Property in Land and Other Resources

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Rights to Pollute

Assessment of Tradable Permits for Air Pollution

NIVES DOLŠAK

Scholars often attribute deterioration of natural resources to ill-defined property rights and suggest privatization and tradable-permit markets as a solution to environmental problems. Environmental resources differ in physical characteristics and use patterns, and policies regulating their use must be tailored to these differences as well as to resource user characteristics. This chapter draws on Dolšak (2007) to examine factors that contribute to well-performing tradable-permit markets for reducing air pollution. Comparative analysis of market performance suggests that tradable permits are not successful in all cases. Much analytic work has identified individual sources of transaction costs that have led to market failures. However, a holistic framework that looks at key sources of costs enables one to find cases of successful use of tradable permits where partial analytic frameworks would have predicted failure. For example, nonuniform pollution cases (cases where emission of one unit, e.g, a ton of pollutant, causes different pollution outcomes across space) should have high exchange costs and a low level of trading, but appropriate design of a tradable-permit system can reduce these challenges.

The problem of air pollution dates to at least the nineteenth century. Indeed, from the 1860s to the 1880s, many cities passed ordinances to regulate smoke and odor (see chapter 5 by Daniel Cole in this volume). At the federal level, the key statutes were not enacted until the 1970s. The Clean Air Act Amendments (CAAA) of 1970 mandated that the newly established Environmental Protection Agency (EPA) list substances causing local air deterioration and set maximum allowable concentration levels for these pollutants to avoid health hazards and destruction of property.[1] Consequently, the EPA established the National Ambient Air Quality Standards (NAAQS) for six major criteria pollutants: ozone, carbon monoxide (CO), nitrogen oxides (NO_x), sulfur dioxide (SO_2), particulate matter (PM), and lead. The EPA requires states to develop and implement policies to meet NAAQS guidelines. If the ambient quality in an area is worse than the standard set by the EPA, the area is designated as "nonattainment." States with nonattainment areas must submit a state implementation plan (SIP) to the EPA indicating how they will meet the

[1] For a detailed discussion of the legislative and judicial processes involved in the Clean Air Act and its amendments, see chapter 5 by Daniel Cole.

standards in the future. The sanction for noncompliance is economic: nonattainment areas can be denied permits to construct new facilities unless they come into compliance with the NAAQS. Additionally, in some cases, the EPA may not accept the SIPs and may decide to administer the clean air policy itself (Crotty 1987; Lester 1986).

Despite four decades of federal air-quality regulation, some areas continue to suffer from serious air pollution. In April 2011 a large number of counties in southern California failed to meet the NAAQS for several criteria pollutants, and areas in Nevada, Montana, Arizona, midwestern states, and the northeast states failed to meet NAAQS for one or two pollutants. (For detailed county level information, see the EPA Green Book information at http://epa.gov/airquality/greenbk/mapnpoll.html.) Although lead is no longer a significant pollutant in the United States, many states continue to find it impossible to meet standards for ozone and PM, and several still exceed standards for SO_2.

The EPA's Air Quality Index (AQI) measures health hazards of air pollution above certain levels. AQI values above 100 indicate levels of air pollution that pose risk to sensitive populations with respiratory and coronary problems. AQI values above 150 indicate pollution levels that are unhealthy for anyone undertaking extensive outdoor activities. In 2008, the most recent year for which data are available for the entire United States, air quality in a large number of counties was at a level that was unhealthy for sensitive populations. As depicted in figure 6.1, most of southern California, central and southern Arizona, and many areas in the Midwest, Florida, the Atlantic coast, and the Northeast experienced AQIs above 100 for more than 10 days a year.

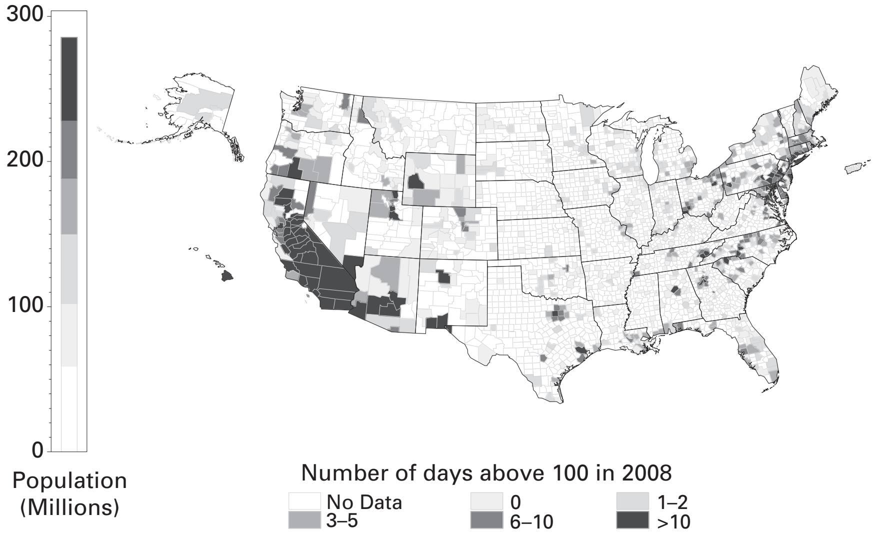
A comparison of data for 1998 and 2008 gives a sense of changes over time. The data for 1998 suggest that air quality was significantly worse 10 years earlier in the eastern United States, the Midwest, the southern states, and states along the Atlantic coast (figure 6.2). Although air quality has significantly improved in the eastern United States since 1998, several western states have not seen such improvement.

In 2008, 52.4 million people lived in areas where the 90th percentile value of annual AQI exceeded 100, and 31.9 million lived in areas where the 90th percentile value of annual AQI exceeded 150, the value at which air becomes unhealthy for outdoor activities for anyone. Although these numbers are high, notable improvement had occurred since 1998. In that year, 111.1 million people lived in areas where the 90th percentile value of annual AQI exceeded 100, and 21.4 million lived in areas where the 90th percentile value of annual AQI exceeded 150.[2] The key point here is that the increase in the number of people living in the most polluted areas is the result of population growth (Riverside–San Bernardino and the Los Angeles Metropolitan Statistical Area) rather than pollution increases.

What is local often becomes global. Winds transport pollutants across countries and continents. Recent data suggest that some toxic pollutants cross oceans and travel around the globe. Similarly, the use of certain chemicals, such as ozone-depleting

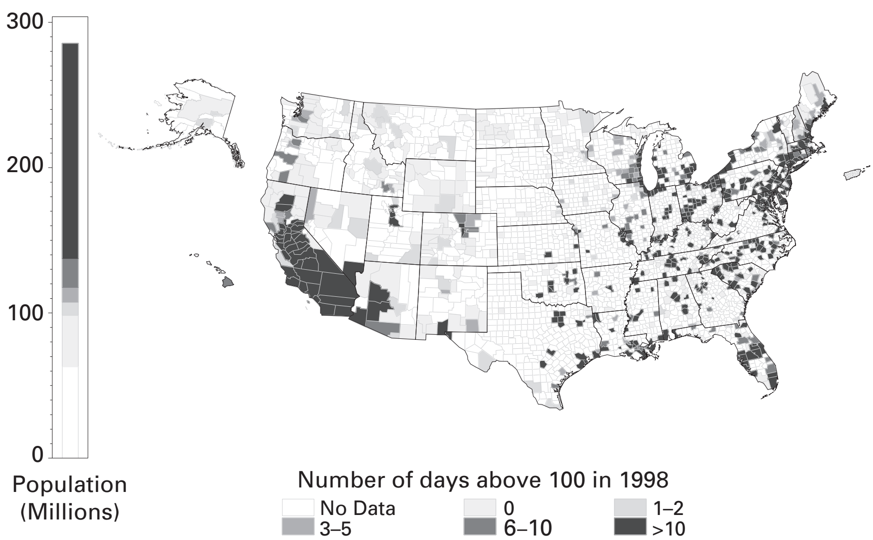
[2] These estimates are based on data on air quality and population for metropolitan statistical areas, available at the EPA (http://www.epa.gov/air/data/monaqi.html?us~USA~United%20States) and the U.S. Census Bureau (http://www.census.gov/prod/3/98pubs/98statab/saappii.pdf and http://www.census.gov/population/www/metroareas/metroarea.html).

FIGURE 6.1
Air Quality in U.S. Counties in 2008



SOURCE: Environmental Protection Agency, http://www.epa.gov/air/data/msummary.html?us~usa~United%20States.

FIGURE 6.2
Air Quality in U.S. Counties in 1998



SOURCE: Environmental Protection Agency, http://epa.gov/airquality/greenbk/mapnpoll.html.

substances, has led to deterioration of the atmospheric shield that protects the earth's surface from harmful radiation. Since the late 1970s, scientists have measured a hole in the ozone layer over Antarctica where up to 66 percent of the ozone is depleted. The Vienna Convention for the Protection of the Ozone Layer and its subsequent protocols since 1989 devised a phaseout schedule for ozone-depleting substances.

This regime has been fairly successful. The United Nations Environment Programme (UNEP) estimates that the total consumption of a key group of ozone-depleting substances, chlorofluorocarbons (CFCs), dropped from 1.1 million tons in 1986 to 35,000 tons in 2006.[3] Global problems require international cooperation in developing, monitoring, and enforcing institutions for pollution reduction. As it would be extremely difficult to exclude anybody from benefitting from reduction in global pollution, incentives for free-riding are significant. Although the institutional challenges of devising and enforcing effective international regimes for air pollution are beyond the scope of this chapter, the use of tradable permits in the regime protecting stratospheric ozone is reviewed here and is compared with national and regional tradable-permit systems.

Governments have historically developed policies to reduce air pollution. First, governments have regulated the use of the atmosphere as a pollution sink by prescribing technologies polluters must employ, such as catalytic converters for vehicles and scrubbers to clean up exhausts of power plants, or by setting minimum requirements for the use of renewable sources of energy in electricity generation (the so-called Renewable Portfolio Standards). Second, governments have enacted policies requiring firms to provide information to consumers about the environmental impact of their products and production processes. For example, many U.S. states require electric utilities to provide information on the portfolio of energy sources used in electricity generation. The idea is that informed consumers will vote with their dollars to reward firms that minimize environmental harm (Dietz, Ostrom, and Stern 2003). Finally, governments have created market incentives, such as taxes, fees, and tradable quotas, to alter the benefits and costs of air pollution for individual actors. This approach has drawn much attention, especially in the context of the ongoing global climate change debate.

If ill-defined property rights are an important cause of the overuse of environmental resources and of negative externalities (Coase 1960; Cornes and Sandler 1996), defining property rights better may solve the externality problem. Dales suggested that "it is high time that we began to devise some new forms of property rights, not to air and water, but to the *use* of air and water" (1968, 76). Because "a property right is enforceable authority to undertake particular actions related to a specific domain" (Commons 1968, cited in Ostrom and Schlager 1996, 130), one expects that property owners will design institutions to prevent resource overuse and deterioration.

Privatization proponents suggest that when property rights are well defined and easily enforced, markets efficiently determine what and how much should be produced (by means of market prices), how it should be produced (through the relationships between marginal productivities of inputs and their prices), how it should be distributed (depending on individuals' income and preferences), and how consumption should be allocated over time (through differences in individuals' discount rates).[4] In addition, tradable permits have dynamic advantages over command-and-control instruments, the alternative to privatization. Various studies suggest that

[3] UNEP, http://ozone.unep.org/Events/ozone_day_2008/press_backgrounder.pdf.

[4] Although this chapter focuses on the effectiveness of tradable permits for reduction of air pollution, fairness in access to the resource may be more important than effectiveness and/or economic efficiency. The right-allocation

market-based systems create greater incentives for technological innovation and diffusion than command-and-control instruments (Jaffe and Stavins 1995; Montero 2002).[5]

However, empirical analyses of individual cases suggest that this broad endorsement of privatization is too optimistic. Tradable-permit markets have been found to be thin and to have high transaction costs (Gangadharan 2000; Hahn and Hester 1989a; 1989b). Data problems have impeded monitoring and enforcement of trading rules (Coy et al. 2001; EPA 2002; Wilkinson and Thompson 2006). Scholars question whether tradable-permit systems stimulate innovation (Driesen 2003; Montero 2005) and have the ability to respond to sudden and substantial changes in the market (Coy et al. 2001; EPA 2002). About 40 years after the implementation of the first tradable-permit market in the United States, researchers are somewhat careful about endorsing tradable-permit markets as a universal approach to solve the pollution problem: "All of our analysis suggests one final observation: Experience with and lessons learned from the Acid Rain Program must be applied with care to other environmental objectives" (Ellerman et al. 2000, 321).

Privatization is a complex undertaking. Scientifically uncertain and politically sensitive issues must be resolved before there is any allocation of individual rights among individual users. For example, what total level of resource use can prevent deterioration of future resource stocks? How many fish can be harvested without jeopardizing future stocks (Gordon 1954)? How many tons of SO_2 can be emitted without causing significant damage to physical and biological environments in areas downwind from the pollution? The ability of science to provide answers to such questions significantly diminishes as the complexity of the environmental resource or pollution problem grows.

Scientific estimates become the basis for a government's decisions on what the limits on resource use should be. These decisions, however, are influenced not only by scientific information, but also by political factors. Some environmental resource users may be able to influence the decision on the overall level of the use of a resource, how it should be allocated among current users, and how future users should obtain permits to access and use the resource. Title IV of the CAAA followed more than a decade of political struggle over how SO_2 emissions could be reduced (E. M. Bailey 1998; Burtraw and Palmer 2004; Ellerman et al. 2000).

Arguably, monitoring and enforcement rules might be easier to design for tradable-permit systems for environmental resources that extend across a smaller geographic area and involve a smaller number of resource users (Rose 2001; Tietenberg 2001). On the other hand, a larger area and, therefore, a larger number of resource users may create incentives for specialized brokers to enter the market, thereby reducing transaction costs and increasing trading activity in the market for tradable permits.

process takes place in the political arena and may have redistributional consequences (McCay 2001). It therefore deserves special attention.

[5] Although emissions trading might create more incentives for innovation than technological standards for the regulated industries, technological standards can encourage innovation in pollution-control industries. Hence, which policy approach can encourage innovation across industry types remains an open question. For an empirical analysis of the impacts of the 1971 New Source Performance Standards on patent activity, performance, and costs of emission-control technologies in emission-control industries following the introduction of tradable permits in 1990, see Taylor, Rubin, and Hounshell (2003).

Thus, there is no definitive answer regarding the optimal market size or user base for the functioning of any environmental resource tradable-permit system.

Resolving these issues requires the development of an analytic framework that enables a comparison of different tradable-permit regimes and their effectiveness, controlling for factors that are external to the regime's design, such as characteristics of the resource, patterns of its use, and characteristics of resource users. The following section outlines an analytic framework to compare effectiveness of tradable-permit regimes. This framework is then employed to compare the tradable-permit performance of four U.S. tradable-permit regimes.

The Analytic Framework

A growing body of literature on tradable-permit markets identifies several factors that affect their performance. Although empirical studies since the 1970s have analyzed the performance of tradable permits and other transferable quotas, these analyses have mostly focused on a single environmental resource, with a few notable exceptions, such as Sorrell and Skea (1999) and Tietenberg (2006).[6] Little systematic effort has been invested in developing comprehensive frameworks that would "identify the universal elements that any theory relevant to the same kind of phenomena would need to include" (Ostrom 1999, 40).

Two approaches to comparative work are noteworthy. The first is often used when an existing tradable-permit design is used as a template for designing a system for another pollutant (e.g., the Acid Rain Program for a carbon dioxide allowance market). This approach focuses on similarities and differences between these environmental resources and/or pollution issues (P. Bailey and Jackson 1999; Ellerman et al. 2000; Farrell and Morgan 2003; Schmalensee 1998). There is no clear guidance, however, about what characteristics of the environmental resource and its users have to be included in such an analysis and how they will interact. An alternative approach provides a list of factors that affect the performance of tradable-permit systems, including characteristics of the tradable permit (timescales, banking, and allocation), spatial characteristics of the resource (local versus global impact of resource use), enforcement and monitoring, and size and knowledge of regulatees. This approach, however, does not specifically address how these variables are interlinked (Sorrell and Skea 1999).

The problem with these frameworks is not merely analytic, but also practical. Recommendations from such partial frameworks can lead policy makers to devise a tradable-permit system that addresses the impact of a subgroup of factors, not recognizing that they may be sacrificing critical aspects of the tradable-permit design and, thereby, performance. For example, when a greenhouse gas tradable-permit design follows a frequent recommendation that regulatees be large, point-source emitters, it focuses on fossil-fuel-burning electricity generators. Although focusing on large stationary sources that are easy to monitor may be a plausible start for many developed economies, thereby signaling the willingness of the major emitters to

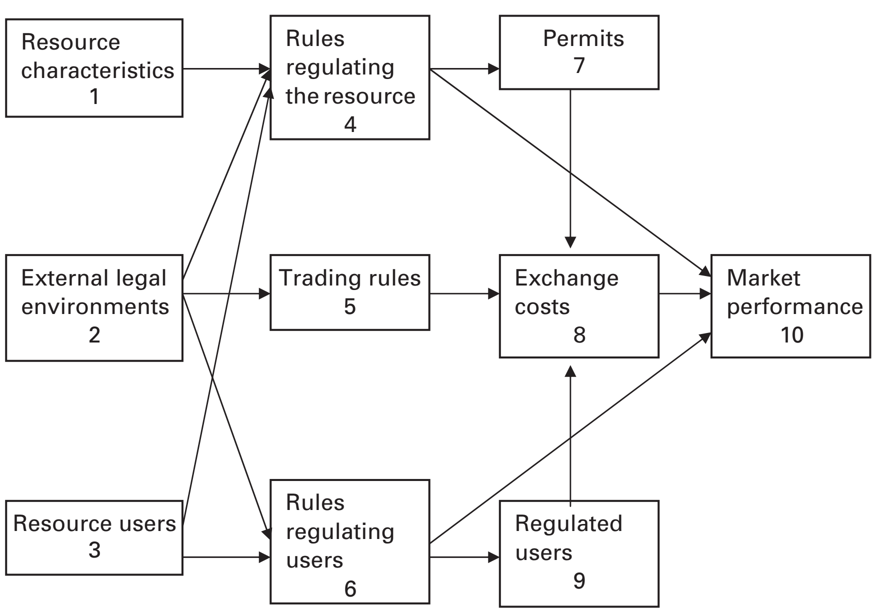
[6] For an excellent example of a study of the Acid Rain Program, see Ellerman at al. (2000).

curtail their emissions, this approach misses the fact that in some areas, the largest sources of greenhouse gas emissions are not electricity generation, but deforestation, waste management, and/or transportation. It is important, then, that a framework be developed that incorporates all major factors affecting market performance so that the limitations of tradable-permit designs can be recognized and supplemented with other regulatory approaches.

The analytic framework used in this chapter seeks to overcome the limitations of partial frameworks. It is based on the comparative literature on common-pool resources (Dietz, Ostrom, and Stern 2003; Ostrom 1990). It also builds on the literature on transaction costs and the factors affecting market performance. Performance of a tradable-permit system depends on the ability of permit holders to trade permits at low cost and of system administrators to monitor and enforce rules (Cason and Gangadharan 2003; Ellerman at al. 2000; Gangadharan 2000; Hahn and Hester 1989a; 1989b; Tietenberg 2006). Therefore, the analytic framework applied here augments the institutional analysis and development (IAD) framework developed by Elinor Ostrom in her influential book *Governing the Commons* (1990) by incorporating elements of the transaction-costs literature. The IAD framework is used because it focuses on how resource characteristics and resource use patterns affect institutional design and therefore performance.

The following groups of factors are identified as affecting market performance (figure 6.3):

FIGURE 6.3
Factors Affecting Performance of Tradable Permits



1. External factors, such as resource characteristics, external legal and regulatory environments, and characteristics of resource users (boxes 1, 2, and 3).
2. Rules regulating resource use and users (boxes 4 and 6).
3. Characteristics of the tradable-permit markets, including trading rules, permits, and exchange costs (boxes 5, 7, and 8) and their participants (box 9).

Resource Characteristics

Resource characteristics (box 1 in figure 6.3) include predictability of the resource stocks (i.e., air quality), availability of reliable indicators of resource flows (emissions), spatial extent of the resource, and effects of use of the resource on resource stocks (uniform versus nonuniform effects). If predictability of the impact of emissions on air quality is low, the rules need to be easily adaptable as new information becomes available. However, frequent changes in the rules make property rights embedded in tradable permits insecure and thereby reduce the incentive for right holders to curb their resource use with the objective of generating a surplus to be traded.

Further, if there are no reliable indicators of resource flows (e.g., emissions of air pollutants from vehicles), monitoring and enforcement become more difficult. However, this obstacle can be addressed by relying on measurements of the use of related resources. For example, if continuous measurements of emissions are not easily obtainable or reliable, one could measure the amount of combusted fossil fuels and thereby estimate pollution using emission coefficients. Consequently, one could devise permits for the levels of inputs into the resource-using activity (e.g., sulfur or carbon content of fuels) or the outputs from the process (e.g., kilowatt hours of electricity generated or barrels of gasoline produced in a refinery).

The spatial extent of the resources is crucially important. If the resource expands over large areas (e.g., the atmosphere or the oceans), monitoring becomes difficult and expensive. It is difficult to relate changes in the stocks of large, dispersed resources to different resource use patterns and to estimate the appropriate level of emissions. For example, when air pollution travels across large areas, computations of the impact of local versus regional or global emissions on air quality require extensive air-pollution modeling and data monitoring. In estimating the impact of the resource use rules on resource stocks, one has to separate so-called background pollution (pollution from upwind sources) from pollution by local sources. When the Houston area in Texas was not meeting the NAAQS in the early 2000s, arguably because of the transport of air pollution from Mexico, the EPA granted it an extension of two years to meet the standards. Such adjustments can decrease the pollution-reduction efforts and affect the environmental outcome and trading activity.

Another problem arises if the effects of resource use by different users are not uniform, and, therefore, the property rights cannot be traded on a unit-to-unit basis (Baumol and Oates 1975). For example, if the effect of a ton of emitted pollutant in the coastal area near Los Angeles is not the same as the effect of a ton of emitted pollutant in the mountains, permits for these pollutants cannot be traded one-to-one. Rules have to be created that reflect varying environmental effects of pollution from different locations and enable their comparability. Alternatively, trading can be allowed only in subareas with uniform impact. Consequently, air-pollution measurability is highest where a unit of pollution has the same impact on air quality no matter where it is emitted, for pollution that spans small areas (local rather than regional or global air pollution), and where emissions of each polluter can easily be measured or reliably estimated.

Resource Users

Resource users (box 3 in figure 6.3) can be few or many. If a small number of resource users appropriate a large proportion of the resource, the institutional arrangements regulating resource use are more likely to be developed and well functioning. A small number of users may find it easier to solve the collective-choice dilemma and craft institutional arrangements to manage the resource. However, even if the institutional arrangement is imposed by an authorized agency, it is easier to monitor a smaller group because transparency of the resource use is higher, and thus, the need for complicated monitoring is reduced. For example, in the case of the CFC production-quota trading system, the number of companies capable of manufacturing CFCs was small. On the other hand, if the number of users is small, the potential for reducing costs through trade decreases.

The number of resource users may be directly related to the spatial extent of the resource. A global resource, such as the atmosphere, would be expected to have a large number of widely dispersed resource users all over the world. This is indeed true of many forms of global pollution, such as ocean pollution or emissions of greenhouse gases. There are, however, also cases where the use of the common pool resource (CPR) requires special technology that is available to a relatively small number of users. Thus, the two variables are not perfectly correlated.

Rules Regulating the Resource

Three types of rules regulating the resource use (box 4 in figure 6.3) are of particular importance: rules determining the severity of resource use limitation, rules requiring adjustments in the severity levels in response to external shocks or newly available information about the resource, and rules regulating how the resource is transformed into a private good (permit). These rules affect the actual environmental effectiveness of the permit system, as well as the level of trading (the value the permits hold and the security of the permits).

As outlined in figure 6.3, resource characteristics influence resource rules. Low measurability of resource stocks or flows requires that the rules be flexible to allow for adjustments in light of new information, and this flexibility potentially reduces the security of the permit for the holder.

Rules on the severity of the resource use limitation prescribe how much resource use must be reduced. The severity of the resource-management targets can significantly affect the success of the institutional arrangement, both in environmental effectiveness and in trading incentives (EPA 2002; Hall and Walton 1996), but in which direction is unclear. Theoretical literature and empirical results for the Regional Clean Air Incentives Market (RECLAIM) in southern California (SCAQMD 2007) suggest that if the targets are not restraining, they have a small environmental effect and also lead to low permit prices and a low level of investment in pollution-reduction technologies. However, empirical work presented elsewhere indicates that severity alone is not sufficient to spur high trading (Dolšak 2007).

Two broad types of rules regulating how the environmental resource is transformed into a private good have been implemented: case-by-case systems, and standardized systems depending on an emission baseline used in their definition and the actors who can participate in the market-based exchange. The case-by-case or emission-reduction credit (ERC) system is based on individual emission standards. Each polluter is required to adhere to a given emission standard, defined as emissions by unit of input or unit of output. The ERC regime does not control the aggregate emissions; it focuses on standards for emissions per unit. The total emissions depend not only on the emission coefficient but also on the level of input (output). Because input is controlled by the polluters, the regulators cannot determine the aggregate level of emissions. Therefore, this system is less environmentally effective than the emission allowance system. In this system, the right to an emission-reduction credit is issued only when the emissions are recognized by the authorized agency. Everyone with a capacity to reduce resource use can participate in this market.

The emission allowance system entitles the user to pollute the air up to the maximum level allowed by the permit. This system requires that an authorized agency determine the total level of pollution for a given period and then allocate the aggregate among the polluters included in the permit system. This system (e.g., the SO_2 allowance system authorized by the CAAA of 1990) is based on setting the maximum resource use rights (total emission allowances) for a particular period, dividing that amount among polluters, and enforcing that an individual polluter does not emit more in that period than the allowed amount. An individual polluter can purchase additional allowances for that period from other polluters whose emissions are lower than the initially allocated allowances. Thus, although every polluter holds a permit to emit an allocated amount of the pollutant, if the allocated amount is not used in the period, it can be sold to other regulated polluters or kept for future use.[7] The level of pollution from the regulated polluters is limited by the maximum amount of allowances.

Each system has advantages and disadvantages. The emission allowance system sets the maximum level of pollution and reduces transaction costs because the traded permit is standardized (a given emission allowance) and issued beforehand. However, a major disadvantage of this closed-market system is that it requires that regulators have knowledge about the pollution problem, especially the sources and acceptable levels of pollution. Also, there must be political consensus about the initial quota allocation. This system also creates barriers for new entrants. (These barriers are one way to secure the political support of existing polluters.)

The ERC system, on the other hand, allows everyone to participate in market exchange once emission reductions are certified. However, a key finding from the initial EPA emissions-trading programs has been that the quantification and cer-

[7] Not all tradable-permit systems, however, allow banking for future use. RECLAIM permits, for example, expire at the end of the compliance year for which they were issued.

tification of the ERCs can require much time and effort, increase exchange costs, and impede trading.

Rules Regulating Users

The most important rules pertaining to resource users (box 6 in figure 6.3) are those that determine who is regulated and how the nonregulated resource users can opt in or become regulated at a later time. If the proportion of resource users that are not regulated is large, these markets may not be environmentally effective. Rules determining which resource users are regulated must be crafted carefully because they may result in significant leakage, that is, a shift of CPR use from regulated to nonregulated users. The Clean Development Mechanisms defined in the Kyoto Protocol are particularly prone to this problem (Richards and Andersson 2001). A developing country without a national emission-reduction target does not have a mechanism to measure whether the pollution reduced at a particular site has leaked to a different project. Therefore, it is necessary to identify the largest resource users and examine how their activities might shift. In some cases, rules can be created to prevent this shift; in others, this may not be possible at low cost and in a short period of time. This shift can be regulated by specific rules stipulating the requirements that nonregulated users must meet. These characteristics must also be closely monitored.

The second challenge is the decision to allow some resource users to opt in for regulation. This decision again must be carefully considered, and the permits must be allocated so that the process does not lead to adverse selection. This clearly was the problem in the SO_2 market, where opt-in units were allocated allowances to undertake emission reductions that they would have undertaken even in the absence of opting in (Ellerman et al. 2000). Permit programs also require that once a polluting facility opts in, it can no longer elect to leave the permit program.

Trading Rules

One of the most important sets of trading rules (box 5 in figure 6.3) determines who can participate in trading. Rules that allow brokers to enter the markets can reduce exchange costs and thereby improve market performance. Brokers do much more than just match potential counterparts. In less standardized traded goods, brokers can provide standardization services (additional measurements that better define property rights) and even insurance services. Once brokers entered the EPA's lead phasedown program and the SO_2 allowance market, trading activity in both markets substantially increased.

Permit Exchangeability and Security

Polluters' incentives to invest in pollution permits depend on the permits' exchangeability and security (box 7 in figure 6.3). Exchangeability has two dimensions: space and time. The more restricted exchangeability is, the lower are the incentives for investing in permits. If resource use is not uniform (its effect on resource stocks varies depending on its location), complex exchangeability rules have to be created.

One unit of resource use at one location cannot be traded for one resource unit at another location because these uses affect stocks in different ways. Policy makers propose several ways to handle this issue (Klaassen 1996). One option is to establish trading rules based not on the flows, but on the effect the flows have on stocks. For example, permits might be allocated to biological oxygen demand, the outcome of pollution, rather than a flow of a pollutant into a water body. A second option is to divide the resource's use rights between two subunits and allow trading only within the two units, but not between them. This option will create two smaller markets, and some economically beneficial trades might not occur because of location restrictions. A third option is to allow resource users to trade between the two subunits but with restrictions on either the stock status or the total flows of resource use. Alternatively, trading can be allowed as long as the total flow of the resource in the subunits is below a threshold level. This creates incentives to use or trade resource permits early and not to bank them because it limits trading over time. All these options have higher transaction costs than single-zone trading, which has been successfully implemented for SO_2 and lead permit trading (Klaassen 1996).

Constraints on exchangeability over time might also reduce the incentives to invest in pollution permits. If permits are issued for a particular compliance year, incentives for polluters to invest in them will be higher at the beginning of the period and will decline toward the end of the period. Arguably, the desired length of the exchangeability period should reflect the level of investment in the pollution-reduction technology and the time required to obtain the permit and install the technology.

Security of permits also increases the incentives for polluters to invest in pollution-reduction technologies with the purpose of generating excess permits, selling them, or banking them for future use. Resource users are most certain that permits will continue in the future if they themselves regulate their use. The external legal and regulatory environments (box 2 in figure 6.3) can grant this authority to them or authorize a regulatory agency to regulate resource use. In many cases, the issue is whether state or federal regulators affect the security of permits. For example, if a state fails to meet the NAAQS, the federal EPA may step in and override the existing permit system. Because permits are not treated as private property, they are rarely protected by the takings laws. Therefore, the security of permits is not certain and can only be estimated by permit holders from the information they have on the past policies of the regulators.

Exchange Costs

High exchange costs (box 8 in figure 6.3) decrease the number of trades and the total trading volume and thereby diminish the potential cost savings of tradable-permit systems. Exchange costs are a function of permit characteristics: the number of issued permits, their denomination, their exchangeability (Hall and Walton 1996), the availability of information, and the existence of past trading among permit holders. Past trades among resource users can reduce current exchange costs if trades produce positive informational externalities.

High exchange costs, however, may not completely prevent trading. Resource users search for options that reduce these costs. These options include trading within firms (if firms consist of smaller units that were allocated permits; intrafirm trade) rather than between firms (interfirm trade), and trading a cluster of pollutants rather than trading each pollutant individually (Foster and Hahn 1995).

Market Performance

Market performance (box 10 in figure 6.3) can be assessed by two indicators: environmental effectiveness and market liquidity. Environmental effectiveness reflects whether and to what extent tradable-permit systems reduce air pollution. Ideally, this indicator would measure the resource stocks before and after the implementation of a tradable-permit system. For example, if a tradable permit has been designed to reduce acidification of surface waters due to SO_2 emissions, the indicator of environmental performance would measure a reduction in acidity levels in surface waters subsequent to the introduction of the permit system.

However, in many cases, data on resource stocks (either before or after the introduction of the permit system) are not available. Therefore, environmental effectiveness is frequently measured in terms of flow*s* from or to an environmental resource. The flows from or to the resource resulting from the regulated resource users are compared with the flows in the absence of the tradable-permit system (Burtraw and Palmer 2004). The difference is then attributed to the tradable-permit system. This measurement is problematic when other policy instruments are implemented concurrently for the same environmental problem.

Although flow measures attempt to isolate program effectiveness, they miss an important aspect of market performance. When a tradable-permit system is adopted to protect an environmental resource, to keep transaction costs low, it must focus on a particular subset of resource users whose flows from or into the resource can easily be measured and monitored and on whom restrictions can be enforced. At the extreme, one could visualize a tradable-permit design that would exclude some resource users if they negatively affected the effectiveness of the market. For example, when prices for NO_x allowances in the RECLAIM market increased by a factor of four beyond what was anticipated in 2000, electric utilities were excluded from RECLAIM. This exclusion of specific market participants significantly reduced the demand for allowances and thereby stabilized the price of allowances, but the pollution merely shifted to resource users outside the tradable-permit system. Although prices stabilized (indicating higher efficiency of the market) and NO_x emissions of market participants were reduced (indicating higher environmental effectiveness of the market), the indicator focusing solely on pollution by the market participants would be an incorrect measure of environmental performance of this tradable-permit market.

In this chapter, environmental effectiveness is operationalized in terms of resource flows (emissions) after the tradable-permit market has been implemented as compared with the resource flows (emissions) before its implementation. In using this measure, all of the caveats discussed earlier should be kept in mind. The value of this variable is coded as low when emissions were reduced by less than 30 percent (RECLAIM [EPA 2002]; early EPA emissions trading); moderate when emissions were reduced by between 30 and 60 percent; and high when CPR use was reduced by more than 60 percent (lead phasedown program [EPA 1995]; ozone-depleting substances production-quota trading [UNEP 1999]).

Market liquidity is the second general indicator of market performance.[8] Market liquidity or trading activity is measured by comparing quantities of the resource flows exchanged in the markets among economically nonrelated entities (quantity of permits traded) with the entire quantity of the resource flow allocated to market participants (quantity of permits issued).[9] The values of this variable are coded as low when fewer than 10 percent of the tradable permits are exchanged in a market (early EPA emissions trading [Hahn and Hester 1989a; 1989b]); moderate when this share was between 10 percent and 20 percent (RECLAIM); and high when this share exceeded 20 percent (lead phasedown program [Nussbaum 1992]; ozone-depleting substances production-quota trading [Lee 1996; UNEP 1999]). Although the coding cutoff points are arbitrary, there is a clear difference between markets with high trading activity and those with low trading activity. Surprisingly, a majority of the analyzed markets have comparable trading activity, with measures of trading varying from 20 percent to about 30 percent.

Tradable-Permit Systems for Reducing Air Pollution

This section examines the factors that affect the performance of four U.S. tradable-permit systems to reduce air pollution. These systems range from local (lead emissions) and regional (NO_x) to global air pollution (ozone-depleting substances). The data used in this analysis come from published empirical case studies on these markets and from regulators, such as the EPA, California's South Coast Air Quality Management District (SCAQMD), and the Department of Energy (DOE).

The Lead Phasedown Program

The lead phasedown program sought to reduce the use of the atmosphere as a sink for lead pollution from gasoline-burning vehicles. By focusing on gasoline producers, the program drastically reduced emissions of lead into the atmosphere. Between 1988 and 1997, the maximum quarterly average lead concentrations decreased by 67 percent. At the end of the program, all gasoline refineries met the standards and required no extensions. Section 218 of the CAAA of 1990 eliminated the last remnants of leaded gasoline in 1996. Today, no area in the United States violates the NAAQS for lead.

[8] Market liquidity is an important precondition for economic efficiency. However, there are exceptions. First, trading may be low if actors choose to bank rather than sell their quotas (Fraas and Richardson 2010). The incentives for banking may be enhanced when there is an expectation that emission-control stringency will increase in the future.

[9] Although some have argued that a larger number of allowances exchanged does not necessarily mean a better-functioning market (Ellerman et al. 2000), others have argued that a low volume of trading indicates barriers to trading and a poorly performing market. Those barriers then have to be identified and removed (E. M. Bailey 1998; Gangadharan 2000; Hahn and Hester 1989a; 1989b; Kerr and Mare, 1997; Lile, Bohi, and Burtraw 1996; Newell and Rogers 2004; Schmalensee 1998).

Lead concentrations closely follow lead emissions. Because of the short residence time of lead, changes in emission levels are followed within a year by changes in concentrations (Brown, Kasperson, and Raymond 1993). This makes it easier to devise the necessary reductions in resource use to accomplish the required effects on resource stocks. Because lead content does not differ across gallons of gasoline produced in the same refinery at any given time, it was measured in samples, and the values were averaged over a three-month production period (the compliance period).

The extent to which air pollution travels depends on the physical state of the pollutant (gaseous, vapor, or particulate), the height of the emission source, and wind speed. Lead does not travel very far. In the past, concentrations were highest in the vicinity of highways. They decrease drastically as the distance from highways increases. Therefore, lead concentration monitoring must occur not only at road sites, but also at a neighborhood level.

Gasoline combustion accounted for nearly 86 percent of lead emissions in 1970, whereas the next-largest contributors, such as primary lead smelting, coal burning, and waste oil combustion, each accounted for about 2 percent of the total emissions (EPA 1984). The lead phasedown program, therefore, focused on gasoline refineries. Three hundred to four hundred oil refineries were included in the lead phasedown program. Gasoline production was fairly concentrated: 25 percent of the refineries accounted for 70 percent of production, while the remaining 75 percent of the refineries accounted for about 30 percent of production (Nussbaum 1992). Modern refineries (mainly located on the west coast) were capable of producing gasoline with 0.7 grams of lead per gallon. Some opting in was allowed, but gasoline blenders played a minor role in the market.

At the peak in the 1970s, the average content of lead in a gallon of gasoline was about 2 grams per liquid gallon (gplg), but the content could be as high as 4 gplg. In 1982 the standard of 1.1 gram of lead per gallon of gasoline was introduced for refineries. The use of lead as a gasoline additive was to be phased out by 1987. During the initial period, from 1983 to the end of 1985, the resource use regulations were not stringent. The actual lead content was significantly below the standard. The standard was tightened on 1 July 1985 (0.5 gplg) and again on 1 January 1986 (0.1 gplg).

Refineries were allocated rights to use lead as a gasoline additive for each quarter of a year on the basis of their quarterly production of gasoline, multiplied by the currently valid lead content standard. With banking (exchangeability of rights across time periods), trading became more active and new actors entered the market. Gasoline suppliers could blend gasoline with alcohol, thereby reducing lead content per gallon and obtaining lead-use permits. At the end of 1984, there were about one hundred blenders. By late 1985, the number grew to nine hundred. Many of them, unfortunately, did not know how to report lead usage, and many trades included rights that could not be claimed legitimately (Nussbaum 1992). Brokers also entered the market, but they merely acted as intermediaries.

Allocated rights were fully tradable within the commitment period. No prior EPA approval was required for transferring the permits. In 1985 banking (exchangeability of rights across time periods) was allowed; refineries could bank unused rights for the last two years of the program (1986 and 1987), when the leaded gasoline would be phased out As a result, trading became more active, and new actors entered the market.

Trading of lead permits did not cause high exchange costs for refineries, which were used to trading with one another. In addition to trading gasoline, they would add the lead rights. For them, trading lead rights resulted in "little more paper work costs than the addition of a contractual paragraph and, perhaps, the price of a stamp" (Nussbaum 1992, 32). Although brokers were allowed to enter the market, they faced the same information problems as the traders. Large traders did not rely on brokers because of high costs; it was mostly small traders who used their services.

The program exhibited active trading and banking, even though no price information was publicly available, and brokers played only a limited role (predominantly for small traders). With the enactment of tighter standards in 1985 and 1986, refineries had to use the banked rights to comply with the standards. From the beginning of the program in 1983 to its end in 1987, between one-fifth and one-third of the facilities purchased lead rights (Nussbaum 1992). Intrafirm trading was also important. Kerr and Mare (1997) report that about 67 percent of trades in their sample, covering the second half of 1983 and 1984, were internal.

The value of lead rights rose with the introduction of banking. Initial prices were about 3.5 cents per gram and subsequently increased to slightly more than 4 cents per gram. Price information was not collected and reported by any entity in the market. The only way to learn the price was to negotiate it with potential trading partners (Kerr and Mare 1997).

Emissions of lead from on-road vehicles declined from about 172,000 short tons of lead in 1970 to about 62,000 short tons in 1980 and to only 1,387 in 1995 (EPA 1995). At the same time, total lead emissions were reduced from about 220,000 short tons in 1970 to about 5,000 short tons in 1995 (EPA 1995). The average lead content in gasoline fell from about 2 gplg in 1973 0.7 gplg in the first quarter of 1985. At that time, the standard was 1.1 gplg. In the third quarter of 1985, the average lead content fell to 0.4 gplg. In the same year, the standard was tightened to 0.5 gplg (Nussbaum 1992). In 1987, at the end of the lead phasedown program, no refinery asked for additional time to comply with the 0.1 gplg standard (Nussbaum 1992).

Early EPA Trading and RECLAIM

This section reviews two tradable-permit systems implemented in southern California: a federal early EPA trading program implemented in the 1970s and the subsequent local tradable-permit system, the Regional Clean Air Incentives Market (RECLAIM), implemented in the 1993 by the SCAQMD.[10]

The following discussion offers lessons about the impact of institutions on environmental effectiveness and market liquidity.[11] First, predefined, standardized emission allowances do not necessarily outperform case-by-case-defined emission-

[10] The SCAQMD was established in 1977 to address the persistent air-pollution problems in Southern California.

[11] Generalization of the SCAQMD trading data to other air-quality-control regions, however, is limited. The SCAQMD exhibited a more active market than other areas, most likely because of high demand caused by rapid development in the area and more stringent classification requirements for major emission sources.

reduction credits in environmental effectiveness when predictability of resource stocks is low and resource use is not severely constrained. Market liquidity, however, improves. The number of trades increases, traded rights represent a larger proportion of resource use, and specialized agents enter the market. However, market liquidity improvements do not occur instantaneously with the shift to standardized allowances, because right holders must take time to learn the new property institutions.

Measurability of resource stocks and flows is moderate in southern California. More than 30 locations throughout the SCAQMD measure air pollution. However, attributing the pollution data directly to any policy is problematic because 60 to 80 percent of variability in daily maximum ozone concentrations depends on weather.

The reliability of measurements of resource flows, that is, air emissions, varies. Stationary sources, such as power plants, have been installing devices for continuous measurements of emissions of SO_x and NO_x since the 1990s to meet the RECLAIM requirements (about two-thirds of the RECLAIM units are required to do so). However, such devices were not in place before the 1990s, and measurability was limited in the early EPA trading systems. Mobile sources' emission rates are measured annually during emissions tests.

SO_x and NO_x pollution in this area has a clear nonuniform effect; a ton of criteria pollutant emitted in the coastal area affects air quality not only in this area but also in the inland area because of the direction of prevailing winds. The reverse, however, does not hold.

The main polluters are transportation, electric utilities, and industry. Transportation accounts for more than half the emissions of five important pollutants. Ozone's two major precursor gases, NO_x and volatile organic compounds (VOCs), come predominantly from transportation and industrial facilities. Two major emission sources of NO_x are transportation and stationary fuel-combustion sources (electric utilities and industrial boilers). Industrial and commercial sectors are the second-largest polluters. Major sources of ambient SO_2 are coal and oil combustion, steel mills, refineries, pulp and paper mills, and nonferrous smelters (EPA, 2010).

The severity of restrictions on resource use depends on three factors: (1) the level of resource stocks—the most stringent regulation of resource users is in the nonattainment areas, for example, SCAQMD; (2) the history of resource use by a given resource user (its presence in the resource area and its past compliance); and (3) planned future resource use (major or minor user).[12]

Rules for defining tradable permits differed between the early EPA trading and RECLAIM. The early EPA trading system defined permits through a case-by-case review of emission levels. This review was fraught with uncertainty regarding the amount of ERCs to be given. Because neither the baseline (until 1986) nor the estimation method was defined, neither the seller nor the buyer knew how many ERCs were actually involved in the trade (National Academy of Public Administration 1994).

In RECLAIM, each existing facility was allocated RECLAIM Trading Credits (RTCs) for equipment or processes that emitted NO_x and SO_x, starting in 1994. The

[12] Chapter 5 by Daniel Cole addresses the issues of access to resources in greater detail.

allocation was based on past peak emissions and was very generous. However, NO_x allowances were scheduled to decline annually by 7.1 percent until 2000 and by 8.1 percent from 2000 to 2003. SO_x emissions were scheduled to decline 4.1 percent annually until 2000 and by 8.1 percent until 2003 (Gangadharan 2000). New facilities have to purchase emission rights (as in the EPA policy in the 1970s). To obtain a permit, they have to demonstrate that best available control technology will be applied. Permit review is more stringent for "major" than for "minor" emitters.

The early EPA trading program and RECLAIM regulate different resource users. The former was focused on new sources that had to obtain ERCs to offset their emissions. RECLAIM, on the other hand, focuses on existing sources and also regulates new sources. As of 1992, about 30,000 firms had obtained permits in the Los Angeles Basin, but only facilities emitting more than four tons or more from permitted equipment per year are included in RECLAIM. There are approximately 390 facilities in the NO_x market, which account for about 65 percent of the emissions from all permitted stationary sources in the basin, and about 41 facilities in the SO_x market, which account for about 85 percent of reported emissions from all permitted stationary sources. Stationary sources, however, account for only 40 percent of the air pollution in the area, whereas mobile sources account for 60 percent (SCAQMD 1997). Electric utilities (until 2000), industrial boilers, manufacturers, and refineries are included in RECLAIM. Transportation projects are allowed to opt in. RECLAIM resource users have two different compliance cycles, the calendar year and the fiscal year, to avoid fluctuations in the market.

Nonregulated resource users can employ three programs to enter RECLAIM: ERCs resulting from scrapping of old vehicles (ERCs are converted to RTCs at a 1.2 discount factor); area source credits (ASCs) resulting from changes in applied technologies (ASCs are converted to RTCs at a ratio of 10 to 9); and Air Quality Investment Program credits, obtained from an investment fund operated by the SCAQMD.

State regulatory agencies served as brokers in the early EPA trading, bringing "the two partners to the negotiation table 'kicking and screaming'" (Liroff 1980, 18). RECLAIM, on the other hand, has several active broker companies. Data from the 1990s suggest that brokers played an important role in reducing transaction costs. By 1996 about 70 percent of the trades were conducted by brokers. RTC exchanges in which brokers were involved were traded at prices about 43 percent higher than those in which the polluters were trading directly (Gangadharan 2000). These price differences were eliminated by 2010.

The early EPA trading allowed four types of transfers of ERCs: bubbles, netting (used for existing sources), offsets (used by new sources), and banking. Bubbles (after 1979) provided flexibility to a facility to achieve emission reductions across its many processes rather than implementing a prescribed technology for each one. Netting (after 1974) allowed an existing facility to use ERCs in planning for a technology modification. The key advantage of using netting was that it allowed modifications and sources to qualify as "minor" rather than "major" and thereby avoid more stringent regulations. Offsets were devised in 1976 to be purchased by new major emitters from existing emitters. ERCs were discounted at each trade, and the extent of discounting was not known beforehand (National Academy of Public Administration 1994). Banking (beginning in 1979) allowed firms to hold unused ERCs for either future use or future sale. As of 1986, only five states had their banking regulations approved by the EPA.

The EPA's 1975 exchange rules used discounting to adjust for the nonuniform effect of pollution and restricted areas in which ERCs could be traded. Offsets for SO_x, for example, had to originate from sources in the immediate vicinity of the polluter. Similarly, the farther apart the seller and the buyer were, the more discounted the offsets were.[13] The discounting factors and sophisticated air-pollution modeling, required for such trades, impeded trading. Banking involved additional uncertainty; there was no assurance that the ERCs would not be discounted or even confiscated if the NAAQS were not met.

RECLAIM simplified the exchange rules with respect to nonuniformity of the pollutant. The SCAQMD area was divided into two homogeneous areas: coastal and inland. Trading was allowed within areas. Trading between areas, however, was restricted; facilities in the coastal areas could purchase only RTCs originating in the coastal area. This resulted in higher prices of RTCs from coastal areas. RECLAIM does not allow exchange of RTCs across time periods. Although facilities with different compliance periods can trade RTCs, RTCs can be used only in the year for which they are issued. This rule does not stimulate early achievement of environmental targets, but it does prevent occurrence of temporal hot spots.

The early EPA trading program had low trading volumes. Furthermore, trading activity varied across the four ERC transfer options because of different exchange rules and severities of resource use regulation. Netting was the most widely used (Hahn and Hester 1989a; 1989b; Liroff 1980). It brings significant savings because it allows that a source be classified as a minor source. Thereby, the firm avoids more stringent emission limits, modeling, and monitoring, which can all be costly (Hahn and Hester 1989a). Unfortunately, detailed data are available only for 1984. In this year only, nine hundred sources applied for netting in the entire United States.

Offsets were the second most used form of trading. Between 1977 and 1980, approximately 1,500 sources used offsets, and between 1981 and 1986, 500 sources used offsets. Traded offsets represented a small proportion of total emissions. For example, in 1985 there were five NO_x external offsets in the SCAQMD, with a total volume of 575 tons per year traded, accounting for less than 0.5 percent of total emissions. There were only two external SO_x offsets, with 310 tons per year traded at an average price of $3,000, accounting for less than 0.5 percent of total emissions.

Initial trading in RECLAIM was also modest, but then increased substantially over the years. In the first year of the policy, only about 9,000 tons of NO_x RTCs were traded. In subsequent years, the volume of NO_x RTCs increased to about 40,000 per year until 2000. The SCAQMD removed electric utilities from RECLAIM in 2000. Trading volumes after 2000 declined to about 15,000 tons annually. Traded volumes of SO_x RTCs were similarly high in the years before 2000. The volume of trade varied between a low of 10,000 tons in 1998 and a high of 22,000 in 1999. The crisis of 2000 reduced the traded volume to about 5,000 tons of SO_x. Since then, annual traded volume has varied from fewer than 1,000 tons to 13,000 tons.

[13] If the sources were relatively close, predetermined ratios were used. If the distance exceeded 30 miles, air-pollution modeling was required.

RTC prices also varied over the years. Average NO_x RTC prices were about $800 per ton in 1995, but more than $31,000 in 2001. After 2001 the prices decreased substantially, ranging from $3,147 in 2004 to $9,400 in 2006 (Coy and Luong, 2007). Prices for SO_x RTCs similarly started low at $600 per ton in 1995 and were at their highest in 2002, $7,850 per ton. After two years of high prices in 2002 and 2003, prices dropped to $2,000 in 2004 and $3,500 in 2005.

As previously noted, there are approximately 390 facilities in the NO_x market and about 41 facilities in the SO_x market. In 1995 only about 95 facilities (about 24 percent) actually traded in the NO_x market. The number of companies trading in the market increased over the years. In the third quarter of 2010 alone, almost 60 nonbroker companies traded in both markets.

If the environmental effectiveness of RECLAIM is judged by the actual emissions of the units included in the program, it was a success. Emission targets for SO_x were met in all years. Emission targets for NO_x were met in all years but 2000 and 2001, when electricity shortages required that old, polluting units be used again for electricity generation, resulting in increased pollution.

A review of resource stocks similarly suggests that pollution is decreasing in this area, although it still exceeds the NAAQS. For example, in the late 1970s, this area exceeded the permissible concentrations of ozone standards for almost two hundred days. In the late 1980s, the number of days when ozone concentrations exceeded standards were around one hundred and fifty. In 1993, the year in which RECLAIM was implemented, the number dropped below 120. In addition to the decline in the number of days in violation, there has also been a reduction in maximum concentrations of pollutants. In 1976 the highest eight-hour concentration of ozone was 0.268 parts per million (ppm); in 2008 the highest eight-hour concentration of ozone was 0.131ppm (the standard in 2008 was 0.075 ppm). As noted earlier, the major causes of ground ozone are NO_x (addressed by RECLAIM) and VOCs (SCAQMD, 2011).

Ozone-Depleting Substances

The ozone (O_3) layer is found in the earth's atmosphere between about six and thirty-one miles above the ground. In the early 1970s, scientists discovered human interference with stratospheric ozone. The substances causing the damage were CFCs emitted on the earth's surface but reaching the stratosphere because of their chemical stability. The reason for concern about ozone depletion and increased ultraviolet-B (UV-B) radiation is that the latter is associated with skin cancers and cataracts. The U.S. EPA estimated that continued depletion of the ozone layer could result in an additional 800,000 cancer deaths in the United States over the next century (McKinney and Schoch 1998). Further, increased UV-B radiation can adversely affect photosynthesis, metabolism, and growth of a number of plants. Phytoplankton, which form the basis of many food chains, are also susceptible to increased UV-B radiation. The international community limited the use of ozone-depleting substances (ODSs) in the Montreal Protocol to the Vienna Convention in 1987 that was then subsequently revised seven times. Countries accepted limitations on their production and consumption of ODSs. The analysis in this chapter focuses on the U.S. tradable-permit system only.

Although resource stocks were fairly well understood in the 1990s, the impact of resource flows, that is, emissions of ODSs, on the ozone hole was estimated only with high uncertainty (Hollandsworth and Binder 1998). Emissions of ODSs are measured by using production data. By multiplying production data for these substances by their ozone-depleting potential (ODP), one can estimate the total quantity of ODSs. The damage caused by emissions of ODSs varies across the globe. Maximum ozone depletion occurs in high latitudes, where trends are close to a 6 and an 8 percent reduction per decade in the Southern and Northern hemispheres, respectively. There is less depletion in the tropics, and there are even positive trends at the equator (Hollandsworth and Binder 1998). However, ODSs have uniform effects in that it does not matter where a ton of ODSs is emitted.

ODSs are used in a variety of products and processes, and resource users vary from households to the car industry. CFCs are widely used in plastic foams (32 percent), solvents (21 percent), car air conditioning (20 percent), other refrigeration (17 percent), medical sterilants (6.5 percent), and aerosols (3.5 percent) (Cook 1996).

Production and consumption of ODSs were regulated in the United States by imposing production and consumption limits, as well as by imposing taxes on these substances. The 1990 CAAA also banned nonessential products containing CFCs or hydrochlorofluorocarbons (HCFCs), such as flexible and packaging foams, most aerosols, and pressurized dispensers, and required labeling of products that contained or were manufactured with ODSs. To implement the CAAA, the EPA issued deadlines for phasing out the production and consumption of Class I substances (CFCs, carbon tetrachloride, methyl chloroform, and halons) in 1992. End users were also regulated. Standards were set for servicing equipment using CFCs, and car air-conditioning substitutes were established (62 Fed. Reg. 68,026; EPA, 2010).

Permits to produce ODSs were allocated to five CFC producers, three halon producers, fourteen CFC importers, and six halon importers (Lee 1996). This is the smallest group of regulated users of all markets examined in this chapter. Because this group is also very homogeneous and has a history of intragroup trading, identifying potential partners was not too difficult or costly.

Users of CFCs and halons (or products containing these substances) were allocated consumption permits. The EPA estimated that in the 1980s there were more than 10,000 CFC and halon user sectors (Lee 1996). Their emissions were regulated by standards. For example, the 1990 CAAA enacted standards for CFC-recycling equipment for air conditioners and refrigerators. Users, however, also adjusted their use patterns to the new market situation. This shift in demand (for example, the solvent industry stopped using CFC-113, recycled used substances, and started substituting) made CFC-113 allowances available for trading. They were traded and used as allowances for substances that were still in demand, such as CFC-12.

As the United States banned some uses of CFCs and scheduled other ODSs for a phaseout by 1996, the severity of resource use limits in the case of CFCs is comparable to that of the lead phasedown program. However, some of these substances and products containing them were taxed, and their prices increased; in some cases (CFC-11 and CFC-12), they doubled. By 1995 the taxed price was nearly triple the untaxed price.

The right to use the atmosphere as a sink for ODSs was expressed as the right to produce a given amount of these substances. However, because manufacturers produced substances with various ODPs, the issue of assigning rights arose. Should the permits be allocated for each substance, or should they be allocated as an ODP-weighted aggregate? Under the first system, each user is allocated permits for each ODS. Under the second system, the user is issued ODP-weighted allowances.

The CFC and halon trading systems were initially based on weighted averages. The 1990 CAAA enacted substance permits. If a user wanted to trade allowances of one substance for allowances of another substance, the EPA based the calculation of the trading ratio on the ODP. In this way, the EPA could track who had allowances of which substance. At the time of reducing allowances or completely phasing out a substance, information was then available on whose allowances were being reduced. If allowances are based on ODP-weighted averages, information about which substances are the base of the allocated permits is lost. The EPA does not know how much to issue to which user the next time the permits are issued.

Exchangeability of permits is based on the ODPs of the substances and the origin of the substance (manufactured or recycled). Difficulties in distinguishing between manufactured and recycled substances in the absence of a carefully documented recycling process result in illegal trades of these substances. Each manufacturer must have a sufficient amount of permits to cover the production of the substances. Each consumer must have a sufficient amount of consumption permits to cover the purchased substances. Each transfer of substances must be accompanied by a transfer of permits. If substances are imported from countries that do not have limits on CFC production (Article 5 countries), no transfer of permits is possible. Transfers of substances from these countries are therefore limited to recycled substances. Close examination of the flow of recycled substances, their origin, and the recycling capabilities of the industry in those countries indicates that large amounts of CFCs and halons are illegally exported to the United States and the European Union, predominantly from China, India, and Russia. The economic motivation is clear. The excise tax and the domestic production reduction and phase-out increased the price of these substances in non–Article 5 countries, such as the United States. The price of domestically reclaimed halon 1301 is about $26 per kilogram, whereas the price of supposedly recycled halon from China is $7.5 per kilogram (EIA, 1998).

The market for ODSs in the United States was a small club with a total of 28 producers and importers who were initially allocated permits. All these facilities were listed in a *Federal Register* notice. The Alliance for Responsible Atmospheric Policy, an industrial organization, brought together potential traders. Trades had to be preapproved by the EPA, which promised to process requests within three days. Two EPA offices were involved in the program: the Office of Atmospheric Programs tracked allowances, and the Office of Enforcement monitored compliance. Because the definitions of traded goods and quantities did not depend on a case-by-case review, but on issued allowances, approval required only that the two offices cross-check their data. All these aspects suggest that exchange costs were low.

Targets for reductions in production and consumption of CFCs and halons were not only met, but exceeded. The 1989 target for CFC production was about 342,000 ODP tons, but actual production was about 320,000 tons. The target for 1994 was about 109,000 tons, but actual production was about 78,000 tons. By 1996 the United States produced only 676 tons of CFCs. The CFC consumption-reduction targets were also exceeded. The target for 1989 was about 337,000 tons, but actual consumption was about 318,000 tons. The 1994 target was about 107,000 tons, with actual consumption of about 73,000 tons, and the 1996 target was about 46,000 tons, with actual consumption of about 2,000 tons. Similarly, the 1992 target for halon production was about 65,000 tons, but actual production was about 26,000 tons, and the 1992 target for halon consumption was about 64,000 tons, but actual consumption was about 24,000 tons. The 1994 production and consumption targets were each about 6,000 tons, but the United States had completely phased out halon production and consumption by then (UNEP, 1999).

However, the question is to what extent these reductions can be attributed to tradable permits and to what extent they were caused by the excise taxes, introduced in 1990. Allowances were already assigned in 1989, but production had not fallen drastically, whereas in 1990, the first year when the tax was paid, production decreased substantially.

Lee (1996) claims that permit trading helped American companies exceed the Montreal Protocol targets. He argues that the drop in anticipated costs of reducing CFC production from $3.55 per kilogram of CFC to $2.45 was a result of the marketable system, which lowered administrative costs and offered the flexibility needed to meet the reduction goals. There is no question that allowance trading offers flexibility.

However, the trading data for 1989 and 1990 suggest modest trading levels. Each year, allowances for not more than 1 million kilograms of ODSs were traded between companies. In 1991 the trading increased to 80 million kilograms of ODSs (45 trades). The amount of traded ODSs then remained fairly stable until 1995, when it dropped to 60 million kilograms. Therefore, the major overcompliance that Lee (1996) identifies in 1990 cannot plausibly be attributed to trading. Further, in 1990, when U.S. actual production was 6 percent below the allowed limit, the productions of France, Germany, and the United Kingdom were 29, 23, and 33 percent, respectively, below their allowed production.

The market saw little activity in trades until 1992 (5, 15, and 45 trades in 1989, 1990, and 1991, respectively). In 1992 the number of trades increased to 217 and then declined to 95 in 1995. Although production and consumption were reduced beyond the targets, it is not likely that these results were accomplished solely because of the tradable-permit system. Without additional data, it is impossible to conclude how much of the effectiveness was due to the excise tax and how much to the tradable-permit system.

Findings

This chapter has examined the effectiveness of four tradable-permit systems for emissions of pollutants: emissions of lead (lead phasedown program); emissions of nitrogen oxides (early EPA emissions trading and RECLAIM); and emissions of ODSs (CFC and halon production and consumption quota trading). Table 6.1 presents an overview of the characteristics of these tradable-permit systems.

The examined air-pollution cases vary from small and local (for example, using the atmosphere as a sink for pollutants that have local effects, such as lead) to those with a global extent (emission of ODSs). They include cases with a very small number of users (CFCs) and with large numbers (RECLAIM). The security and the exchangeability of permits vary from very limited in some markets (early EPA emissions trading) to high for others (lead phasedown program in the second stage).

Two markets exhibited high environmental performance and high trading activity: the lead phasedown program and ODS production-quota trading. High environmental performance is associated with severe limits on resource use in all markets. Although theory would predict that nonuniform impacts of emissions would have reduced trading, the institutional design appropriately addressed it. Instead of assigning permits to the users who cause pollution (cars), the permits were assigned to those who produced polluting fuel (refineries). So-called hot spots could have occurred if gasoline from the lead-permit buyers (high lead-content gasoline) were concentrated in one market. However, because gasoline from overcomplying refineries was mixed in the pipeline system with gasoline from refineries that used higher amounts of lead, the hot spots did not occur. Nonetheless, the accomplished reductions in resource use cannot be attributed solely to the tradable permits; other policy instruments (excise taxes) and external factors (technological changes that provided substitutes for polluting substances) were also involved. Therefore, we must be cautious about attributing environmental effectiveness to tradable permits.

In sum, comparative case analysis suggests that two factors commonly perceived to affect market effectiveness negatively—large spatial extent of the environmental resource and nonuniform effect of resource flows on resource stocks—do not necessarily reduce the effectiveness of tradable permits. Transaction costs, identified as significantly hindering trading activities in earlier markets, have successfully been reduced over time. These findings suggest that tradable permits can be applied to a larger number of environmental problems than the critics of this approach would admit.

In contrast, the other two tradable-permit markets exhibited low environmental performance. This result can be linked to low severity of limits on resource use and to lack of information available for monitoring and enforcement (RECLAIM). Therefore, future research will need to focus on political factors that affect the design of tradable permits, especially the severity of resource use limits and rules regulating opting in.

Several lessons emerge from this study. First, the environmental effectiveness of tradable-permit regimes is influenced by the severity of restrictions imposed on resource users. Of course, severity leads to the erosion of political support among existing polluters. Thus, tradable-permit regimes need to find a balance between political imperatives and institutional requirements. Second, policy makers should

TABLE 6.1

Tradable-Permits Systems for Air Pollution

	Independent Variables					Outcome	
Permit System	Resource	Users	Regulated Users	Permits	Exchange Costs	Environmental Effectiveness	Trading Activity
Lead phasedown program	High predictability; local once emitted; nonuniform	Moderate number	Homogeneous group; trading experience; little opting in; samples monitored	Clearly defined; no property right; severe limits; exchangeable in space, but not in time	Low	High	High
RECLAIM	Moderate predictability; local extent; nonuniform	Large number	Not very homogeneous; opting in; partially continuous monitoring	Clearly defined; exchange ratios (space); exchangeable in time; modest severity	Moderate	Low	Moderate
ODS production-quota trading	Moderate predictability; global extent; uniform effect	Small number	Very small number; homogeneous group; no opting in; phased self-reporting	Clearly defined; substance-specific; exchange ratios; high value; severe limits on resource use	Low	High	High
Early EPA emissions trading	Moderate predictability; local extent; nonuniform	Large number	Large number; not very homogeneous; opting in	Case-by-case definition; exchange ratios; high severity	High	Low	Low

introduce tradable permits at appropriate points of the resource use cycle and devise and enforce credible rules for reducing resource use over time. Because the findings presented in this chapter are based on limited data, the causal inferences have high uncertainty. To reduce this uncertainty, future efforts should target compilation of a set of measures that are comparable across tradable-permit markets. In particular, researchers need to identify better data to measure cost savings of tradable-permit systems and to develop comparable measures of environmental performance. They also need to control for the share of the resource that is not regulated by the tradable-permit system, for potential leakage of resource use to non-participants in the market, and for the policy effect of related policy instruments implemented concurrently with tradable permits.

Resource users participate in a variety of markets, including the market for the tradable permits. Two kinds of markets are especially important: the market for products requiring the use of the regulated resource (increased electricity demand will likely increase demand for tradable permits for air pollution; increased timber and cash-crop demand will likely lead to increased deforestation); and markets that enable resource users to decouple their income from resource use (for example, markets for technologies that reduce the use of the atmosphere as a sink for pollution). These markets can either increase or alleviate pressure on environmental resources and thereby influence the efficacy of tradable-permit regimes. In the future, it will be important to examine the impact of these markets on the effectiveness of tradable-permit markets. Globalization and access to markets with high demand for resource conservation may actually help reduce the exploitation of natural resources in countries with lower ability to postpone current resource use with the goal of protection. It is obvious, then, that it is not possible to discuss only one way in which private rights allocation and privatization affect natural resources and air pollution. Rather, it is necessary to study factors affecting market outcomes and perhaps suggest how privatization can be used to protect environmental resources.

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