, reporting, and allowance tracking
A critical support structure for any emissions trading program includes emissions monitoring and reporting, and tracking of both emissions and allowances. In all these cases, a government body handled these functions by administering an emissions monitoring system in parallel to an allowance tracking system of some sort, which maintained the allowance accounts and recorded allowance transfers.

Owners and operators of regulated sources were required to monitor and report emissions for each affected unit, though the methods varied to some degree based on the type of facility. Most large EGUs used a certified CEMs, while industrial units and smaller EGUs generally did not employ CEMs but were given several options for monitoring emissions, such as periodic stack testing combined with fuel flow data. For these sources, emissions reports were due to the EPA by October 30, one month after the end of the ozone season.

5.2 Coordination with Other Policies

While the focus of this analysis is on the application of the OTC NO_x rules to a potential multi-state GHG trading system, a number of other policies are relevant and can have an impact on the effectiveness of an emissions trading program. In particular, in the development of a GHG program, there are potential synergies and conflicts arising from simultaneous policies dealing with renewable energy, electricity deregulation, and energy supply security, as follows:

- **Renewable portfolio standard:** California has long been a leader in promoting renewable energy, both through actions by California Energy Commission (created in 1974) and legislation dating back to the 1976 Small Power Act (Hirsh 1999 p. 94). More recent examples include the 1996 restructuring legislation, Assembly Bill 1890, which contained provisions such as the Renewable Energy Fund (Wiser et al. 2002). In the future, the most important renewable energy policy is likely to be the renewable portfolio standard (RPS) implemented by Senate Bill 1078, which was signed into law in September 2002.

An RPS requires power providers to acquire a certain percentage of their supply from renewable resources, and one of the drivers for RPS policy is reduction of air pollution. Depending on policy design, an RPS may take a market approach that allows for trading of “renewable energy certificates,” or RECs. One REC is created for each MWh of renewable generation, and regulated power providers can trade these RECs to satisfy their compliance obligations. Additional demand for RECs stems from businesses, municipalities, universities, and other energy consumers, leading to the formation of a “voluntary market” (Hanson and Van Son 2003).

California’s RPS is the most aggressive such policy in the United States, where about a dozen states have implemented such standards (e.g., Texas and New Jersey) (Golden 2003). Under this policy, by 2017 at least 20% of California’s electricity supply will come from renewable sources, under a schedule that should yield a percentage point increase each year.

In relation to emissions markets for pollutants or GHGs, a question emerges as to whether the holder of a given REC can ultimately attain ownership of any emissions reductions that may be attributable to the renewable generation, perhaps leading to the award of emissions credits. One review of nine state and four national RPS programs notes that all of these jurisdictions continued to use other policy tools to support renewable energy even after their RPS took effect, and that at least two (Denmark and the Netherlands) used environmental taxation as well (Berry and Jaccard 2001). However, this paper does not discuss the interaction in any detail. To date there has been no analytic study of the interactions between emission trading and RECs.

The significance of potentially awarding emission-reduction credits in addition to RECs is that they would be a new revenue stream that could allow project developers to finance new renewable generation. The emissions trading markets and REC markets would be linked, potentially enhancing the rate of market penetration of clean renewable technology. Policy makers would need to solve a number of complex issues, however, including the ownership of emissions reductions from renewable generation, the emissions reductions that are (or are not) represented by a given REC, and the rules for exchanging RECs with emissions credits.

The prospect of linking emissions and REC markets raises the question of whether a single policy approach is simpler and more effective. For example, would an emissions trading market, designed in such a way as to provide a powerful incentive for renewable energy, make an RPS superfluous? The answer to this question would have to take into consideration the desired level of renewable generation, based not only on the clean air goals but other policy objectives such as energy diversity, coupled with an analysis of which policy approach, either emissions trading or RPS or a combination, achieves the goal on time and at least cost. While such an analysis is beyond the scope of this paper, suffice it to say that an emissions market with sufficient stringency in the cap could eventually make an RPS unnecessary, but, in the near term, state RPS policies are providing valuable information and lessons about their effects on renewable power generation, and are therefore useful. It may be that RPS and emissions markets both effectively stimulate renewable power generation, albeit by different buyers and sellers in different parts of the electric power market.

- **Electricity deregulation:** During the period in which NO_x markets were being established, the United States was also undergoing significant market reform in the electric power sector. Deregulation was being promoted as a way to increase competition in the industry, promising both reduced prices and increased innovation in the sector. A key focus of electricity deregulation was opening the transmission lines to all electricity producers. As discussed in Section 2 relative to leakage, this had the effect of allowing high-emitting NO_x sources, often outside of the local power producing region, to compete with low-emitting NO_x sources. Given the initially high NO_x prices, there was concern that power generation facilities would close in the OTC region with concomitant job losses. This did not transpire. The implications for GHG markets, however, may depend on the relative costs of GHG control technology as a function of electricity costs. With CO_2 prices at, for example, \$5 per ton of CO_2, coal-fired power plants might see costs rise by approximately \$5 per MWh—a significantly larger increase than that from NO_x controls.

  However, it is not clear how this increase in *costs* will affect *prices* in competitive markets, or how it will affect the actions of electricity generators. Coal-fired power plants rarely set electricity prices since they tend to have fairly low costs compared to gas-fired plants. An increase in the cost of (not the price for) coal-fired power will affect shareholders, not consumers. Studies that have modeled relatively modest CO_2 restrictions on electric power systems tend to forecast slight changes in the utilization of existing capital (less coal, with more gas and renewables) with similar changes in new generation investment (Johnson and Keith 2004). Given the cost and public opposition to increased electricity transmission, it is hard to see how the location of power generation would change very much, even in a deregulated system, thus limiting the potential for emissions and/or economic leakage. Moreover, changes in the long-term path of technological innovation, which are arguably the most important outcomes of deregulation, are far less clear. In theory, firms in deregulated electric power markets will innovate faster and better than traditional monopoly franchises, though this is yet to be demonstrated in practice.

- **Energy supply security:** Over the past several years, the issue of energy security has risen to considerable political prominence (Farrell et al. 2004). Concerns over diversity of supply, electricity grid failure, and reliability of individual power sources have all received much attention. Inasmuch as the consequences of environmental regulation had little effect on the power generation mix, there has been little interaction between the issues of energy security and NO_x. Even in the most extreme case in RECLAIM, careful analyses of the interaction of California's electricity crisis and emission trading have been found to be unrelated (Joskow 2001; Israels, Grow et al. 2002; Kolstad and Wolak 2003; Farrell 2004). Rather, this research shows that problems in the air pollution program were used as an excuse to try to justify illegal manipulation of the electricity market, and the problems of the air pollution program were due, in part, to a failure of the regulated sources in the electricity sector to install the necessary control equipment, despite obvious information and warnings.

Moreover, an analysis of the reliability impacts of the NO_x SIP Call by the organization charged with ensuring electrical reliability in North America indicated no major challenges (Reliability Assessment Subcommittee 2000), and reports documenting the sources of reliability problems in North America do not mention environmental regulations or controls as important reliability issues (North American Electric Reliability Council 2001). Indeed, the major blackout that occurred in August 2003 in the United States and Canada was due to inadequate maintenance of the electric power grid and control systems, improper system operation, and substandard training on the part of the relevant firms (Amin 2003; ELCON 2004; U.S.-Canada Power System Outage Task Force 2004). No evidence has been presented suggesting that environmental regulations played any significant role in causing the 2003 blackout.

- **Potential GHG taxes:** It may not be feasible or desirable to implement a single instrument for climate policy, some combination may be desired instead. A "sectoral hybrid" approach has been suggested for the United States, in which automobiles and appliance standards would be regulated by tradable standards (like the current CAFE standard), while electricity generators and other stationary sources would be regulated with a C/T system (Nordhaus and Danish 2003). This proposal also includes the idea that this hybrid program could be transformed into a more comprehensive C/T system aimed "upstream" at fossil fuel suppliers. However, there is no relevant experience (save some unanalyzed cases in the last few years in Europe) and no analytical research into such combinations or transitions. Therefore, it is unclear about how such a program might best be developed. Two key principals are indicated, however.

  First, if an emission trading program is used, a C/T system is generally preferable to an ERC system, because a C/T system is likely to lower costs more, and provide firms with greater flexibility. Second, if a C/T system is used, distribution of allowances for free to existing firms (grandfathering), as has been done in the past is likely to be both less efficient and less fair than auctioning the allowances off. Grandfathering GHG allowances could impose higher costs (on a percentage basis) on the lowest income households in the state while making the richest households better off. Auctioning the allowances would create significant revenues for the state, and the use of these revenues is critical to the efficiency and equity properties of the policy. For instance, using allowance auction revenues to decrease corporate taxes or capital gains taxes would have the same regressive characteristics as a system of grandfathered allowances. Generally, the best outcomes will come from using these revenues to reduce pre-existing distortionary taxes, especially payroll and income taxes. The greatest equity improvements would probably result from equal lump-sum payments to all households.

  The same issues of how to use the revenues would arise under a tax on GHG emissions, and the analysis is essentially the same; "recycling" these revenues in ways that reduce the effect of distortionary taxes can enhance both efficiency and equity. In principal, there is no reason why both a tax and C/T system could not be used at the same time. However, it is not clear how a tax and C/T system would interact if they were applied to the same sector, nor is there any obvious way to design a transition from one system to another, based on current understandings.

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