ECONOMIC INCENTIVES FOR ENVIRONMENTAL REGULATION

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Robert N. Stavins

EXECUTIVE SUMMARY

Environmental policies consist of two components: the identification of an overall goal and some means to achieve that goal. This essay considers the second component, the means — the "instruments" — of environmental policy, and focuses, in particular, on the use of economic-incentive or market-based policy instruments.

The paper begins by considering a cost-minimizing pollution control program for a uniformly-mixed, flow pollutant, for which — as is well known — a cost-effective allocation of the pollution-control burden among sources requires that all sources experience the same marginal abatement cost. To achieve this cost-effective allocation of the pollution-control burden, the government could conceivably establish a non-uniform (source-specific) standard to ensure that all firms would control emissions at the same marginal cost of control, but this would require detailed information about the costs faced by each source, information that could be obtained by the authority only at very great cost, if at all. One way out of this impasse is through the use of economic-incentive or market-based policy instruments.

Economic-incentive instruments are regulations that encourage behavior through price signals rather than through explicit instructions on pollution control levels or methods. These policy instruments, such as tradable permits and pollution charges, have been described as "harnessing market forces," because if they are properly implemented, they encourage firms to undertake pollution control efforts that are in their financial self-interest and that will collectively meet policy goals.

Economic-incentive instruments have captured the attention of environmental policy makers in recent years because of the potential advantages they offer over traditional command-and-control approaches. In theory, properly designed and implemented economic-incentive instruments allow any desired level of pollution cleanup to be realized at the lowest possible overall cost to society, because they provide incentives for the greatest reductions in pollution by those firms that can achieve these reductions most cheaply. Rather than equalizing pollution levels among firms, economic-incentive instruments equalize the incremental amount that firms spend to reduce pollution (their marginal abatement costs).

Economic-incentive instruments can be divided into five categories. *Pollution charge* systems assess a fee or tax on the amount of pollution a firm generates. *Tradable permits* can achieve the same cost-minimizing allocation of the pollution control burden as a charge system, while avoiding the problem of uncertain responses by firms. A special case of a pollution tax is a

deposit refund system, under which consumers pay a surcharge when purchasing potentially polluting products, and receive a rebate when returning the product to an approved center for recycling or proper disposal. *Reducing market barriers* can also help to curb pollution, since, in some cases, substantial gains can be made in environmental protection simply by removing existing government-mandated barriers to market activity. Finally, *elimination of government subsidies* can be a powerful economic incentive for environmental protection, because some subsidies promote inefficient and environmentally unsound economic development.

There have been six major applications of economic-incentive instruments in the United States: the U.S. Environmental Protection Agency's (EPA) Emissions Trading Program, the leaded gasoline phasedown, water quality permit trading, chloroflourocarbon (CFC) trading, the SO_2 allowance system for acid rain control, and the RECLAIM program in the Los Angeles metropolitan region.

Economic-incentive instruments have delivered attractive results where implemented and promise additional future benefits. To date, their effectiveness has been undermined by unrealistic expectations, lack of political will, flaws in design, and constraints imposed by the internal structure of firms. These are all remediable. Policy makers should direct their efforts to making future applications work better than those that came before.

ECONOMIC INCENTIVES FOR ENVIRONMENTAL REGULATION

Robert N. Stavins

This paper provides a brief summary of the theory and the reality of economic-incentive approaches to environmental regulation. The paper begins with a derivation of the necessary and sufficient condition for a policy instrument to be cost effective, namely that the instrument induces all sources to abate emissions at the same marginal cost. Subsequent sections introduce economic-incentive policy instruments, including emission charges and tradeable permits, and show that these instruments — in theory — meet the specified condition. Then, six U.S. applications are described: EPA's emissions trading program; the leaded gasoline phasedown; water quality permit trading; the CFC phaseout; the SO_2 allowance system; and the RECLAIM program. Reasons for the limited use of incentive-based instruments are examined, including: the role of interest groups; ambivalence by regulated firms; and consumers' perspectives. Reasons for the mixed record of implemented instruments are also examined: inaccurate predictions; design problems; and limitations in firms' internal structures.

1. INTRODUCTION: CRITERIA FOR POLICY INSTRUMENT CHOICE

Nearly all environmental policies consist of two components, either explicitly or implicitly: the identification of an overall goal (such as a degree of air quality) and some means to achieve that goal. This essay considers the second component, the means — the "instruments" — of environmental policy, and focuses, in particular, on the use of economic-incentive or market-based policy instruments.

A variety of criteria — both economic and others — can be brought to bear on the choice of policy instruments to achieve some environmental goal or standard. Among economic criteria, three stand out: static cost effectiveness; dynamic cost effectiveness; and distributional equity. Static cost effectiveness refers to the minimization of the short-term aggregate costs of achieving a particular level of pollution control. Similarly, dynamic cost effectiveness refers to the minimization of the present discounted value of the future stream of aggregate costs. Distributional equity refers to the "fairness" of the distribution of the benefits and costs of the environmental policy, both cross-sectionally (among geographic regions, income groups, etc.) and intertemporally.

We begin by considering a cost-minimizing pollution control program for a uniformly-mixed, flow pollutant. For such an environmental problem, we can focus on aggregate emissions per unit of time, where aggregate emissions, E, are simply the sum of emissions, e_i, from N individual firms or sources, where emissions from each source are the difference between unconstrained emissions, u_i, and emission reductions, r_i. A cost-effective emission-control program is one that controls aggregate emissions from all sources at minimum total cost:

$$\min_{\{r_i\}} C = \sum_{i=1}^{N} c_i(r_i) \tag{1}$$

$$\textit{subject to:} \quad \sum_{i=1}^{N} \left[u_i - r_i\right] \leq \bar{E} \tag{2}$$

$$0 \leq r_i \leq u_i \tag{3}$$

If the control cost functions are convex in their relevant ranges, then the necessary and sufficient conditions for cost minimization are that the marginal cost of control be the same among all sources that carry out positive levels of control:

$$\frac{\partial c_i(r_i)}{\partial r_i} - \lambda \geq 0 \tag{4}$$

$$r_i \left[\frac{\partial c_i(r_i)}{\partial r_i} - \lambda \right] = 0 \tag{5}$$

Thus, in a cost-effective allocation of the pollution-control burden among sources, any and all sources that exercise a non-zero level of control must experience the same marginal abatement cost.

To achieve this cost-effective allocation of the pollution-control burden, the government could conceivably establish a non-uniform (source-specific) standard to ensure that all firms would control emissions at the same marginal cost of control, but this would require detailed information about the costs faced by each source, information that could be obtained by the authority only at very great cost, if at all. One way out of this impasse is through the use of economic-incentive or market-based policy instruments.

2. WHAT ARE ECONOMIC-INCENTIVE POLICY INSTRUMENTS?

Economic-incentive instruments are regulations that encourage behavior through price signals rather than through explicit instructions on pollution control levels or methods (Hahn and Stavins 1991). These policy instruments, such as tradable permits and pollution charges, have been described as "harnessing market forces" because if they are properly implemented, they encourage

firms to undertake pollution control efforts that are in their financial self-interest and that will collectively meet policy goals (Stavins 1988, 1991).

Conventional approaches to regulating the environment are frequently referred to as "command-and-control" regulations since they allow little flexibility in the means of achieving goals. Early environmental policies in the United States and other industrialized nations relied almost exclusively on these approaches (Portney 1990).

In general, command-and-control regulations force firms to shoulder identical shares of the pollution-control burden, regardless of the relative costs. Command-and-control regulations do this by setting uniform standards for firms, the most prevalent of which are technology-based and performance-based standards. Technology-based standards specify the method, and sometimes the actual equipment, that firms must use to comply with a particular regulation. For example, all electric utilities might be required to employ a specific type of "scrubber" to remove particulates. A performance standard sets a uniform control target for firms, while allowing some latitude in how this target is met. For example, a regulation might limit the number of allowable units of a pollutant released in a given time period, but might not dictate the means by which this is achieved.

Holding all firms to the same target can be expensive and, in some circumstances, counterproductive. While standards can effectively limit emissions of pollutants, they typically exact relatively high societal costs in the process, by forcing firms to resort to unduly expensive means of controlling pollution. A survey of eight empirical studies of air pollution control found that the ratio of actual, aggregate costs of the conventional, command-and-control approach to the aggregate costs of least-cost benchmarks ranged from 1.07 for sulfate emissions in Los Angeles to 22.0 for hydrocarbon emissions at all domestic DuPont plants (Tietenberg 1985). Because the costs of controlling emissions may vary greatly between firms, and even within the same firm, the appropriate technology in one situation may be inappropriate in another.

Furthermore, command-and-control regulations tend to freeze the development of technologies that might otherwise result in greater levels of control. Little or no financial incentive exists for businesses to exceed their control targets, and both technology-based and performance-based standards discourage experimentation with new technologies. A firm adopting a new technology may be "rewarded" by being held to a higher standard of performance, and not given the opportunity to benefit financially from its investment, except to the extent its competitors have even more difficulty reaching the new standard.

2.1 Characteristics of Economic-Incentive Policy Instruments

Economic-incentive instruments have captured the attention of environmental policy makers in recent years because of the potential advantages they offer over traditional command-and-control approaches. In theory, properly designed and implemented economic-incentive instruments allow any desired level of pollution cleanup to be realized at the lowest possible overall cost to society, because they provide incentives for the greatest reductions in pollution by those firms that can achieve these reductions most cheaply. Rather than equalizing pollution levels among firms,

economic-incentive instruments equalize the incremental amount that firms spend to reduce pollution (their marginal abatement costs).

As suggested above, command-and-control approaches could theoretically achieve this cost-effective solution. However, this would require that different standards be set for each pollution source, and, consequently, that policy makers obtain detailed information about the compliance costs each firm faces. Such information is simply not available to government. By contrast, economic-incentive instruments provide for a cost-effective allocation of the pollution control burden among sources without this information. Additionally, in contrast to command-and-control regulations, economic-incentive instruments have the potential to provide powerful incentives for firms to adopt cheaper and better pollution-control technologies (Magat 1978; McHugh 1985; Downing and White 1986; Milliman and Prince 1989; Malueg 1989; Jaffe and Stavins 1995).

2.2 Types of Economic-Incentive Instruments

Economic-incentive instruments can be divided into five categories: pollution charges, tradable permits, deposit-refund systems, reductions in market barriers, and government subsidy elimination (U.S. Environmental Protection Agency 1991; Organization for Economic Cooperation and Development 1994; U.S. Congress 1995).

Pollution charge systems assess a fee or tax on the amount of pollution a firm generates. For example, a pollution charge might take the form of a charge per unit of sulfur dioxide (SO_2) emissions. The choice of whether to tax pollution quantities, activities preceding discharge, inputs to those activities, or actual damages will depend upon tradeoffs between costs of abatement, mitigation, damages, and program administration, including monitoring and enforcement. With a charge system in place, it becomes worthwhile for each firm to reduce pollutant emissions to the point at which its marginal cost of control is equal to the pollution-tax rate. An individual cost-minimizing firm, *i*, faces the following problem:

$$\min_{\{r_i\}} \left[c_i(r_i) + t \cdot (u_i - r_i) \right] \tag{6}$$

$$\textit{subject to:} \qquad r_i \geq 0 \tag{7}$$

The necessary conditions that follow for each such source include:

$$\frac{\partial c_i(r_i)}{\partial r_i} - t \geq 0 \tag{8}$$

$$r_i \left[\frac{\partial c_i(r_i)}{\partial r_i} - t \right] = 0 \tag{9}$$

Thus, since control levels, r_i, must be non-negative, each and every firm that carries out a non-zero degree of abatement, finds it in its interest to carry out control up to the point where its marginal abatement cost is equal to the tax rate, t. Since all sources equate their marginal abatement costs with the same tax rate, marginal abatement costs will be identical across all sources. Hence, the imposition of a pollution tax leads to the cost-effectiveness allocation of the pollution-control burden among sources. By internalizing the previously external pollution costs, firms control pollution to differing degrees, with high-cost controllers controlling less, and low-cost controllers controlling more.

A.C. Pigou (1920) is generally credited with developing the idea of a corrective tax to discourage activities which generate externalities, such as environmental pollution. But, the difficulty with charges is figuring out where to set the tax. If environmental damages are quantified in economic terms, then the charge should be set equal to social damages at the efficient level of control, that is, at the level of control where marginal benefits (avoided marginal damages) are equivalent to marginal abatement costs. It is rare, however, that reliable economic valuations of damages are available. Instead, policy makers may seek to identify the tax rate that will lead to the politically-determined level of aggregate abatement. But it is difficult for government to know beforehand how firms will respond to a given level of taxation. Despite the availability in a few cases of relevant elasticity estimates, in general it is difficult for government to ascertain with precision what level of cleanup will result from any given charge.

Tradable permits can achieve the same cost-minimizing allocation of the pollution control burden as a charge system, while avoiding the problem of uncertain responses by firms (Dales 1968; Montgomery 1972; Hahn and Noll 1982). Under a tradable permit system, an allowable overall level of pollution is established and then allotted among firms in the form of permits. Firms that keep their emissions below the allotted level may sell or lease their surplus permits to other firms or use them to offset excess emissions in other parts of their facilities.

Recall that emissions by an individual source, e_i, are equal to the difference between unconstrained emissions, u_i, and abatement (reductions), r_i. Let q_{0i} be the initial allocation of emission permits to a individual source, where the sum of such permits is equivalent to the aggregate emission goal:

$$\sum_{i=1}^{N} q_{0i} = \bar{E} \tag{10}$$

Then, an individual cost-minimizing firm, i, faces the following problem, if the price of permits, p, is taken as exogenous in a competitive market:

$$\min_{\{r_i\}} \left[c_i(r_i) + p \cdot (u_i - r_i - q_{0i}) \right] \tag{11}$$

$$\textit{subject to:} \qquad r_i \geq 0 \tag{12}$$

The necessary conditions that follow for each such source include:

$$\frac{\partial c_i(r_i)}{\partial r_i} - p \geq 0 \tag{13}$$

$$r_i \left[\frac{\partial c_i(r_i)}{\partial r_i} - p \right] = 0 \tag{14}$$

Thus, since the control levels, r_i, must be non-negative, each firm that carries out a non-zero degree of abatement, will tend to control up to the point at which its marginal abatement costs are equal to the exogenous permit price, p. Since all sources are equating their marginal abatement costs with the same permit price, marginal abatement costs are identical across sources. Hence, the environmental constraint is satisfied and the cost-effective allocation of the pollution-control burden is achieved.

A special case of a pollution tax is a *deposit refund system*, under which consumers pay a surcharge when purchasing potentially polluting products. Upon return of the product to an approved center for recycling or proper disposal, the deposit is refunded. A number of American states, Canadian provinces, and European nations have successfully implemented this system through "bottle bills," to control litter from beverage containers and to reduce the flow of solid waste to landfills (Bohm 1981; Menell 1990). This concept has also been applied to lead-acid batteries.

Reducing market barriers can also help to curb pollution. In some cases, substantial gains can be made in environmental protection simply by removing existing government-mandated barriers to market activity. For example, measures that facilitate the voluntary exchange of water rights promote more efficient allocation and use of scarce water supplies (Willey and Graff 1988).

Elimination of government subsidies can be a powerful economic incentive for environmental protection. Subsidies are the mirror image of various taxes and, in theory, can provide economic incentives to address environmental problems. In practice, however, many subsidies promote

inefficient and environmentally unsound economic development. A prime example is the below-cost sale of timber by the U.S. Forest Service, which does not allow for the recovery of the cost of making timber available for harvesting by private firms.

In the simplest models, as indicated above, pollution taxes and tradeable permits are symmetric, but that symmetry begins to break down in actual implementations. First, permits fix the level of pollution control while charges fix the costs of pollution control. Second, in the presence of technological change and without additional government intervention, permits freeze the level of pollution control while charges increase it. Third, with permit systems as adopted, resource transfers are private-to-private, while they are private-to-public with ordinary pollution charges. Fourth, while both charges and permits increase costs on industry and consumers, charge systems make the costs more explicit to both groups. Fifth, permits adjust automatically for inflation, while some types of charges do not. Sixth, permit systems may be more susceptible to strategic behavior. Seventh, significant transaction costs can drive up the total costs of compliance, having a negative effect under either system, but particularly with tradeable permits. Eighth and finally, in the presence of uncertainty, either permits or charges can be more efficient, depending upon the relative slopes of the marginal benefit and marginal cost functions (Weitzman 1974) and any correlation between them (Stavins 1996).

The degree of abatement achieved by a pollution tax and the tax's effect on the economy will depend — in part — on what is done with the tax revenue. There is widespread agreement that revenue recycling (that is, using revenues to lower other taxes) can significantly lower the costs of a pollution tax (Jorgenson and Wilcoxen 1994; Goulder 1995). Some researchers have suggested, further, that all of the abatement costs associated with a pollution tax can be eliminated through revenue recycling in the form of cuts in taxes on labor (Repetto, Dower, Jenkins, and Geoghegan 1992). There is now common recognition, however, that this stronger claim is not valid (Bovenberg and de Mooij 1994; Bovenberg and Goulder 1996). Indeed, some pollution taxes can exacerbate distortions associated with remaining taxes on investment or labor.

The revenue raised by an auction of tradeable permits can also be used to finance a reduction in some distortionary tax (Goulder, Parry, and Burtraw 1996). As Fullerton and Metcalf (1996) note, some environmental policy instruments — including those that raise and then refund revenues — do not create scarcity rents for the private sector that then become effective entry barriers.

3. APPLICATIONS OF ECONOMIC-INCENTIVE INSTRUMENTS

There have been six major applications of economic-incentive instruments in the United States: the U.S. Environmental Protection Agency's (EPA) Emissions Trading Program, the leaded gasoline phasedown, water quality permit trading, chloroflourocarbon (CFC) trading, the SO_2 allowance system for acid rain control, and the RECLAIM program in the Los Angeles metropolitan region.

3.1 The Emissions Trading Program

Beginning in 1974, EPA experimented with emissions trading as part of the Clean Air Act's program for improving local air quality. Firms that reduced emissions below the level required by law received credits usable against higher emissions elsewhere. Firms could employ the concepts of "netting" or "bubbles" to trade emissions reductions among sources within the firm, so long as total, combined emissions did not exceed an aggregate limit (Tietenberg 1985; Hahn 1989). The "offset" program, which began in 1976, goes further in allowing firms to trade emission credits. Firms wishing to establish new sources in areas that are not in compliance with ambient standards can offset their new emissions by reducing existing emissions. This can be accomplished through internal sources or through agreements with other firms. Finally, under the "banking" program, firms may retain earned emission credits for future use. Banking allows for either future internal expansion or the sale of credits to other firms.

EPA codified these programs in its Emissions Trading Program in 1986 (U.S. Environmental Protection Agency, *Emissions Trading Policy Statement,* 51 Fed. Reg. 43,814, 1986, final policy statement), but the programs have not been widely used. States are not required to use the program, and uncertainties about its future course seem to have made firms reluctant to participate (Liroff 1986). Nevertheless, numerous firms have traded emissions credits, and a market for transfers has long since developed (Main 1988). Even this limited degree of participation in EPA's trading programs may have saved between $5 billion and $12 billion since the program's inception (Hahn and Hester 1989b).

3.2 Lead Trading

The purpose of the lead trading program, developed in the 1980s, was to allow gasoline refiners greater flexibility in meeting emission standards at a time when the lead-content of gasoline was reduced to 10 percent of its previous level. In 1982, the EPA authorized inter-refinery trading of lead credits (U.S. Environmental Protection Agency. *Regulation of Fuel and Fuel Additives.* 49,322-24, final rule). If refiners produced gasoline with a lower lead content than was required, they earned lead credits. In 1985, EPA initiated a program allowing refineries to bank lead credits, and subsequently firms made extensive use of this program. EPA terminated the program at the end of 1987, when the lead phasedown itself was complete.

The lead program was successful in meeting its environmental target, and the high level of trading activity achieved suggests that the program was relatively cost-effective. In each year of the program, more than 60 percent of the lead added to gasoline was associated with traded lead credits, and over half of all refineries participated in trading with other firms (Hahn and Hester 1989a). EPA estimated savings from the lead trading program of approximately twenty percent over alternative programs that did not provide for lead banking, a cost savings of about $250 million per year (U.S. Environmental Protection Agency 1985). The program experienced some relatively minor implementation difficulties related to the importation of leaded fuel, but it is not clear that a comparable command-and-control approach would have done better in terms of environmental quality (U.S. General Accounting Office 1986).

3.3 Water Quality Permit Trading

Non-point sources of water pollution, particularly agricultural and urban runoff, constitute the major, remaining American water quality problem (Peskin 1986). An experimental program to protect the Dillon Reservoir in Colorado demonstrates how tradable permits could be used, in theory, to reduce nonpoint-source water pollution. The reservoir is the major source of water for the city of Denver. Nitrogen and phosphorus loading threatened to turn the reservoir eutrophic, despite the fact that point sources from surrounding communities were controlled to best-available technology standards (U.S. Environmental Protection Agency 1984). Rapid population growth in Denver, and the resulting increase in urban surface water runoff, further aggravated the problem.

In response, state policy makers developed a point-nonpoint-source control program to reduce phosphorus flows, mainly from nonpoint urban and agricultural sources (Hahn 1989). The program was implemented in 1984. It allowed publicly owned sewage treatment works to finance the control of nonpoint sources in lieu of upgrading their own treated effluents to drinking water standards. EPA estimated that the plan could save over $1 million per year (Hahn and Hester 1989a), due to large differences in the marginal costs of control between nonpoint sources and sewage treatment facilities. However, no trading ever occurred under the program, apparently because high regional precipitation essentially eliminated its need.

3.4 CFC Trading

A market in tradable permits was used in the United States to help comply with the Montreal Protocol, an international agreement aimed at slowing the rate of stratospheric ozone depletion. The Montreal Protocol called for a 50 percent reduction in the production of CFCs from 1986 levels by 1998. In addition, the Protocol froze halon production and consumption at 1986 levels beginning in 1992. The market places limitations on both the production and consumption of CFCs by issuing allowances that limit these activities. The Montreal Protocol recognizes the fact that different types of CFCs are likely to have different effects on ozone depletion. Therefore, each CFC is assigned a different weight on the basis of its depletion potential. If a firm wishes to produce a given amount of CFC, it must have an allowance to do so, calculated on this basis (Hahn and McGartland 1989).

Through mid-1991 there were 34 participants in the market and 80 trades. However, the overall efficiency of the market is difficult to determine, because no studies were conducted to estimate cost savings. The timetable for the phaseout of CFCs was subsequently accelerated, and a tax on CFCs was introduced. Indeed, the tax may have become the binding (effective) instrument. Nevertheless, relatively low transaction costs associated with trading in the CFC market suggest that the system was relatively cost-effective.

3.5 SO_2 Allowance System for Acid Rain Control

A centerpiece of the Clean Air Act Amendments of 1990 is a tradable permit system that regulates sulfur dioxide emissions, the primary precursor of acid rain (Clean Air Act Amendments of 1990, Public Law No. 101-549, 104 Statute 2399, 1990). Title IV of the Act reduces sulfur dioxide and nitrous oxide emissions by 10 million tons and 2 million tons, respectively, from 1980

levels (Ferrall 1991). The first phase of sulfur dioxide emissions reductions was achieved by 1995, with a second phase of reduction to be accomplished by the year 2000.

In Phase I, emissions limits were assigned to 111 electrical plants. After January 1, 1995, these utilities could emit excess SO_2 only if they qualified for extensions or substitutions, or if they obtained allowances. During Phase I, the EPA allocated each affected utility, on an annual basis, a specified number of allowances related to its capacity, plus bonus allowances available under a variety of special provisions (Joskow and Schmalensee 1995). Cost-effectiveness was promoted by permitting allowance holders to transfer their permits among one another.

Under Phase II of the program, beginning January 1, 2000, almost all electric power generating units will be brought within the system. Certain units are excepted to compensate for potential restrictions on growth and to reward units that are already unusually clean. If trading permits represent the carrot of the system, its stick is a penalty of $2,999 per ton of emissions that exceed any year's allowances (and a requirement that such excesses be offset the following year).

A robust market of bilateral SO_2 permit trading has emerged, resulting in cost savings in the area of $1 billion annually, compared with the anticipated costs under a command-and-control regime. Nevertheless, the program has fallen short of predictions in terms of the number of permits traded and the price of permits (Burtraw 1995). This may have more to do with faulty predictions than problematic performance, however. Despite earlier concerns that state regulatory authorities would hamper trading in order to protect their domestic coal industries, preliminary evidence suggests that this has not been a major problem (Bailey 1996). Similarly, in contrast to early assertions that the structure of EPA's permit auction market would cause problems (Cason 1995), the evidence now indicates that this has had little or no effect on the vastly more important bilateral trading market (Joskow, Schmalensee, and Bailey 1996).

3.6 The RECLAIM Program

The South Coast Air Quality Management District (SCAQMD), which is responsible for controlling emissions in a four-county area of Southern California, launched a tradable permit program in January 1994 to reduce nitrogen oxides (NO_x) and sulfur dioxide emissions in the Los Angeles area (Johnson and Pekelney 1996). As of June 1996, 353 participants in this Regional Clean Air Incentives Market (RECLAIM) program, have traded more than 100,000 tons of NO_x and SO_2 emissions, at a value of over $10 million (Brotzman 1996). The RECLAIM program, which operates through the issuance of permits that authorize decreasing levels of pollution over time, governs stationary sources only, but the authority is considering expanding the program to allow trading between stationary and mobile sources (Fulton 1996).

4. ASSESSING THE APPLICATIONS: LIMITED USE OF INSTRUMENTS

Notwithstanding the varying levels of success in the implementation of specific programs, economic-incentive instruments have yet to transform the landscape of environmental policy in fundamental ways. Indeed, economic-incentive instruments still exist only at the fringes of

regulation, and have not become a central component of private firms' environmental decision making. It is beyond the scope of this essay to provide a thorough treatment of the positive political economy of instrument choice in environmental policy (Keohane, Revesz, and Stavins 1997), but it is possible to identify several important factors that have limited the use of economic-incentive policies.

4.1 The Role of Interest Groups

Traditional regulatory programs require regulators with a technical or legal-based skill-set, but economic-incentive instruments require market-trained thinkers, including MBAs, economists, and others. Members of the government bureaucracy are rationally resisting the dissipation of their human capital (Hahn and Stavins 1991). Although some environmental advocacy groups have welcomed the selective use of economic-incentive instruments (Krupp 1986), others are concerned that increased flexibility in environmental regulation will result in the reduction of the overall level of environmental protection. In parts of the environmental community, the sentiment remains that environmental quality is an inalienable right, and that economic-incentive programs condone the "right to pollute." In addition, some of these environmental professionals, like their government counterparts, may be resisting the dissipation of *their* human capital.

4.2 Ambivalence Towards Better Regulation

Many firms applaud economic-incentive instruments in the abstract because of their promise of flexibility and cost savings, but few have actively supported specific incentive-based policies. Much of the hesitation stems from a reluctance to promote any regulation, no matter how flexible or cost effective. Private firms perceive – perhaps seasoned by experience – that political forces beyond their control might unfavorably distort the design and implementation of these instruments.

First, there is a concern that any cost savings will be used to increase the overall degree of environmental clean up. Second, the actual design of instruments may distort their flexibility and penalize some firms. Third, the regulated sector may fear that the rules will change over time. Environmental investments can often be very large (e.g., tens of millions of dollars). For businesses to optimize these investments, regulations not only have to be flexible, but predictable. In the case of acid rain, for example, changes have been proposed by EPA in the permit bidding process, and the American Lung Association has sued EPA in an attempt to force them to tighten SO_2 standards (Lobsenz 1996). Finally, firms are concerned that "buying the right to pollute" under emission trading programs could lead to negative publicity. Even though the trade of permits is legal, and helps improve the environment at lower overall cost to society, some citizens may perceive this behavior as unethical.

4.3 Consumer Experience

The slow penetration of economic-incentive instruments into environmental policies is also a function of these instruments not being well understood by the public. Economic-incentive instruments – especially charges – may suffer from making environmental costs transparent. While encouraging individuals to consciously link environmental costs and benefits may be a good thing,

it can undermine the enthusiasm with which economic-incentive instruments are embraced. These instruments have been an easy target for opponents who paint a picture for consumers that firms are simply paying to pollute. While the fallacy of such arguments is clear, particularly in the context of command-and-control instruments that actually *give away* the right to pollute, the imagery has been compelling.

5. ASSESSING THE APPLICATIONS: A MIXED RECORD ON PERFORMANCE

In section 4, I examined some of the reasons why economic-incentive instruments have not been widely used in the environmental policy arena. In this section, I consider the reasons why, when they have been used, economic-incentive instruments have not always performed as well as predicted.

5.1 Inaccurate Predictions

One of the major reasons economic-incentive instruments have fallen short in delivering the cost savings predicted is that the predictions themselves have often been unrealistic – they were premised on perfect performance under *ideal* conditions. That is, these predictions have implicitly assumed that the cost-minimizing allocation of the pollution-control burden among sources would be achieved, and that marginal abatement costs would be perfectly equated across all sources. In a frequently cited table, Tietenberg calculated the ratio of the cost of an actual command-and-control program to a least-cost benchmark (Tietenberg 1985). Others have mistakenly used this ratio as an indicator of the potential gains of adopting specific economic-incentive instruments. The more appropriate comparison would be between actual command-and-control programs and either actual or reasonably constrained theoretical economic-incentive programs (Hahn and Stavins 1992).

In addition, predictions made during policy debates have typically ignored a number of factors that can adversely affect performance: transaction costs involved in implementing economic-incentive programs (Stavins 1995); uncertainty as to the property rights bestowed under programs; concentration in the permit market (Hahn 1984; Misolek and Elder 1989) or product market (Malueg 1990); the pre-existing regulatory environment (Bohi and Burtraw 1992); and the inability of firms' internal decision-making capabilities to fully utilize program opportunities (Walley and Whitehead 1994).

The SO_2 allowance trading program is a high-profile example where overly optimistic predictions were made. The program was originally predicted to cut the cost of achieving SO_2 reductions by up to $3 billion annually (ICF, Inc. 1986). It is now predicted to result in savings of about $1 billion annually (Hahn and May 1994). The price and quantity of permit trading has been lower than originally predicted, partly because the marginal cost of abatement has been lower than expected for reasons related to changes in input markets, primarily the fall in the price of low-sulfur coal, due to railroad deregulation (Ellerman and Montero 1996) and innovations in fuel blending that have enabled more fuel switching (Burtraw 1995). Furthermore, permit prices may have been lower than marginal abatement costs because of: utilities' reluctance to consider new options; constraints imposed on utilities by contractual precommitments (Coggins and Smith 1993); the preexisting

regulatory environment, including locally binding regulations and rate-of-return regulations; regulatory uncertainty; permit property rights questions (Bohi and Burtraw 1992); and transaction costs.

5.2 Design Problems

Many of the factors cited suggest the need for changes in the design of future economic-incentive instruments. While some program design elements reflect miscalculations of market reactions, others were known to be problematic at the time the programs were enacted, but nevertheless were incorporated into programs to ensure adoption by the political process. One striking example is the adoption of the "20 percent rule" under EPA's Emission Trading Program. This rule, adopted at the insistence of the environmental advocacy community, stipulated that each time a permit is traded, the amount of pollution authorized thereunder is reduced by 20 percent. Since permits that are not traded retain their full value, this regulation discourages trading and thereby increases abatement costs.

5.3 Limitations in Firms' Internal Structures

A third set of explanations for the mixed performance of implemented economic-incentive instruments reflects limitations in private firms' internal structures and skill sets (Hockenstein, Stavins, and Whitehead 1997). Economic-incentive instruments require a very different set of decisions than do traditional command-and-control approaches, and most firms are simply not equipped internally to make the decisions necessary to fully take advantage of these instruments. Since economic-incentive instruments have been used on a limited basis only, and firms are not certain that these instruments will be a lasting component on the regulatory landscape, most firms have not reorganized their environmental, health and safety (EH&S) departments in a manner necessary to exploit fully potential cost savings. Rather, most firms continue to have organizations that are experienced in minimizing the costs of complying with command-and-control regulations, not in making the *strategic* decisions required by economic-incentive instruments (Walley and Whitehead 1994).

In general, EH&S staff members are poorly unequipped to handle emerging environmental issues in a business context. As lawyers and engineers, not MBAs, they are experienced in interpreting detailed regulatory rules and in designing technological solutions to comply with them; they are unprepared to implement the cost-saving decisions that economic-incentive regulations allow. EH&S departments need to be staffed with market-trained thinkers who can analyze the strategic implications of the new options firms face.

Businesses are further impaired by the fact that EH&S functions are not sufficiently integrated with those of the business units. Links have rarely developed between environmental decision-makers and business unit decision-makers. In many cases, environmental costs are not fully measured and are not driven back to the business units from which they derive. This has limited firms' abilities to make even the few strategic decisions allowed under command-and-control approaches. When firms face the much broader set of strategic issues raised by market-instruments, the lack of integration of environmental with business units becomes an even more pressing

problem. Absent this integration, the full potential of economic-incentive instruments – cost-effectiveness and improved incentives for technological change – will not be realized.

6. CONCLUSIONS

Some eighty years ago, Pigou proposed the use of a corrective tax to internalize environmental or other externalities; and fifty years later, the portfolio of potential economic-incentive instruments was expanded to include a quantity-based mechanism, tradeable permits. Thus, economic-incentive approaches to environmental protection are by no means a new policy idea. Over the past two decades they have held varying degrees of prominence in environmental policy. But these instruments remain on the periphery of environmental policy, and when they have been implemented, they frequently have not performed as predicted. Does this suggest that economists and legal scholars should abandon their advocacy of these instruments? The historical record, reviewed in this essay, suggests that the answer is "no."

Economic-incentive instruments have delivered attractive results where implemented and promise additional future benefits. To date, their effectiveness has been undermined by unrealistic expectations, lack of political will, flaws in design, and constraints imposed by the internal structure of firms. These are all remediable. Thus, rather than abandoning the use of market-based instruments, policy makers should direct their efforts to making future applications work better than those that came before.

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