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Economic Prescriptions for Environmental Problems: How the Patient Followed the Doctor's Orders

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One of the dangers with ivory tower theorizing is that it is easy to lose sight of the actual set of problems which need to be solved, and the range of potential solutions. As one who frequently engages in this exercise, I can attest to this fact. In my view, this loss of sight has become increasingly evident in the theoretical structure underlying environmental economics, which often emphasizes elegance at the expense of realism.

In this paper, I will argue that both normative and positive theorizing could greatly benefit from a careful examination of the results of recent innovative approaches to environmental management. The particular set of policies examined here involves two tools which have received widespread support from the economics community: marketable permits and emission charges (Pigou, 1932; Dales, 1968; Kneese and Schultze, 1975). Both tools represent ways to induce businesses to search for lower cost methods of achieving environmental standards. They stand in stark contrast to the predominant "command-and-control" approach in which a regulator specifies the technology a firm must use to comply with regulations. Under highly restrictive conditions, it can be shown that both of the economic approaches share the desirable feature that any gains in environmental quality will be obtained at the lowest possible cost (Baumol and Oates, 1975).

Until the 1960s, these tools only existed on blackboards and in academic journals, as products of the fertile imaginations of academics. However, some countries have recently begun to explore using these tools as part of a broader strategy for managing environmental problems.

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This paper chronicles the experience with both marketable permits and emissions charges. It also provides a selective analysis of a variety of applications in Europe and the United States and shows how the actual use of these tools tends to depart from the role which economists have conceived for them.

The Selection of Environmental Instruments

In thinking about the design and implementation of policies, it is generally assumed that policy makers can choose from a variety of "instruments" for achieving specified objectives. The environmental economics literature generally focuses on the selection of instruments that minimize the overall cost of achieving prescribed environmental objectives.

One instrument which has been shown to supply the appropriate incentives, at least in theory, is marketable permits. The implementation of marketable permits involves several steps. First, a target level of environmental quality is established. Next, this level of environmental quality is defined in terms of total allowable emissions. Permits are then allocated to firms, with each permit enabling the owner to emit a specified amount of pollution. Firms are allowed to trade these permits among themselves. Assuming firms minimize their total production costs, and the market for these permits is competitive, it can be shown that the overall cost of achieving the environmental standard will be minimized (Montgomery, 1972).

Marketable permits are generally thought of as a "quantity" instrument because they ration a fixed supply of a commodity, in this case pollution. The polar opposite of a quantity instrument is a "pricing" instrument, such as emissions charges. The idea underlying emissions charges is to charge polluters a fixed price for each unit of pollution. In this way, they are provided with an incentive to economize on the amount of pollution they produce. If all firms are charged the same price for pollution, then marginal costs of abatement are equated across firms, and this result implies that the resulting level of pollution is reached in a cost-minimizing way.

Economists have attempted to estimate the effectiveness of these approaches. Work by Plott (1983) and Hahn (1983) reveals that implementation of these ideas in a laboratory setting leads to marked increases in efficiency levels over traditional forms of regulation, such as setting standards for each individual source of pollution. The work based on simulations using actual costs and environmental data reveals a similar story. For example, in a review of several studies examining the potential for marketable permits, Tietenberg (1985, pp. 43–44) found that potential control costs could be reduced by more than 90 percent in some cases. Naturally, these results are subject to the usual cautions that a competitive market actually must exist for the results to hold true. Perhaps more importantly, the results assume that it is possible to easily monitor and enforce a system of permits or taxes. The subsequent analysis will suggest that the capacity to monitor and enforce can dramatically affect the choice of instruments.

Following the development of a normative theory of instrument choice, a handful of scholars began to explore reasons why environmental regulations are actually selected. This positive environmental literature tends to emphasize the potential winners and losers from environmental policies as a way of explaining the conditions under which we will observe such policies. For example, Buchanan and Tullock (1975) argue that the widespread use of source-specific standards rather than a fee can be explained by looking at the potential profitability of the affected industry under the two regimes. After presenting the various case studies, I will review some of the insights from positive theory and see how they square with the facts.

The formal results in the positive and normative theory of environmental economics are elegant. Unfortunately, they are not immediately applicable, since virtually none of the systems examined below exhibits the purity of the instruments which are the subject of theoretical inquiry. The presentation here highlights those instruments which show a marked resemblance to marketable permits or emission fees. Together, the two approaches to pollution control span a wide array of environmental problems, including toxic substances, air pollution, water pollution and land disposal.

Marketable Permits

In comparison with charges, marketable permits have not received widespread use. Indeed, there appear to be only four existing environmental applications; three of them in the United States. One involves the trading of emissions rights of various pollutants regulated under the Clean Air Act; a second involves trading of lead used in gasoline; a third addresses the control of water pollution on a river; and a fourth involves air pollution trading in Germany and will not be addressed here because of limited information (see Sprenger, 1986). These programs exhibit dramatic differences in performance, which can be traced back to the rules used to implement these approaches.

Wisconsin Fox River Water Permits

In 1981, the state of Wisconsin implemented an innovative program aimed at controlling biological oxygen demand (BOD) on a part of the Fox River (Novotny, 1986, p. 11).[1] The program was designed to allow for the limited trading of marketable discharge permits. The primary objective was to allow firms greater flexibility in abatement options while still maintaining environmental quality. The program is administered by the state of Wisconsin in accord with the Federal Water Pollution Control Act. Firms are issued five-year permits which define their wasteload allocation. This allocation defines the initial distribution of permits for each firm.

Early studies estimated that substantial savings, on the order of $7 million per year, could result after implementing this trading system (O'Neil, 1983, p. 225).

[1] BOD is a measure of the demand for dissolved oxygen imposed on a water body by organic effluents.

However, actual cost savings have been minimal. In the six years that the program has been in existence, there has been only one trade. Given the initial fanfare about this system, its performance to date has been disappointing.

A closer look at the nature of the market and the rules for trading reveals that the result should not have been totally unexpected. The regulations are aimed at two types of dischargers: pulp and paper plants and municipal waste treatment plants. David and Joeres (1983) note that the pulp and paper plants have an oligopolistic structure, and thus may not behave as competitive firms in the permit market. Moreover, it is difficult to know how the municipal utilities will perform under this set of rules, since they are subject to public utility regulation (Hahn and Noll, 1983). Trading is also limited by location. There are two points on the river where pollution tends to peak, and firms are divided into "clusters" so that trading will not increase BOD at either of these points. There are only about 6 or 7 firms in each cluster (Patterson, 1987). Consequently, markets for wasteload allocations may be quite thin.

In addition, Novotny (1986) has argued that several restrictions on transfers may have had a negative impact on potential trading. Any transaction between firms requires modifying or reissuing permits. Transfers must be for at least a year; however, the life of the permit is only five years. Moreover, parties must waive any rights to the permit after it expires, and it is unclear how trading will affect the permit renewal process. These conditions create great uncertainty over the future value of the property right. Added to the problems created by these rules are the restrictions on eligibility for trades. Firms are required to justify the "need" for permits. This effectively limits transfers to new dischargers, plants which are expanding, and treatment plants that cannot meet the requirements, despite their best efforts. Trades that only reduce operating costs are not allowed. With all the uncertainty and high transactions costs, it is not surprising that trading has gotten off to a very slow start.

While the marketable permit system for the Fox River was being hailed as a success by economists, the paper mills did not enthusiastically support the idea (Novotny, 1986, p. 15). Nor have the mills chosen to explore this option once it has been implemented. Indeed, by almost any measure, this limited permit trading represents a minor part of the regulatory structure. The mechanism builds on a large regulatory infrastructure where permits specifying treatment and operating rules lie at the center. The new marketable permits approach retains many features of the existing standards-based approach. The initial allocations are based on the status quo, calling for equal percentage reductions from specified limits. This "grandfathering" approach has a great deal of political appeal for existing firms. New firms must continue to meet more stringent requirements than old firms, and firms must meet specified technological standards before trading is allowed.

Emissions Trading

By far the most significant and far-reaching marketable permit program in the United States is the emissions trading policy. Started over a decade ago, the policy attempts to provide greater flexibility to firms charged with controlling air pollutant emissions.[2] Because the program represents a radical departure in the approach to pollution regulation, it has come under close scrutiny by a variety of interest groups. Environmentalists have been particularly critical. These criticisms notwithstanding, the Environmental Protection Agency Administrator Lee Thomas (1986) characterized the program as "one of EPA's most impressive accomplishments."

Emissions trading has four distinct elements. Netting, the first program element, was introduced in 1974. Netting allows a firm which creates a new source of emissions in a plant to avoid the stringent emission limits which would normally apply by reducing emissions from another source in the plant. Thus, net emissions from the plant do not increase significantly. A firm using netting is only allowed to obtain the necessary emission credits from its own sources. This is called *internal trading* because the transaction involves only one firm. Netting is subject to approval at the state level, not the federal.

Offsets, the second element of emissions trading, are used by new emission sources in "non-attainment areas." (A non-attainment area is a region which has not met a specified ambient standard.) The Clean Air Act specified that no new emission sources would be allowed in non-attainment areas after the original 1975 deadlines for meeting air quality standards passed. Concern that this prohibition would stifle economic growth in these areas prompted EPA to institute the offset rule. This rule specified that new sources would be allowed to locate in non-attainment areas, but only if they "offset" their new emissions by reducing emissions from existing sources by even larger amounts. The offsets could be obtained through internal trading, just as with netting. However, they could also be obtained from other firms directly, which is called *external trading*.

Bubbles, though apparently considered by EPA to be the centerpiece of emissions trading, were not allowed until 1979. The name derives from the placing of an imaginary bubble over a plant, with all emissions exiting at a single point from the bubble. A bubble allows a firm to sum the emission limits from individual sources of a pollutant in a plant, and to adjust the levels of control applied to different sources as long as this aggregate limit is not exceeded. Bubbles apply to existing sources. The policy allows for both internal and external trades. Initially, every bubble had to be approved at the federal level as an amendment to a state's implementation plan. In 1981, EPA approved a "generic rule" for bubbles in New Jersey which allowed the state to give final approval for bubbles. Since then, several other states have followed suit.

Banking, the fourth element of emissions trading, was developed in conjunction with the bubble policy. Banking allows firms to save emission reductions above and beyond permit requirements for future use in emissions trading. While EPA action was initially required to allow banking, the development of banking rules and the administration of banking programs has been left to the states.

[2]Pollutants covered under the policy include volatile organic compounds, carbon monoxide, sulfur dioxide, particulates, and nitrogen oxides (Hahn and Hester, 1986).

Table 1
Summary of emissions trading activity

Activity		*Estimated number of internal transactions*	*Estimated number of external transactions*	*Estimated Cost savings (millions)*	*Environmental quality impact*
Netting		5,000 to 12,000	None	$25 to $300 in Permitting costs: $500 to $12,000 in emission control costs	Insignificant in individual cases; Probably insignificant in aggregate
Offsets		1800	200	See text	Probably insignificant
Bubbles:	Federally approved	40	2	$300	Insignificant
	State approved	89	0	$135	Insignificant
Banking		< 100	< 20	Small	Insignificant

Source: Hahn and Hester (1986)

The performance of emissions trading can be measured in several ways. A summary evaluation which assesses the impact of the program on abatement costs and environmental quality is provided in Table 1. For each emissions trading activity, an estimate of cost savings, the environmental quality effect, and the number of trades is given. In each case, the estimates are for the entire life of the program. As can be seen from the table, the level of activity under various programs varies dramatically. More netting transactions have taken place than any other type, but all of these have necessarily been internal. The wide range placed on this estimate, 5000 to 12,000, reflects the uncertainty about the precise level of this activity. An estimated 2000 offset transactions have taken place, of which only 10 percent have been external. Fewer than 150 bubbles have been approved. Of these, almost twice as many have been approved by states under generic rules than have been approved at the federal level, and only two are known to have involved external trades. For banking, the figures listed are for the number of times firms have withdrawn banked emission credits for sale or use. While no estimates of the exact numbers of such transactions can be made, upper bound estimates of 100 for internal trades and 20 for external trades indicate the fact that there has been relatively little activity in this area.

Cost savings for both netting and bubbles are substantial. Netting is estimated to have resulted in the most cost savings, with a total of between $525 million to over $12 billion from both permitting and emissions control cost savings.[3] By allowing new or modified sources to locate in areas that are highly polluted, offsets confer a major

[3]The wide range of this estimate reflects the uncertainty which results from the fact that little information has been collected on netting.

economic benefit on firms which use them. While the size of this economic benefit is not easily estimated, it is probably in the hundreds of millions of dollars. Federally approved bubbles have resulted in savings estimated at $300 million, while state bubbles have resulted in an estimated $135 million in cost savings. Average savings from federally approved bubbles are higher than those for state approved bubbles. Average savings from bubbles are higher than those from netting, which reflects the fact that bubble savings may be derived from several emissions sources in a single transaction, while netting usually involves cost savings at a single source. Finally, the cost savings from the use of banking cannot be estimated, but is necessarily small given the small number of banking transactions which have occurred.

The performance evaluation of emissions trading activities reveals a mixed bag of accomplishments and disappointments. The program has clearly afforded many firms flexibility in meeting emission limits, and this flexibility has resulted in significant aggregate cost savings—in the billions of dollars. However, these cost savings have been realized almost entirely from internal trading. They fall far short of the potential savings which could be realized if there were more external trading. While cost savings have been substantial, the program has led to little or no net change in the level of emissions.

The evolution of the emissions trading can best be understood in terms of a struggle over the nature and distribution of property rights. Emissions trading can be seen as a strategy by regulators to provide industry with increased flexibility while offering environmentalists continuing progress toward environmental quality goals. Meeting these two objectives requires a careful balancing act. To provide industry with greater flexibility, EPA has attempted to define a set of property rights that places few restrictions on their use. However, at the same time, EPA has to be sensitive to the concerns of environmentalists and avoid giving businesses too clear a property right to their existing level of pollution. The conflicting interests of these two groups have led regulators to create a set of policies which are specifically designed to deemphasize the explicit nature of the property right. The high transactions costs associated with external trading have induced firms to eschew this option in favor of internal trading or no trading at all.

Like the preceding example of the Fox River, emissions trading is best viewed as an incremental departure from the existing approach. Property rights were grandfathered. Most trading has been internal, and the structure of the Clean Air Act, including its requirement that new sources be controlled more stringently, was largely left intact.

Lead Trading

Lead trading stands in stark contrast to the preceding two marketable permit approaches. It comes by far the closest to an economist's ideal of a freely functioning market. The purpose of the lead trading program was to allow gasoline refiners greater flexibility during a period when the amount of lead in gasoline was being significantly reduced. (For a more detailed analysis of the performance of the lead trading program, see Hahn and Hester, 1987.)

Unlike many other programs, the lead trading program was scheduled to have a fixed life from the outset. Interrefinery trading of lead credits was permitted in 1982. Banking of lead credits was initiated in 1985. The trading program was terminated at the end of 1987. Initially, the period for trading was defined in terms of quarters. No banking of credits was allowed. Three years after initiating the program limited banking was allowed, which allowed firms to carry over rights to subsequent quarters. Banking has been used extensively by firms since its initiation.

The program is notable for its lack of discrimination among different sources, such as new and old sources. It is also notable for its rules regarding the creation of credits. Lead credits are created on the basis of existing standards. A firm does not gain any extra credits for being a large producer of leaded gasoline in the past. Nor is it penalized for being a small producer. The creation of lead credits is based solely on current production levels and average lead content. For example if the standard were 1.1 grams per gallon, and a firm produces 100 gallons of gasoline, it would receive rights entitling it to produce or sell up to 110 (100 × 1.1) grams of lead. To the extent that current production levels are correlated with past production levels, the system acknowledges the existing distribution of property rights. However, this linkage is less explicit than those made in other trading programs.[4]

The success of the program is difficult to measure directly. It appears to have had very little impact on environmental quality. This is because the amount of lead in gasoline is routinely reported by refiners and is easily monitored. The effect the program has had on refinery costs is not readily available. In proposing the rule for banking of lead rights, EPA estimated that resulting savings to refiners would be approximately $228 million (U.S. EPA, 1985a). Since banking activity has been somewhat higher than anticipated by EPA, it is likely that actual cost savings will exceed this amount. No specific estimate of the actual cost savings resulting from lead trading have been made by EPA.

The level of trading activity has been high, far surpassing levels observed in other environmental markets. In 1985, over half of the refineries participated in trading. Approximately 15 percent of the total lead rights used were traded. Approximately 35 percent of available lead rights were banked for future use or trading (U.S. EPA, 1985b, 1986). In comparison, volumes of emissions trading have averaged well below 1 percent of the potential emissions that could have been traded.

From the standpoint of creating a workable regulatory mechanism that induces cost savings, the lead market has to be viewed as a success. Refiners, though initially lukewarm about this alternative, have made good use of this program. It stands out amidst a stream of incentive-based programs as the "noble" exception in that it conforms most closely to the economists' notion of a smoothly functioning market.

Given the success of this market in promoting cost savings over a period in which lead was being reduced, it is important to understand why the market was successful.

[4]One of the reasons EPA set up the allocation rule in this way was to try to transfer some of the permit rents from producers to consumers. This will not always occur, however, and depends on the structure of the permits market as well as the underlying production functions.

The lead market had two important features which distinguished it from other markets in environmental credits. The first was that the amount of lead in gasoline could be easily monitored with the existing regulatory apparatus. The second was that the program was implemented after agreement had been reached about basic environmental goals. In particular, there was already widespread agreement that lead was to be phased out of gasoline. This suggests that the success in lead trading may not be easily transferred to other applications in which monitoring is a problem, or environmental goals are poorly defined. Nonetheless, the fact that this market worked well provides ammunition for proponents of market-based incentives for environmental regulation.

New Directions for Marketable Permits

An interesting potential application for marketable permits has arisen in the area of nonpoint source pollution.[5] In 1984, Colorado implemented a program which would allow limited trading between point and nonpoint sources for controlling phosphorous loadings in Dillon Reservoir (Elmore et al., 1984). Firms receive an allocation based on their past production and the holding capacity of the lake. At this time, no trading between point and nonpoint sources has occurred.

As in the case of the Fox River program, point sources are required to make use of the latest technology before they are allowed to trade. The conventional permitting system is used as a basis for trading. Moreover, trades between point and nonpoint sources are required to take place on a 2 for 1 basis. This means for each gram of phosphorous emitted from a point source under a trade, two grams must be reduced from a nonpoint source. Annual cost savings are projected to be about $800,000 (Kashmanian et al., 1986, p. 14); however, projected savings are not always a good indicator of actual savings, as was illustrated in the case of the Fox River.

EPA is also considering using marketable permits as a way of promoting efficiency in the control of chlorofluorocarbons and halons which lead to the depletion of stratospheric ozone.[6] In its notice of proposed rulemaking, EPA suggested grandfathering permits to producers based on their 1986 production levels, and allowing them to be freely traded. This approach is similar to earlier approaches which the agency adopted for emissions trading and lead trading.

The applications covered in this section illustrate that there are a rich array of mechanisms that come under the heading of marketable permits. The common element seems to be that the primary motivation behind marketable permits is to provide increased flexibility in meeting prescribed environmental objectives. This flexibility, in turn, allows firms to take advantage of opportunities to reduce their expenditures on pollution control without sacrificing environmental quality. However, the rules of the marketable permits can sometimes be so restrictive that the flexibility they offer is more imaginary than real.

[5] Point sources represent sources which are well-defined, such as a factory smokestack. Nonpoint sources refer to sources whose emission points are not readily identified, such as fertilizer runoff from farms.

[6] EPA's decision to use a market-based approach to limit stratospheric ozone depletion is examined in Hahn and McGartland (1988).

Charges in Practice

Charge systems in four countries are examined. Examples are drawn from France, Germany, the Netherlands, and the United States. Particular systems were selected because they were thought to be significant either in their scope, their effect on revenues, or their impact on the cost effectiveness of environmental regulation. While the focus is on water effluent charges, a variety of systems are briefly mentioned at the end of this section which cover other applications.

Charges in France

The French have had a system of effluent charges on water pollutants in place since 1969 (Bower et al., 1981). The system is primarily designed to raise revenues which are then used to help maintain or improve water quality. Though the application of charges is widespread, they are generally set at low levels.[7] Moreover, charges are rarely based on actual performance. Rather, they are based on the expected level of discharge by various industries. There is no explicit connection between the charge paid by a given discharger and the subsidy received for reducing discharges (Bower et al., 1981, p. 126). However, charges are generally earmarked for use in promoting environmental quality in areas related to the specific charge. The basic mechanism by which these charges improve environmental quality is through judicious earmarking of the revenues for pollution abatement activities.

In evaluating the charge system, it is important to understand that it is a major, but by no means dominant, part of the French system for managing water quality. Indeed, in terms of total revenues, a sewage tax levied on households and commercial enterprises is larger in magnitude (Bower et al., 1981, p. 142). Moreover, the sewage tax is assessed on the basis of actual volumes of water used. Like most other charge systems, the charge system in France is based on a system of water quality permits, which places constraints on the type and quantity of effluent a firm may discharge. These permits are required for sources discharging more than some specified quantity (Bower et al., 1981, p. 130).

Charges now appear to be accepted as a way of doing business in France. They provide a significant source of revenues for water quality control. One of the keys to their initial success appears to have been the gradual introduction and raising of charges. Charges started at a very low level and were gradually raised to current levels (Bower et al., 1981, p. 22). Moreover, the pollutants on which charges are levied has expanded considerably since the initial inception of the charge program.[8]

Charges in Germany

The German system of effluent charges is very similar to the French system. Effluent charges cover a wide range of pollutants, and the charges are used to cover

[7] Charges cover a wide variety of pollutants, including suspended solids, biological oxygen demand, chemical oxygen demand, and selected toxic chemicals.

[8] For example, Brown (1984, p. 114) notes that charges for nitrogen and phosphorous were added in 1982.

administrative expenses for water quality management and to subsidize projects which improve water quality (Brown and Johnson, 1984, p. 934, 939, 945). The bills that industry and municipalities pay are generally based on expected volume and concentration (Brown and Johnson, 1984, p. 934). Charges vary by industry type as well as across municipalities. Charges to industries and municipalities depend on several variables, including size of the municipality, desired level of treatment, and age of equipment (Brown and Johnson, 1984, pp. 934, 938).

Charges have existed in selected areas of Germany for decades (Bower et al., 1981, p. 299). Management of water quality is delegated to local areas. In 1981, a system of nationwide effluent charges was introduced (Bower et al., 1981, p. 226). The federal government provided the basic framework in its 1976 Federal Water Act and Effluent Charge Law (Brown and Johnson, 1984, p. 930). Initially, industry opposed widespread use of charges. But after losing the initial battle, industry focused on how charges would be determined and their effective date of implementation (Brown and Johnson, 1984, p. 932). While hard data are lacking, there is a general perception that the current system is helping to improve water quality.

Charges in the Netherlands

The Netherlands has had a system of effluent charges in place since 1969 (Brown and Bresssers, 1986, p. 4). It is one of the oldest and best administered charge systems, and the charges placed on effluent streams are among the highest. In 1983, the effluent charge per person was $17 in the Netherlands, $6 in Germany, and about $2 in France (Brown and Bressers, 1986, p. 5). Because of the comparatively high level of charges found in the Netherlands, this is a logical place to examine whether charges are having a discernible effect on the level of pollution. Bressers (1983), using a multiple regression approach, argues that charges have made a significant difference for several pollutants. This evidence is also buttressed by surveys of industrial polluters and water board officials which indicate that charges had a significant impact on firm behavior (Brown and Bressers, 1986, pp. 12–13). This analysis is one of the few existing empirical investigations of the effect of effluent charges on resulting pollution.

The purpose of the charge system in the Netherlands is to raise revenue that will be used to finance projects that will improve water quality (Brown and Bressers, 1986, p. 4). Like its counterparts in France and Germany, the approach to managing water quality uses both permits and effluent charges for meeting ambient standards (Brown and Bressers, 1986, p. 2).[9] Permits tend to be uniform across similar discharges. The system is designed to ensure that water quality will remain the same or get better (Brown and Bressers, 1986 p. 2). Charges are administered both on volume and concentration. Actual levels of discharge are monitored for larger polluters, while small polluters often pay fixed fees unrelated to actual discharge (Bressers, 1983, p. 10).

[9]Emission and effluent standards apply to individual sources of pollution while ambient standards apply to regions such as a lake or an air basin.

Charges have exhibited a slow but steady increase since their inception (Brown and Bressers, 1986, p. 5). This increase in charges has been correlated with declining levels of pollutants. Effluent discharge declined from 40 population equivalents in 1969 to 15.3 population equivalents in 1980, and it was projected to decline to 4.4 population equivalents in 1985 (Brown and Bressers, 1986, p. 10). Thus, over 15 years, this measure of pollution declined on the order of 90 percent.

As in Germany, there was initial opposition from industry to the use of charges. Brown and Bressers (1986, p. 4) also note opposition from environmentalists, who tend to distrust market-like mechanisms. Nonetheless, charges have enjoyed widespread acceptance in a variety of arenas in the Netherlands.

One final interesting feature of the charge system in the Netherlands relates to the differential treatment of new and old plants. In general, newer plants face more stringent regulation than older plants (Brown and Bressers, 1986, p. 10). As we shall see, this is also a dominant theme in American regulation.

Charges in the United States

The United States has a modest system of user charges levied by utilities that process wastewater, encouraged by federal environmental regulations issued by the Environmental Protection Agency. They are based on both volume and strength, and vary across utilities. In some cases, charges are based on actual discharges, and in others, as a rule of thumb, they are related to average behavior. In all cases, charges are added to the existing regulatory system which relies heavily on permits and standards.

Both industry and consumers are required to pay the charges. The primary purpose for the charges is to raise revenues to help meet the revenue requirements of the treatment plants, which are heavily subsidized by the federal government. The direct environmental and economic impact of these charges is apparently small (Boland, 1986, p. 12). They primarily serve as a mechanism to help defray the costs of the treatment plants. Thus, the charges used in the United States are similar in spirit to the German and French systems already described. However, their size appears to be smaller, and the application of the revenues is more limited.

Other Fee-Based Systems and Lessons

There are a variety of other fee-based systems which have not been included in this discussion. Brown (1984) did an analysis of incentive-based systems to control hazardous wastes in Europe and found that a number of countries had adopted systems, some of which had a marked economic effect. The general trend was to use either a tax on waste outputs or tax on feedstocks that are usually correlated with the level of waste produced. Companies and government officials were interviewed to ascertain the effects of these approaches. In line with economic theory, charges were found to induce firms to increase expenditures on achieving waste reduction through a variety of techniques including reprocessing of materials, treatment, and input and output substitution. Firms also devoted greater attention to separating waste streams because prices for disposal often varied by the type of waste stream.

The United States has a diverse range of taxes imposed on hazardous waste streams. Several states have land disposal taxes in place. Charges exhibit a wide degree of variation across states. For example, in 1984, charges were $14/tonne in Wisconsin and $70.40/tonne in Minnesota (U.S. CBO, 1985, p. 82). Charges for disposal at landfills also vary widely. The effect of these different charges is very difficult to estimate because of the difficulty in obtaining the necessary data on the quantity and quality of waste streams, as well as the economic variables.

The preceding analysis reveals that there are a wide array of fee-based systems in place designed to promote environmental quality. In a few cases, the fees were shown to have a marked effect on firm behavior; however, in the overwhelming majority of cases studied, the direct economic effect of fees appears to have been small. Several patterns repeat themselves through these examples.

First, the major motivation for implementing emission fees is to raise revenues, which are then usually earmarked for activities which promote environmental quality.[10] Second, most charges are not large enough to have a dramatic impact on the behavior of polluters. In fact, they are not designed to have such an effect. They are relatively low and not directly related to the behavior of individual firms and consumers. Third, there is a tendency for charges to increase faster than inflation over time. Presumably, starting out with a relatively low charge is a way of testing the political waters as well as determining whether the instrument will have the desired effects.

Implementing Market-Based Environmental Programs

An examination of the charge and marketable permits schemes reveals that they are rarely, if ever, introduced in their textbook form. Virtually all environmental regulatory systems using charges and marketable permits rely on the existing permitting system. This result should not be terribly surprising. Most of these approaches were not implemented from scratch; rather, they were grafted onto regulatory systems in which permits and standards play a dominant role.

Perhaps as a result of these hybrid approaches, the level of cost savings resulting from implementing charges and marketable permits is generally far below their theoretical potential. Cost savings can be defined in terms of the savings which would result from meeting a prescribed environmental objective in a less costly manner. As noted, most of the charges to date have not had a major incentive effect. We can infer from this that polluters have not been induced to search for a lower cost mix of meeting environmental objectives as a result of the implementation of charge schemes. Thus, it seems unlikely that charges have performed terribly well on narrow efficiency grounds. The experience on marketable permits is similar. Hahn and Hester (1986)

[10]The actual application of fees is similar in spirit to the more familiar deposit-refund approaches that are used for collecting bottles and cans.

argue that cost savings for emissions trading fall far short of their theoretical potential. The only apparent exception to this observation is the lead trading program, which has enjoyed very high levels of trading activity.

The example of lead trading leads to another important observation; in general, different charge and marketable permit systems exhibit wide variation in their effect on economic efficiency. On the whole, there is more evidence for cost savings with marketable permits than with charges.

While the charge systems and marketable permit systems rarely perform well in terms of efficiency, it is important to recognize that their performance is broadly consistent with economic theory. This observation may appear to contradict what was said earlier about the departure of these systems from the economic ideal. However, it is really an altogether different observation. It suggests that the performance of the markets and charge systems can be understood in terms of basic economic theory. For example, where barriers to trading are low, more trading is likely to occur. Where charges are high and more directly related to individual actions, they are more likely to affect the behavior of firms or consumers.

If these instruments are to be measured by their effect on environmental quality, the results are not very impressive. In general, the direct effect of both charges and marketable permits on environmental quality appears to be neutral or slightly positive. The effect of lead trading has been neutral in the aggregate. The effect of emissions trading on environmental quality has probably been neutral or slightly positive. The direct effect of charges on polluter incentives has been modest, although the indirect environmental effect of spending the revenue raised by charges has been significant.

The evidence on charges and marketable permits points to an intriguing conclusion about the nature of these instruments. Charges and marketable permits have played fundamentally different roles in meeting environmental objectives. Charges are used primarily to improve environmental quality by redistributing revenues. Marketable permits are used primarily to promote cost savings.

The positive theory of instrument choice as it relates to pollution control has been greatly influenced by the work of Buchanan and Tullock (1975). They argue that firms will prefer emission standards to emission taxes because standards result in higher profits. Emission standards serve as a barrier to entry to new firms, thus raising firm profits. Charges, on the other hand, do not preclude entry by new firms, and also represent an additional cost to firms. Their argument is based on the view that industry is able to exert its preference for a particular instrument because it is more likely to be well-organized than consumers.

While this argument is elegant, it misses two important points. The first is that within particular classes of instruments, there is a great deal of variation in the performance of instruments. The second is that most solutions to problems involve the application of multiple instruments. Thus, while the Buchanan and Tullock theory explains why standards are chosen over an idealized form of taxes, it does little to help explain the rich array of instruments that are observed in the real world. In particular, under what situations would we be likely to observe different mixes of instruments?

Several authors have explored these different issues for instrument choice within this basic framework (Coelho, 1976; Dewees, 1983; Yohe, 1976). The basic insight of this work is that the argument that standards will be preferred to taxes depends on the precise nature of the instruments being compared.

Another weakness in the existing theory is that the instruments are not generally used in the way that is suggested by the theory. Most emissions charges, for example, are used as a revenue raising device for subsidizing abatement activity, but a few also have pronounced direct effects on polluters. Most marketable permit approaches are not really designed to create markets. Moreover, the different types of trading schemes perform with widely varying success.

The data from the examples given earlier can be used to begin to piece together some of the elements of a more coherent theory of instrument choice. For example, it is clear that distributional concerns play an important role in the acceptability of user charges. The revenue from such charges is usually earmarked for environmental activities related to those contributions. Thus, charges from a noise surcharge will be used to address noise pollution. Charges for water discharges will be used to construct treatment plants and subsidize industry in building equipment to abate water pollution. This pattern suggests that different industries want to make sure that their contributions are used to address pollution problems for which they are likely to be held accountable. Thus, industry sees it as only fair that, as a whole, they get some benefit from making these contributions.

The "recycling" of revenues from charges points up the importance of the existing distribution of property rights. This is also true in the case of marketable permits. The "grandfathering" of rights to existing firms based on the current distribution of rights is an important focal point in many applications of limited markets in pollution rights (Rolph, 1983; Welch, 1983). All the marketable permit programs in the United States place great importance on the existing distribution of rights.

In short, all of the charge and marketable permit systems described earlier place great importance on the status quo. Charges, when introduced, tend to be phased in. Marketable permits, when introduced, usually are optional in the sense that existing firms can meet standards through trading of permits or by conventional means. In contrast, new or expanding firms are not always afforded the same options. For example, new firms must still purchase emission credits if they choose to locate in a non-attainment area, even if they have purchased state-of-the-art pollution control equipment and will pollute less than existing companies. This is an example of a "bias" against new sources. While not efficient from an economic viewpoint, this pattern is consistent with the political insight that new sources don't "vote" while existing sources do.

Though the status quo is important in all applications studied here, it does not explain by itself the rich variety of instruments that are observed. For example, there has been heated controversy over emissions trading since its inception, but comparatively little controversy over the implementation of lead trading. How can economists begin to understand the difference in attitudes towards these two programs?

There are several important differences between emissions trading and lead trading. In the case of lead standards, there appears to be agreement about the distribution of property rights, and the standard that defined them. Refiners had the right to put lead in gasoline at specified levels during specified time periods. Lead in gasoline was reduced to a very low level at the end of 1987. In contrast to lead, there is great disagreement about the underlying distribution of property rights regarding emissions trading. Environmentalists continue to adhere to the symbolic goal of zero pollution. Industry believes and acts as if its current claims on the environment, without any emission reductions, represent a property right.

In the case of lead trading, output could be relatively easily monitored using the existing regulatory apparatus. This was not the case for emissions trading. A new system was set up for evaluating proposed trades. This was, in part, due to existing weaknesses in the current system of monitoring and enforcement. It was also a result of concerns that environmentalists had expressed about the validity of such trades.

The effect that emissions trading was likely to have on environmental quality was much less certain than that of the lead trading program. Some environmentalists viewed emissions trading as a loophole by which industry could forestall compliance, and Hahn and Hester (1986) found some evidence that bubbles were occasionally used for that purpose. The effects of lead trading were much more predictable. Until 1985, there was no banking, so the overall temporal pattern of lead emissions remained unchanged under the program. With the addition of banking in 1985, this pattern was changed slightly, but within well-defined limits.

To accommodate these differing concerns, different rules were developed for the two cases. In the case of lead trading, rights are traded on a one-for-one basis. In contrast, under emissions trading, rights are not generally traded on a one-for-one basis. Rather, most trades must show a net improvement in environmental quality. In the case of lead, all firms are treated equally from the standpoint of trading. In the case of emissions trading, new firms must meet stringent standards before being allowed to engage in trading.

This comparison suggests is that it is possible to gain important insights into the likely performance and choice of instruments by understanding the forces that led to their creation. Analyzing the underlying beliefs about property rights to pollution may be vital both for the political success of the measure and for how well it works in terms of pure economic efficiency.

This view of efficiency is similar to, but should not be confused with, the notion of efficiency advanced by Becker (1983). Becker argues that government will tend to choose mechanisms which are more efficient over those which are less efficient in redistributing revenues from less powerful to more powerful groups. To the extent that his argument is testable, I believe it is not consistent with the facts. For example, the U.S. currently has a policy that directs toxic waste dumps to be cleaned up in priority order. The policy makes no attempt to examine whether a greater risk reduction could be attained with a different allocation of expenditures. Given a finite budget constraint, this policy does not make sense from a purely economic viewpoint. However, it might make sense if environmentalists hoped that more stringent policies would emerge in the future. Or it might make sense if Congress wants to be perceived as doing the job "right," even if only a small part of the job gets done.

A second example can be drawn from emissions trading. It is possible to design marketable permit systems which are more efficient and ensure better environmental quality over time (Hahn and Noll, 1982; Hahn, 1987), yet these systems have not been implemented. Environmentalists may be reluctant to embrace market alternatives because they fear it may give a certain legitimacy to the act of polluting. Moreover, they may not believe in the expected results. Thus, for Becker's theory to hold in an absolute sense, it would be necessary to construct fairly complicated utility functions. The problem is that the theory does not explicitly address how choices are made by lobbyists, legislators and bureaucrats (Campos, 1987).

These choices may be made in different ways in different countries. How can it be explained, for example, that a large array of countries use fees, while only two countries use marketable permits (and the application of permits in Germany is fairly limited)? Noll (1983) has argued that the political institutions of different countries can provide important clues about regulatory strategy. In addition, the comparison of lead trading and emissions trading revealed that the very nature of the environmental problem can have an important effect on interest group attitudes.

Interest group attitudes can be expected to vary across countries. In the Netherlands, Opschoor (1986, p. 15) notes that environmental groups tend to prefer charges while employer groups prefer regulatory instruments. Barde (1986, pp. 10–11) notes that the political "acceptability" of charges is high in both France and the Netherlands. Nonetheless, some French airlines have refused to pay noise charges because the funds are not being used (Barde, 1986, p. 12). In Italy, there has been widespread opposition from industry and interest groups (Panella, 1986, pp. 6, 22). While German industry has accepted the notion of charges, some industries have criticized the differential charge rates across jurisdictions. In the United States, environmentalists have shown a marked preference for regulatory instruments, eschewing both charges and marketable permits. These preferences may help to explain the choice of instruments in various countries as well as the relative utilization of different instruments. In addition, interest groups in different countries will share different clusters of relevant experiences, which will help to determine the feasible space for alternatives.

In short, existing theories could benefit from more careful analysis of the regulatory status quo, underlying beliefs about property rights, and how political choices are actually made in different countries.

The review of marketable permits and charge systems has demonstrated that regulatory systems involving multiple instruments are the rule rather than the exception. The fundamental problem is to determine the most appropriate mix, with an eye to both economic and political realities.

In addition to selecting an appropriate mix of instruments, attention needs to be given to the effects of having different levels of government implement selected policies. It might seem, for example, that if the problem is local, then the logical choice for addressing the problem is the local regulatory body. However, this is not always true. Perhaps the problem may require a level of technical expertise that does not reside at the local level, in which case some higher level of government involvement may be required. What is clear from a review of implementing environmental policies is that the level of oversight can affect the implementation of policies. For example, Hahn and Hester (1986) note that a marked increase in bubble activity is associated with a decrease in federal oversight.

Because marketable permit approaches have been shown to have a demonstrable effect on cost savings without sacrificing environmental quality, this instrument can be expected to receive more widespread use. One factor which will stimulate the application of this mechanism is the higher marginal costs of abatement that will be faced as environmental standards are tightened. A second factor which will tend to stimulate the use of both charges and marketable permits is a "demonstration effect." Several countries have already implemented these mechanisms with some encouraging results. The experience gained in implementing these tools will stimulate their use in future applications. A third factor which will affect the use of both of these approaches is the technology of monitoring and enforcement. As monitoring costs go down, the use of mechanisms such as direct charges and marketable permits can be expected to increase. The combination of these factors leads to the prediction that greater use of these market-based environmental systems will be made in the future.

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