The Use of Economic Incentives in Developing Countries: Lessons from International Experience with Industrial Air Pollution

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Abstract

To what extent should developing countries eschew conventional command and control environmental regulation that is increasingly seen as inefficient and rely instead on economic incentives? This paper addresses this question as it pertains to industrial air pollution. The paper discusses the advantages and disadvantages of various economic incentive instruments, presents in-depth case studies of their application in Sweden, the United States, China, and Poland, and proposes a number of policy guidelines. We argue that both design deficiencies and pervasive constraints on monitoring and enforcement impede the effectiveness of economic instruments in developing countries. The latter are difficult to rectify, at least in the medium term. As a result, tradable permits are generally not practical. Suitably modified however, emissions fee policies probably are appropriate. They can provide a foundation for a transition to an effective economic incentive system, and can raise much needed revenue for environmental projects and programs. In addition, if political opposition can be overcome, environmental taxes constitute a second-best but potentially effective pollution control instrument.

Key Words: environmental policy, economic incentives, market-based instruments, developing countries, air pollution, Sweden, China, Poland

JEL Classification Numbers: Q25, Q28

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Allen Blackman and Winston Harrington*

1. INTRODUCTION

Financial, institutional, and political constraints make environmental regulation in developing countries far more problematic than in industrialized countries. Yet policy makers faced with the daunting task of crafting effective regulatory regimes in the developing countries have at least one advantage over their historical counterparts in the West--they have several decades of environmental regulatory history to learn from. What lessons can be distilled from this history? In particular, does it imply that developing countries should eschew conventional command and control policies that, while often effective, are increasingly seen as inefficient, and should rely instead on economic incentive instruments such as emissions fees, emissions permits, and taxes on dirty inputs?

This paper reviews international experiences with economic incentive instruments to distill lessons for developing countries. Many aspects of these experiences vary across different types of pollution sources (e.g., industrial versus non-point, air versus water). Given space limitations, an all-encompassing discussion would necessarily sacrifice considerable depth. Therefore, we focus on industrial air pollution. The paper is organized as follows. The next section discusses constraints on environmental regulation in developing countries. The third section describes the different types of economic incentive regulatory instruments and briefly reviews the advantages and disadvantages of each. The fourth section presents a brief overview of the economic incentive policies in OECD countries followed by four in-depth case studies. The last section summarizes and develops policy recommendations.

2. CONSTRAINTS ON ENVIRONMENTAL REGULATION IN DEVELOPING COUNTRIES

To varying degrees, all of the types of environmental regulation discussed in this paper require a public-sector institution capable of establishing rules of conduct for polluters, monitoring performance with respect to these rules, and enforcing compliance.[1] In many

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[1] We focus on conventional regulatory instruments that depend on public sector monitoring and enforcement. Alternative "informal" regulatory instruments that shift some of the burden for monitoring and enforcement onto the private sector are beyond the scope of this paper. For a review see Afsah, LaPlante and Wheeler (1996); Pargal and Wheeler (1996); and Tietenberg (1998).

developing countries, a number of financial and institutional constraints undermine such capabilities. The literature has identified four key constraints (e.g., Eskeland and Jimenez, 1992; OECD, 1993; Afsah, LaPlant and Wheeler, 1996; Krupnick, 1997). First, public sentiment generally favors economic development over environmental protection. In addition, private-sector environmental advocacy--historically a critical stimulus to effective environmental regulation--is generally less prevalent and less well-organized than in industrialized countries. As a result, it is often difficult to muster the political will to enforce environmental regulations. Second, environmental regulatory institutions, along with complementary judicial, legislative and data collection institutions, are generally much weaker than in industrialized countries. Third, fiscal and technical resources for environmental protection are generally in short supply. Finally, production is often dominated by hard-to-monitor small-scale firms.

As discussed below, some regulatory instruments are more robust to these four constraints than others. In the next section we discuss several different types of regulatory instruments paying particular attention to this issue.

3. ECONOMIC INCENTIVE INSTRUMENTS

Environmental regulatory instruments can be classified according to two criteria: (i) whether they dictate how much to abate and what abatement technology to use or simply create financial incentives for firms to abate, and (ii) whether they require the regulator to monitor emissions. Regulatory instruments that dictate abatement decisions are known as "command and control" (CAC) regulations. Examples include emissions standards and technology standards. Policies that create financial incentives for abatement by putting an explicit or implicit price on emissions but which do not dictate abatement decisions are referred to as "economic incentive" (EI) policies. The three chief examples of EI policies are emissions fees, wherein firms pay a fee per unit of emissions; marketable permits, wherein firms are assigned "allowances" to emit a certain amount of pollution which they may trade with other firms if they wish; and environmental taxes, which are simply taxes on the inputs used by polluters or outputs produced by them.[2] Policies that require the regulator to monitor emissions are called "direct" instruments and policies that do not are called "indirect" instruments. Emissions standards, emissions fees, and marketable permits are examples of direct instruments while environmental taxes and technology standards are examples of indirect instruments. The two criteria discussed above imply a classification scheme that is summarized in Table 1 (Eskeland and Jimenez, 1992).

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[2] Note that our use of the terms "fee" and "tax" is somewhat arbitrary--these terms are used interchangeably in the literature. Nevertheless, here we will use the term "fee" to refer only to charges on emissions, and the term "tax" to refer only to charges on pollution intensive inputs and outputs.

Table 1. A classification of environmental regulatory instruments

	Direct Instruments	*Indirect Instruments*
Economic Incentives	• emissions fees • marketable permits	• environmental taxes
Command and Control	• emissions standards	• technology standards

3.1 An Indirect Economic Incentive Instrument: Environmental taxes

Given the constraints on environmental regulation discussed in Section 2, indirect instruments like environmental taxes may stand a better chance of being effective since by definition they are less demanding of regulators than direct instruments. There are three types of environmental taxes: taxes on final products associated with pollution (such as motor vehicles), taxes on goods which are generally used as inputs into a polluting activity (such as coal), and taxes on polluting substances contained in inputs (such as the sulfur contained in coal).[3] Each of these types of taxes has advantages and disadvantages which we discuss below.

Fiscal and environmental impacts. Environmental taxes can have two types of advantageous impacts: fiscal (i.e., they raise revenue) and environmental. Unfortunately, these two impacts are inversely related. Which impact dominates depends on the elasticity of demand for the taxed good. For example, consider a tax on gasoline. If demand for gasoline is inelastic (i.e., price increases have little effect on demand), then the tax will generate revenue, but will not significantly reduce gasoline consumption or vehicular emissions.[4] But if demand for gasoline is elastic (i.e., price increases significantly curtail demand), the tax will generate relatively little revenue but will reduce gasoline consumption and (presumably) vehicular emissions. Demand is more elastic the more widely available are substitutes. Therefore, demand is more elastic when taxes are narrowly targeted. For example, demand for high-sulfur coal is likely to be more elastic than demand for coal, and demand for leaded gasoline more elastic than demand for gasoline. Hence, narrowly targeted taxes are more likely to have a significant environmental impact than broadly targeted taxes. In addition, for most goods, demand is more elastic in the long run than in the short run, since consumers

[3] While technically not environmental taxation, removal of subsidies on goods linked with pollution has the same impact. In Central and Eastern Europe, for example, removing long standing fuel subsidies initiated during the Soviet period has probably done more to improve environmental quality than any explicit environmental policy. See, e.g., Reid and Goldenberg (1998) and Larsen and Shah (1992). However, subsidies do not always affect environmental quality adversely. Eskeland et al. (1994) found that energy subsidies in Indonesia tended to *favor* the use of cleaner fuels, and that the removal of these subsidies could have adverse environmental effects.

[4] The price elasticity of demand is a measure of the responsiveness of the demand for a good to changes in the price of the good. Specifically, it indicates the percent change in demand due to a one percent change in price.

have more time to substitute in the long run. Therefore, the dominant impact of an environmental tax is likely to be fiscal in the short run and environmental in the long run.

The chief motivation for most environmental taxes has generally been revenue generation. Much of the debate about tax and fee revenue concerns whether or not it should be earmarked for environmental expenditures (e.g., OECD, 1993 and 1996).[5] The main argument against earmarking is that it limits the discretion of the government to allocate revenue to different uses. The optimal amount of expenditure on pollution control may be more or less than environmental tax and fee revenue. For example, in some countries, the highest valued use for such revenue may be poverty reduction, not pollution control. In addition, earmarking may encourage rent seeking in the sectors targeted for subsidies.[6] Nevertheless, earmarking is popular because it makes environmental taxes and fees more politically palatable (by returning revenue to those who are disadvantaged by these instruments) and because it is seen as a means of correcting for market failures that prevent firms from obtaining the investment credit.[7]

Ease of administration. Environmental taxes are relatively easy to administer for several reasons. First, quantities of goods are usually much easier to monitor than quantities of emissions. Second, environmental taxes operate through government tax collection institutions rather than environmental regulatory institutions, and in most developing countries, the former are more established and effective than the later. In fact, taxes on fuels are already quite common in developing countries (Sterner, 1996). Finally, planners trying to reduce aggregate emissions by a certain amount need less information to set the requisite tax than to set the requisite fee. To set the right fee, planners need to know firms' marginal abatement costs. To set the right tax, they only need to know the price elasticity of demand for the good in question, a parameter that can generally be easily estimated using historical market data.

Incentives. Unfortunately, environmental taxes entail a number of disadvantages. Most important they do not create incentives to abate emissions per se, only to limit purchases of a good linked with emissions. This problem is mitigated to the extent there is a direct and

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[5] For example, in Central and Eastern Europe environmental taxes revenue has been used to create "environmental funds" that finance investment in pollution control as well as in public sector regulatory capacity building. For a review, see Anderson and Zylicz (1996).

[6] The U.S. experience with gasoline taxation provides an instructive example. By law, all federal (and most state) gasoline taxes are earmarked for transportation purposes, an allocation that has encouraged the development of powerful interest groups at both the state and federal level fighting strenuously to prevent the use of fuel tax revenues for non-transportation (and indeed non-highway) purposes. See Nivola and Crandall (1994).

[7] The advisability of using revenue from environmental taxes to reduce other taxes has also received considerable attention. This debate has focused on the extent to which a revenue-neutral swap of environmental taxes for pre-existing distortionary taxes such as taxes on labor would increase social welfare by both reducing pollution and curtailing economic distortions (the so-called "double dividend"). Some research has suggested that, under some circumstances, environmental taxes exacerbate pre-existing tax distortions. As yet there is no consensus on this issue. For a review of the literature, see Parry (1998).

predictable the link between the taxed good and emissions. Thus, from the standpoint of incentives, a tax on the polluting content of a good is preferable to a tax on an input, and a tax on an input is preferable to a tax on a final product. For example, of the three different types of environmental taxes that could be used to reduce emissions from power plants, a tax on the final product, electricity, can reduce emissions by reducing electricity demand (and hence electricity production), but can not create incentives to cut emissions per unit of electricity generated. A tax on a polluting input, coal, can do this, but can not create incentives to use clean coal. A tax on the polluting content of an input, such as a tax on the sulfur content of coal, can do this but even this type of environmental tax can not create incentives to install end-of-the-pipe pollution abatement equipment, since plants with such equipment pay the same unit tax as those without them.[8]

Targeting. A related disadvantage of environmental taxes is that they may affect non-targeted activities. For example, a tax on coal intended to reduce sulfur emissions from combustion will affect chemical manufacturers who use coal as a feedstock, not as a fuel. Here again, a tax on the polluting content of a good is preferable to a tax on an input. To continue the above example, a tax on the sulfur content of coal would enable chemical manufacturers to reduce their tax liabilities by switching to low-sulfur coal. One solution to the targeting problem is to exempt certain types of consumers from the tax. However, this policy may encourage the creation of black markets for the taxed good.[9]

Political barriers. A third disadvantage of environmental taxes is that they may be less politically acceptable than some other regulatory instruments. While the costs of CAC regulations are largely hidden and directly incurred only by polluters (ignoring the indirect costs consumers pay in higher prices), the costs associated with environmental taxes are highly visible and also, as noted above, are sometimes directly incurred by non-polluters (Drayton, 1978).

Distributional impacts. Finally, environmental taxes may have adverse distributional impacts, that is, they may have a more severe impact on poor households than on rich ones (Eskeland and Kong, 1998). Such impacts are likely to be exacerbated when the taxed good is a necessity item with few substitutes (such as gasoline). Here again, the problem is mitigated to the extent that the tax is narrowly targeted rather than broad-based. Distributional impacts may be redressed by using tax revenue to finance new expenditures which benefit poorer households, or to cut other regressive taxes.

[8] One solution to this incentive problem is to combine an environmental tax with a second indirect instrument--a technology standard. The resulting hybrid policy can be more effective and more efficient than either the tax or the standard alone (Eskeland and Devarajan, 1996).

[9] A related disadvantage is that taxes are difficult to differentiate geographically. Non-uniform taxes may encourage black markets.

3.2 Direct Economic Incentive Instruments: Emissions fees and marketable permits

3.2.1 Efficiency and flexibility

Economists have long argued that direct EI instruments are superior to CAC instruments in terms of static efficiency, dynamic efficiency, and flexibility--advantages that would appear to be especially attractive to developing countries given that financial and administrative resources are typically in short supply. Since these properties are well known (see e.g., Bohm and Russell, 1985), we will only touch on them briefly here.

Static efficiency. The static efficiency advantages of direct EI instruments stem in part from the fact that they leave firms free to choose abatement technologies that minimize costs given their *individual* circumstances. By contrast, under CAC technology and emissions standards, the regulator more or less dictates that whole classes of firms choose certain technologies.[10] Perhaps more important, direct EI instruments create incentives for individual firms to choose levels of abatement that minimize the aggregate costs of achieving a given level of environmental quality. Specifically, firms with low abatement costs are driven to undertake more abatement than those with higher abatement costs.[11] This type of behavior is probably sufficient by itself to make direct EI policies more cost-effective than most CAC policies. For a CAC policy to achieve the same result, the central authority must know the marginal abatement cost of every polluter which is extremely unlikely in practice.[12]

Dynamic efficiency. Although advocates of direct EI instruments generally focus on static efficiency arguments, the advantages of dynamic efficiency and flexibility may be of greater long-run importance. Because firms in direct EI programs can always increase profits

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[10] Though CAC emissions standards do not explicitly dictate firms technology decisions, in practice they usually create strong incentives for firms to choose only officially sanctioned technologies. Therefore, they can be regarded as "technology forcing." In the United States, emissions standards on point sources administered under both the Clean Air Act (e.g., Lowest Achievable Emissions Rates) and the Clean Water Act (e.g., effluent guidelines) are developed with reference to the abatement capabilities of specific technologies. Hence, firms that want to minimize their risks of being found in violation of such standards will want to adopt the technologies underlying the standards. The risk of paying a high penalty for using alternative approaches turns a *de jure* emissions standard into a *de facto* technology standard.

[11] In emissions fee programs, firms whose marginal abatement costs are lower than the fee will abate, while those whose marginal abatement costs are higher than the fee will not abate; they will pay the emissions fee instead. In marketable permit programs, firms with abatement costs below the market price of permits will abate and sell their emissions permits while those with marginal abatement costs above the permit price will not abate; they will purchase permits instead. More technically, in both types of EI programs, in theory the marginal abatement costs of all firms are equated; each source abates to the point where the marginal cost of further abatement equals the either the fee or the permit price. The equalization of marginal costs is a necessary condition, in the standard theoretical model, for least-cost emissions reductions. The extent to which real-world EI policies resemble the theoretical ideal is discussed in next section.

[12] In the U.S. at least, most CAC policies adopted before 1990 did not even attempt to achieve cost-effective emission reductions. The most important economic considerations were distributional: to avoid shutting down plants, to give breaks to small facilities, and to treat existing facilities more leniently than new ones (Magat et al., 1986).

by reducing emissions, such programs provide continuing incentives for emissions-reducing innovation. By contrast, in a CAC system, the incentive to innovate is often offset by the enforcement risks associated with using a non-approved technology and the risk that a well-performing new technology will serve as the technology-based standard in a new round of CAC rulemaking--the so-called "regulatory ratchet."

Flexibility. Finally, EI instruments more easily accommodate change, whether of environmental standards, economic conditions, or abatement technologies. In a CAC system, the regulator must formulate and promulgate thousands of rules concerning different types of polluters. By contrast, in an EI system, firms retain control over facility-specific abatement decisions while the regulator simply sets fees or permit quantities to achieve an environmental quality standard. As a result, changes in response to new technologies and economic conditions are spontaneous and decentralized--the regulator need take no action at all. Changing the environmental quality standard is also relatively simple. In an emissions fee system, all that is required is to change the fee or fees. In a marketable permit system, all that is required is to change the quantity of permits (either by buy them or setting expiration dates on them).[13]

3.2.2 Monitoring, enforcement and instrument choice

As discussed above, because institutional and economic factors in developing countries limit regulators' ability to monitor and enforce environmental regulations, indirect instruments like technology standards may be more effective than direct instruments like emissions standards. But what do constraints on monitoring and enforcement imply about the choice between direct CAC instruments like emissions standards and direct EI instruments like emissions fees? Is one type of direct instrument more demanding of monitoring and enforcement capabilities than the other, and therefore less appropriate for some developing countries?

There is no simple answer to this question. Monitoring and enforcement requirements for EI and CAC direct instruments vary across polluting activities and pollutants. Moreover, there is very little empirical evidence on the monitoring costs of different types of instruments. Nevertheless, it is probably fair to say that in many contexts, monitoring and enforcement requirements for direct EI instruments are more demanding than for direct CAC instruments. To explain further, it is necessary to provide some brief background on monitoring.

Monitoring methods. The gold standard of source monitoring is the continuous emissions monitor (CEM), an electronic device permanently attached to the smoke stack that

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[13] Despite the often cited advantages of EI instruments, it is striking how infrequently they are encountered. At least two factors are relevant. First, the principal selling point of EI policies--improved economic efficiency--is something that decision-makers do not seem to care much about since there is rarely a constituency for it. Instead, there is usually tremendous pressure to assure that the benefits of the regulatory system continue to accrue to the current "winners" e.g., regulators, consulting firms, lawyers and certain segments of industry (Buchanan and Tullock, 1975). A second reason for the persistence of CAC policies is inertia--in most countries CAC policies are the status quo. This inertia might be easier to overcome if CAC approaches had obviously failed to improve air quality. But they have not.

produces a continuous record of emissions. Although the cost of CEMs have fallen over the last decade--and will undoubtedly continue to do so--they are still relatively expensive. For example, the average annual per plant cost of the CEMs used in the US Sulfur Dioxide Program (including capital, operation, and maintenance costs) has been estimated at $124,000 per unit (Schmalensee et al., 1998). While CEMs may eventually come into widespread use, today they are relatively rare, even in industrialized countries.[14] To date, monitoring of emissions has principally relied on a number of second-best methods and intermittent auditing.

Second-best monitoring methods include emissions factors, materials-balances, and indirect monitoring. Emissions factors are empirically estimated parameters that indicate of the quantity of a pollutant emitted per unit of output given a variety of equipment and input characteristics. For example, an emissions factor for a power plant might indicate the quantity of sulfur dioxide emitted per kilowatt hour by a plant using a certain type of coal, boiler, and electrostatic precipitator. The materials balance approach involves measuring quantities of a pollutant contained in both inputs and in outputs (including waste streams), and then using the difference between these measurements to deduce emissions. For example, to estimate sulfur dioxide emissions from a coal-burning power plant, one would calculate the difference between the quantity of sulfur contained in the coal the plant burns and the quantity of sulfur contained in the fly-ash it collects. Indirect monitoring involves measuring some indicator presumed to be correlated with emissions. Examples include the opacity of flue gas and the temperatures of boilers.

Each of these second-best methods has serious drawbacks. Emissions factors can be grossly inaccurate since for some pollutants such as nitrogen oxides and particulate matter, emissions depend critically on plant-specific operating conditions that emissions factors do not take into account. Not surprisingly, indirect indicators of emissions such as boiler temperatures can be grossly inaccurate for similar reasons. The materials balance approach is the most reliable second-best method. However it is only applicable to limited number of industrial activities where there is a relatively simple relationship between pollutant quantities in input, output, and waste streams. For example, it can be used to estimate sulfur dioxide and carbon dioxide emissions but not to estimate nitrogen oxides or particulate emissions.

Regardless of the methodology used, most regulatory authorities do not continuously monitor compliance. In many countries they generally settle for a two stage effort--early monitoring to ensure "initial compliance," and subsequent periodic monitoring to ensure "continuous compliance." The rationale for this approach is that once plants have installed the requisite pollution control equipment--which is easy for inspectors to verify--they are likely to comply with regulations since the capital cost of the pollution control equipment is sunk.[15]

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[14] Two of the principal exemplars of direct economic incentive programs in industrialized countries--the Sulfur Dioxide Program in the US and the nitrogen oxides fee system in Sweden--employ CEMs.

[15] This rationale is not always valid as operating costs for some pollution control equipment can be considerable. See Drayton (1978) and Russell et al. (1986).

In addition, regulators often adopt a fairly informal "voluntary compliance" process in which plants are encouraged to report violations and are given a chance to return to compliance without appreciable penalties. This process usually works reasonably well (Harrison, 1995; Russell et al., 1986). One of the reasons is that polluters tend to install emission control equipment that over-meets standards, often by a substantial margin (Arora and Cason, 1995).[16] The extensive use of voluntary compliance is no doubt testimony to firms' political clout, especially in local jurisdictions, but it also is to some degree impelled by the difficulty of monitoring.

Notwithstanding their weaknesses, second best and intermittent monitoring methods appear to have been fairly effective in ensuring compliance in the United States and Europe. Most authorities agree that emissions of many air and water pollutants have fallen in absolute terms in the U.S. and western Europe.[17]

EI instruments and second-best monitoring. There are at least four reasons to believe that second-best and intermittent monitoring methods are likely to be more problematic when used in the context of direct EI systems than when used in the context of direct CAC systems.[18] First, in a direct EI system, industry is likely to be far less tolerant of second-best monitoring than in a direct CAC system. In a fee or permit system, firms' out-of-pocket regulatory costs (fee and permit payments) are tied directly to emissions, while in an emissions standard system, such costs do not depend on their emissions as long as firms are deemed to be in compliance. Thus, political support for a direct EI system depends on critically on the perception that monitoring is fair and accurate (Drayton, 1978).

Second, second-best monitoring is likely to seriously degrade the static efficiency properties that are the principal selling points of emissions fee and marketable permit systems since these properties depend on plant managers and regulators having accurate information about emissions. For example, consider a fee system for nitrogen oxides emissions. As noted above, nitrogen oxides emissions (like particulate emissions) depend critically on idiosyncratic plant-specific operating conditions which can generally be adjusted at relatively low cost (Sterner and Hoglund, 1998). In a fee system where fee levels approximate average marginal

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[16] It is not clear why the phenomenon is so widespread, but among the possibilities are the existence of indivisibilities in abatement equipment and sources' need for a margin of error given variations in emissions. In addition, it is likely to be related to the combination of permit requirements that encourage bargaining between regulators and sources to allow some noncompliance in one regulatory arena in return for overcompliance in another. See Heyes and Rickman (forthcoming).

[17] Several types of evidence support this conclusion: (i) source-specific emission data, when it exists, shows that most sources are in compliance much of the time; (ii) data showing improvements, in ambient air quality; (iii) game-theoretic model which show that it can be in firms' self-interest to comply with regulations even when expected penalties for noncompliance are low or nonexistent. See: Harrington (1989); Harford (1993); Swierzbinski (1994); Heyes and Rickman (forthcoming).

[18] A specious argument to the contrary is that since EI instruments minimize aggregate abatement costs, industry will be more likely to comply, and therefore regulators' monitoring and enforcement costs will be lower. Malik (1992) shows that aggregate monitoring and enforcement costs can be higher for EI policies even though aggregate abatement costs are lower.

abatement costs and where emission are accurately measured, firms with relatively high emissions of nitrogen oxides due to sub-optimal operating conditions will have incentives adjust these conditions in order to lower fee payments. But in a fee system that uses emission factors, relatively high emissions due to suboptimal operating conditions will not be picked up, and firms will have no incentives to adjust these conditions. Similar problems would arise in a marketable permit system that uses second-best monitoring.

Third, in a marketable permit system, actual trading of permits that gives rise to static efficiency will not occur unless polluters perceive that monitoring and enforcement is credible and consistent: firms will only be willing to undertake abatement in order to sell permits if they believe they will receive full credit for emissions reductions, and they will only be willing to buy permits if they believe emissions limits implied by allowances will be strictly enforced. In short, the viability of the permit market depends on credible monitoring and enforcement. Second-best monitoring methods are less likely to provide the required level of credibility.

Finally, in a permit system, regulators would not be able to rely on firms over-controlling emissions to reduce the monitoring burden. In such a system, firms have a disincentive to overcontrol, since every unit of emissions abated can be sold to other firms.

Modified EI systems. It may be possible to mitigate some of the above problems with second-best monitoring by shifting some of the responsibilities for monitoring onto polluters. As discussed in Section 3, in some emissions fee systems, monitoring depends on emissions factors but firms are given the option of investing in first-best monitoring methods to prove their actual emissions are lower than the estimated emissions. In principle, this mechanism preserves the desirable incentives of direct EI systems, and reduces political opposition (Swierzbinski, 1994). However, it is not without drawbacks. Relatively clean but small or cash-poor firms may be forced to pay unwarranted fees based on emission factors because they lack the capital needed to invest in continuous emission monitoring. Of course, this creates disincentives for poor firms to be clean.

4. INTERNATIONAL EXPERIENCES WITH ECONOMIC INCENTIVES

This section reviews international experiences with EI regulatory instruments. Subsections 4.1 very briefly overviews the experiences of OECD countries. The remaining subsections present detailed case studies of the use of EI instruments in two industrialized countries--Sweden and the United States--and two developing countries--China and Poland.

4.1 The OECD

During the last two decades, industrialized countries have increasingly grafted EI policies onto existing CAC regimes. While European countries have typically opted for emissions fees and environmental taxes, the United States has favored permit trading. Today, although CAC policies still dominate, almost all industrialized countries have adopted some

EI instruments, and their popularity continues to grow.[19] Opschoor (1994) presents the results of a 1992 survey on the use of EI instruments among Organization for Economic Cooperation and Development (OECD) countries for the control of air pollution. The next three subsections summarize Opschoor's findings.[20]

4.1.1 Environmental taxes

OECD countries have instituted three types of taxes specifically aimed at abating air pollution: carbon taxes, sulfur taxes, and taxes on ozone depleting chemicals. Six countries levy carbon taxes: Denmark, Finland, Italy, the Netherlands, Norway, and Sweden. In three of the these countries--Denmark, Norway, and Sweden--the taxes were intended to have an incentive effect. All of these programs are still in their infancy and it is too early to tell if they have succeeded in cutting emissions. However, as discussed below, Swedish carbon taxes are relatively high and some impacts are evident. Three OECD countries--Norway, Sweden, and Finland--tax the sulfur content of fuels. The Swedish tax seems to have reduced sulfur emissions. Finally, three OECD countries--Denmark, Australia, and the USA--tax ozone depleting chemicals. Both the Danish and American taxes seem to have had a significant impact.

4.1.2 Emissions fees

Six OECD countries have emissions fee programs specifically geared toward air pollution: Canada (general air pollution), France (acidifying emissions), Japan (sulfur dioxide), Portugal (sulfur dioxide and nitrogen oxides), Sweden (nitrogen oxides), and the United States (criteria air pollutants). Though the programs in Canada, Japan, Sweden, and the United States were intended to have an impact on emissions, only the Swedish program has clearly had such an effect. It is discussed in detail below.

4.1.3 Tradable permits

Three OECD countries have tradable permit programs aimed at air quality control: the USA, Canada, and Germany (the only other tradable permit programs of any type are in Sweden and Australia). The American programs are the oldest and have received the most attention. The three principal American permit trading programs are the Emissions Trading Program, the Sulfur Dioxide Program, and the Ozone Program. The first two US programs are discussed in more detail below.

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[19] In eight countries studied by Opschoor (1994) the number of economic incentive policies grew by 50% between 1987 and 1994.

[20] For a more detailed analysis of the OECD experience, see Vos et al. (1994). For other case studies see Anderson and Lohof (1997) and Huber et al. (1996).

4.2 Sweden

As noted above, Sweden's tax and fee programs stand out among EI programs in OECD countries as having clearly changed firm behavior. Sweden has three EI air pollution control policies: a carbon tax, a sulfur tax, and a nitrogen oxides emissions fee. The two taxes are levied on the carbon and sulfur contents of various fossil fuels and therefore, in our lexicon, are "taxes" not "fees."

4.2.1 Sweden's carbon tax[21]

Initiated in January 1991, Sweden's carbon tax grew out of a drastic reform of the national tax system that essentially consisted of cutting income taxes and energy taxes and raising or initiating value added taxes, sales taxes, and carbon taxes to offset the lost revenue. Although energy taxes were halved, the imposition of new carbon taxes more than compensated for this reduction. As a result, coal prices rose by 80 percent, oil prices by 20 percent, and natural gas prices by 100 percent. However, the 1991 tax regime was short-lived. Extensive lobbying by industry led to a second round of tax reforms in 1993 which differentiated taxes across sectors.[22] The net effect of these reforms was to reduce energy prices relative to the 1991 level for industry, and to raise energy prices relative to the 1991 level for other sectors. A third round of tax reforms in 1997 doubled carbon taxes paid by industry. Revenue from carbon taxes is not earmarked. The administration of carbon taxes has been handled by existing tax authorities.

It is difficult to evaluate the impact of the carbon tax on emissions since the imposition of the tax coincided with other fiscal reforms and since the tax has changed so frequently. Nevertheless, Bohlin (1998) finds that carbon taxes have significantly reduced emissions in some non-industrial sectors such as district residential heating. Revenue from carbon taxes has been substantial. In 1995, it generated US $1.6 billion, approximately one percent of Sweden's GDP. Administrative costs attributed to carbon taxes are estimated at roughly five percent of total revenue.

4.2.2 Sweden's sulfur tax

Sweden's sulfur tax, also initiated in January 1991 as part of the national tax reform, is a tax on the sulfur content of coal, peat, and oil. The tax rate, US $3,900 per ton, was based on a calculation of the average marginal cost of abating sulfur emissions and is considerably higher than rates used in other countries (Lovgren, 1994). Fuels that are used for purposes other than energy (e.g., petrochemicals) and fuels containing less than one percent sulfur by weight are exempted from the tax. To reward firms that have installed end-of-pipe treatment

[21] This section is based on Bohlin (1998).

[22] The new law set carbon taxes for industry at US $27 per ton for coal, $31 per cubic meter for oil, and $23 per cubic meter for natural gas, and set carbon taxes for other sectors at $107, $123, and $91 respectively.

systems, the tax is refunded when emissions are controlled by scrubbers.[23] To prevent cheating, firms that claim refunds are subjected to continuous emissions monitoring. The sulfur tax supplements a pre-existing CAC regime involving permitting and emissions standards. Revenue from the sulfur tax is not earmarked. Administration of the tax is handled by the same authorities that handle energy and carbon taxes.

The impact of the sulfur tax is difficult to evaluate because of parallel CAC regulations, simultaneous tax reforms, and structural changes in the Swedish economy. Nevertheless, there are strong indications that the tax has significantly reduced sulfur emissions. Aggregate sulfur emissions decreased by 25 percent in the first year the tax was administered. More tellingly, after the imposition of the tax in 1991 but prior to the tightening of CAC emissions standards in 1993, the average sulfur content of heavy fuel oil fell from 0.65 percent to 0.40 percent (Lovgren, 1994). According to OECD (1997), annual sulfur emissions have fallen by about 6,000 tons per year as a direct result of the tax. The sulfur tax generated around US $39 million in 1992 (Lovgren, 1994). Revenue has been less than expected because the tax has reduced the demand for high-sulfur fuels (OECD, 1997). Administrative costs are estimated to have been less than one percent of total revenues (OECD, 1997).

4.2.3 Sweden's nitrogen oxides fee

In 1992, Sweden imposed a fee on emissions of nitrogen oxides from electricity and heat generating plants. The fee, US $5,200 per ton, together with a tightening of CAC regulations, was designed to reduce emissions by 30 percent over three years. Continuous monitoring is used to measure emissions for most plants.[24] CEMs were deemed necessary because, as noted above, emissions of nitrogen oxides depend on the plant-specific operating conditions and as a result, second-best monitoring methods can be grossly inaccurate. The annual cost of operating and maintaining CEMs has been estimated at US $39,000 per plant or US $520 per ton of nitrogen oxides abated (Lovgren, 1994). Given the magnitude of these costs, small sources have been exempted from the fee.[25] To avoid giving small plants a competitive advantage, the revenue from the fee is refunded to the payees. To avoid dampening the incentive effect of the fee, revenue is refunded in proportion to the amount of energy produced. As a result, plants with high emissions per unit output are net providers of funds--that is, their fee payments exceed their refunds--while plants with low emissions per unit output are net recipients of funds. Unfortunately, entire sectors which for technological

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[23] Over half of firms that pay sulfur taxes receive such refunds (OECD, 1997)

[24] Plants have the option of using emission factors instead of CEMs, but emissions factors are set at levels intended to discourage this option.

[25] Originally the program exempted plants with less than 10 MW installed capacity or 50GW annual output, so that it covered approximately 200 plants that accounted for 40 percent of the nitrogen oxides emissions from the energy sector. In 1997, the criteria for participation was changed to exempt only plants producing less than 40GWh per year. This change brought 600 more plants into the program (OECD, 1997; Sterner and Hoglund, 1998)

reasons tend to have higher emissions per unit output (e.g., metal production) have tended to become net providers of funds while other sectors with lower emissions per unit output (e.g., waste incineration) have become net recipients (OECD, 1997).

The nitrogen oxides fee has clearly had a strong impact on emissions. Total emissions from monitored plants fell by 40 percent in the first two years of the program.[26] For most plants, there were no changes in CAC regulations during this time, so most of the reduction can be attributed to the emission fee (Lovgren, 1994, Sterner and Hoglund, 1998).[27] The fee program generates approximately $80 million per year which is refunded to payees. The annual cost of administering the program has been estimated at approximately 0.2 to 0.3 percent of the annual revenues (Sterner and Hoglund, 1998).

4.3 The United States

This section discusses two of the most important permit trading programs in the United States, the Emission Trading Program and the Sulfur Dioxide Program.

4.3.1 The Emissions Trading Program[28]

Background. The Emissions Trading Program (ETP), the oldest US air permit program, was grafted onto a complex CAC regime for industrial air pollution control.[29] The ETP grew out of frustration with existing CAC regulation that, if interpreted strictly, would have prohibited the building of new sources in areas not in attainment with ambient air quality standards. To accommodate economic growth under the CAC regime, the EPA established a program of "offsets" whereby new sources are allowed to locate in non-attainment zones if they are able to secure sufficient "emissions reductions credits" from existing firms. Emissions reductions credits evolved into the hard currency of a tradable permit market. By 1986, the EPA had formalized rules to allow three other kinds of transactions: "netting", "bubbles", and "banking." Netting allows old sources wanting build new facilities to avoid strict new source regulations by, among other things, applying emissions reductions credits earned in old facilities to new facilities. Banking, instituted in 1979, allows firms to store emissions reductions credits for subsequent use. Bubbles, also instituted in 1979, allow two or more sources to be treated as one emissions source. The bubble provisions were supposed to encourage both internal and external trading of emissions reductions credits and, according

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[26] Emissions reductions have been achieved by improving combustion efficiency, installing selective noncatalytic reduction systems, and flue gas cleaning (scrubbing).

[27] There is some evidence that, despite the supposed neutrality of the refund mechanism, plants that are net providers of fee refunds have undertaken more abatement than those that are net recipients (OECD, 1997).

[28] This section is based on Atkinson and Tietenberg (1991); Foster and Hahn (1995); Hahn and Hester (1989); Kete (1994); and Tietenberg (1990a and 1990b).

[29] The CAC regime has three main components: national ambient air quality standards for six different pollutants, state implementation plans that specify emissions standards and other controls necessary to meet the ambient air quality standards, and best available technology standards for a variety of sources.

to Atkinson and Tietenberg (1991), were supposed to be the centerpiece of the ETP since they most closely resembled theoretical models of emissions trading.

Impacts. In general, the impacts of the ETP have been significant but much more limited than its proponents had hoped. The number of trades has been far smaller than expected. By 1986 there had been about 150 bubble transactions, 2,000 offset transactions, 5,000-12,000 netting transactions, and only about 100 banking transactions (Hahn and Hester, 1989). Moreover, most of the bubble trades were internal, that is, between firms owned by the same parent corporation. Most analysts agree that the environmental impacts of the ETP have been negligible, a result that is not surprising given that cost effective compliance with existing standards was the primary motivation for the program. Although the program has resulted in significant cost savings--on the order of $10 billion as of 1986 according to Hahn and Hester (1989)--the savings have been lower than expected. Moreover, the bulk of the savings have come from netting transactions which more closely resemble regulatory relief than regulatory reform (Tietenberg, 1990a).

Lessons. Why has the ETP not performed as well as advertised? Most analysts agree that five factors are to blame. First, expectations were unrealistically high. Second, existing CAC regulations have restricted trading. For example, current rules require that no individual trade can result in increased emissions from a participating firm. This restriction rules out multilateral trades that increase emissions of some firms but reduce overall emissions. As a result, cost savings are lower. Also, ETP rules severely restrict new firms' ability to trade even though they have the greatest incentives to do so since they are subject to the strictest standards. Third, transactions costs for firms involved in permit trading are high, due in no small part to the regulatory administrative requirements. Foster and Hahn (1995) found that in the Los Angeles basin, which has the greatest level of trading in the ETP, transactions costs frequently exceed the market value of the emissions reductions credits. Fourth, firms have limited information about the market for permits. And finally, firms are often reluctant to participate in the permit market because of uncertainty about future regulation.

4.3.2 The Sulfur Dioxide Program

Background. While the Emissions Trading Program is the oldest US air permit trading program, the Sulfur Dioxide Program has arguably been the most successful. A centerpiece of the 1990 Clean Air Act Amendments, the Sulfur Dioxide Program was designed to reduce sulfur dioxide emissions from electric power plants to half of their 1980 levels. It is being implemented in phases. In Phase I which began in January 1995 and runs through December 1999, an interim cap on aggregate emissions of 5.7 million tons per year was established for 110 of the most polluting power plants.[30] In Phase II which will begin in

[30] These 110 plants are mostly comprised of coal-fired plants east of the Mississippi river. In addition, units which are not covered by the program have the option of volunteering to participate. 263 units at 110 plants were originally covered in Phase I but 182 more units "volunteered' to participate (Schmalensee et al., 1998).

January 2000, a cap of 8.95 million tons per year will be established for all generating units in the continental United States larger than 25 MW as well as all new units of any size. To enforce the aggregate emissions caps in each phase, utilities are allocated a certain number of allowances, each of which entitles them to emit one ton of sulfur. Allocations are based on historical levels of fuel consumption. Utilities are audited at the end of each year to ensure that their emissions have not exceeded their allowances. Exceedances are severely sanctioned.[31] To reduce the cost of meeting the emission caps, utilities are permitted to trade allowances with any party anywhere in the continental United States or to "bank" them, i.e., carry them forward into the next year. Unlike the Emissions Trading Program, there are no restrictions on trading on the basis of environmental or economic benefits.[32] To ensure that new plants are able to obtain allowances (and to improve information about the market for allowances), the EPA auctions off between two and three percent of the total allocation of allowances each year.

Impacts. The Sulfur Dioxide Program has had strong environmental and economic impacts. Regarding the former, the program has achieved its ambitious emissions reductions targets. As of 1997, sulfur dioxide emissions from power plants were more than four million tons below their 1980 levels (Tietenberg, 1999).[33] Regarding economic impacts, the Sulfur Dioxide Program has entailed considerable costs for firms--emissions reductions have cost about $200 per ton (Schmalensee et al., 1998)--but has also delivered considerable cost savings. According to Stavins (1998a) compliance cost savings from the program are on the order of $1 billion annually compared to CAC regulatory alternatives.[34] There has been some debate about the extent to which cost savings are due to actual trading of allowances as opposed to the substitution of emissions standards for technology standards. According to Schmalensee et al. (1998), actual trading has reduced costs by 25 to 34 percent compared to an allowance regime with no trading.

Lessons. Recent evaluations of the Sulfur Dioxide Program points to a number of factors that have contributed to its effectiveness (Stavins, 1998b, Tietenberg, 1999). First, there was a conscious effort to minimize transactions costs by, among other things, allowing trades without prior approval. Second, the use of CEMs has helped build market confidence. Third, the program has not limited the means by which firms can meet emissions standards.

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31 Sanctions include a forfeit of $2000 and one allowances per ton of exceedance.

32 However, power plants are required to meet all local and national pollution control standards regardless of the number of allowances they hold.

33 In fact, in 1995 and 1996, emissions were significantly below allowed levels. The reasons included expectations that allowance prices would rise in the future, and unanticipated declines in the prices of low-sulfur coal. Switching to low- and lower-sulfur fuels--a compliance strategy that would have been strongly discouraged under a technology standard system--has accounted for over half of the total reductions in sulfur emissions (Schmalensee et al., 1998).

34 Though large in absolute magnitude, these savings only constitute about 0.5 percent of the total annual costs of electricity generation (OECD, 1997).

Fourth, in contrast to the Emissions Trading Program which has been overlaid on existing regulation, the Sulfur Dioxide Program was designed to substitute for existing regulation. Relatedly, the baseline for the program was clearly defined as an aggregate level of emissions, whereas in Emissions Trading Program, it was defined as emissions reduction above and beyond complex existing legal requirements. Finally, the program was oriented toward emissions reductions as well as cost savings and was therefore successful in garnering political support. But, despite some clear successes, several aspects of Sulfur Dioxide Program raise concerns about the appropriateness of permit trading programs for developing countries.

Administrative and monitoring costs. Incremental administrative costs associated with the Sulfur Dioxide Program (i.e., administrative costs incurred by both regulators and firms above and beyond costs incurred in conventional CAC system) are significant. They include the costs of: keeping track of all trades via the Allowance Tracking System; holding yearly auctions; buying permits; trading permits; and monitoring firms' emissions to ensure that they do not exceed permitted levels. Though *ex-post* information on all administrative costs (except monitoring costs) has yet to be tabulated, rough *ex ante* estimates for 1993-2000 are available. Total incremental administrative costs (leaving aside monitoring costs, at least some of which would be paid under a CAC system) were estimated at between $270 and $481 million of which $50.6 to $57.1 million would be paid by the regulator and the balance by firms. The bulk the costs incurred by firms consist of transactions costs (ICF, 1992).[35]

Importantly, monitoring costs associated with the Sulfur Dioxide Program are also significant. All plants that participate in the program are required to install CEMs, flow monitors and opacity monitors. Among the reasons for this provision were fears that if an emissions factor approach was used, utilities would receive credit for installing scrubbers but might not operate them, and also a recognition that the viability of the allowance market would depend on the credibility of enforcement. The average annual cost of continuous emissions monitoring (including operating and annualized capital costs) is approximately $124,000 per generating unit (Schmalensee et al., 1998, 55).[36]

Volume of trading. In the first years of the program, the volume of allowances traded was lower than expected. Between the time the program was inaugurated in January 1990 and March 1993, only 130,00 allowances were traded, and the majority of these were between units owned by the same firm. However, by March 1997, the volume of trades had increased 20 fold (Schmalensee et al., 1998). Low trading volumes in the first years of the program were blamed on a variety of institutional rigidities including: rules passed by local utility

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[35] The cost of the Allowance Tracking System, including transactions costs incurred by firms, was estimated at roughly $304 million, the cost of holding auctions at $4.5 million, the cost of buying permits at $68 million. Although there are considerable transactions costs associated with the Sulfur Dioxide Program, Tietenberg (1999) argues that they have been minimal compared to precursor programs like the Emissions Trading Program. Economic analysis of the program supports this view (Montero, 1997).

[36] EPA estimated that the fixed capital costs of continuous monitoring for each firm would be approximately $302,200 while the operation and maintenance costs would be $78,700 per year (ICF, 1992).

regulators that discouraged trading and encouraged the continued use of local coal together with scrubbers; uncertainty about whether allowances would be limited by other pollution control legislation or affected by coming utility market deregulation; defects in the auction mechanism; and transactions costs associated with trading (Klaassen, 1996). However some analysts have argued that low trading volumes resulted not so much from institutional rigidities as from low demand for allowances due to unanticipated cost-saving innovations in the markets for scrubbers and low-sulfur coal transportation (Burtraw, 1996).

4.4 China

Although China's market reforms have sparked two decades of extraordinary economic growth, the environmental cost has been significant. Particulate and sulfur dioxide levels in many Chinese cities are two to five times higher than World Health Organization standards (World Bank, 1997). The principal contributors to this problem are a multitude of aging state-owned dirty industries and a heavy dependence on coal for primary energy.[37] In this section, we describe China's efforts to use emissions fees to control air pollution.

Key features. China's first comprehensive Environmental Protection Law, passed rather belatedly in 1979, established a mixed regulatory system based on both emissions fees and emissions standards.[38] The original intent of the fee system was to enforce compliance with emissions standards. Therefore, polluters are only required to pay a fee only on those emissions that exceed emissions standards. To encourage eventual compliance with emissions standard, firms that violate standards for three consecutive years are assessed a fee increase of five percent per year. To create incentives for newly built plants to install pollution control equipment, all fees for plants built after 1979 (the year the fee system was initiated) are doubled, and fee increases for non-compliant plants increase by 100 percent per year (instead of 5 percent per year). Fees are set by the central government but provincial and local governments may raise them. As a result, there is substantial variation in fees. Fees tend to be higher in more developed provinces and, within provinces, for old and state-owned sources (Wang and Wheeler, 1996; Dasgupta, Huq, and Wheeler, 1997). Fees are charged on 20 different air pollutants. However, when more than one pollutant is above the permissible level, enterprises are only required to pay fees for the "worst case pollutant," i.e., the one pollutant that involves the largest fee payment. The national floor on fee rates is roughly US $280 to $700 per ton for particulate matter (depending on the source) and roughly US $280 per ton for most other common airborne pollutants including sulfur dioxide, nitrogen oxides and carbon monoxide (Yang et al., 1997).

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[37] Industrial sources account for roughly three-quarters of sulfur dioxide and particulate emissions (Wang and Wheeler, 1996)

[38] The fee system has evolved through three stages: trial implementation in selected cities (1979-81); nationwide implementation (1982-87); and reform and refinement (1999-present) (Yang et al., 1997).

Monitoring and enforcement. The monitoring needed to assess fees is based on self-reporting, periodic auditing, and a crude monitoring technology. Each enterprise is required to monitor emissions concentrations daily and to report monitoring data to the local Environmental Protection Bureau (EPB). To check the accuracy of firms' reports EPBs compare them with past reports and with reports for similar firms and also make unannounced spot checks. Actual monitoring of emissions by both firms and EPBs is based on visual inspection of the opacity of flue gases. Inspectors rank opacity on a scale of one to five using the a set of gray-scale cards commonly known as the Ringelmann scale.[39] This opacity measure is combined with estimates of emissions volumes to assess fees. Thus, even though fees are ostensibly differentiated across 20 pollutants, actual fees are based on a single crude measure of concentration (Yang et al., 1997). Fines may be imposed for false reporting, and for interfering with inspections. However, both the probability of getting caught for under-reporting and the penalty for doing so are quite low. In most cases, firms caught under-reporting are simply required to provide an explanation (Yun, 1997).

Revenue. Approximately $3 billion in fee revenue was collected between 1979 and 1995. Twenty-nine percent of this amount--an average of roughly $54 million per year--was paid by air polluters (Yang et al, 1997). Fee revenue is earmarked for investments in pollution control. Law dictates that 80 percent of fee revenue be used to subsidize pollution control investments by the enterprises that pay the fees and the remaining 20 percent (along with all fines) be used to fund the operations of EPBs.[40] Of the funds provided to enterprises, 92 percent are used for plant-specific projects and the remainder for collective treatment facilities (Wang, Zhang, and Wu, 1997). Revenue from emissions fees accounts for only about 6 percent of total capital investment in pollution control.[41]

Impact. There is some disagreement in the literature about the impact of the fee system on emissions. On one hand, proponents of the fee system point to evidence that emissions per unit of output fell precipitously after the establishment of the fee system. In addition, some econometric studies using province-level data show that there is a statistically significant negative correlation between "effective fee levels" (the actual amount of fees paid

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[39] The Ringelmann scale was developed in France in late 1800's as a means of measuring the efficiency of coal-fired boilers. Darker smoke meant poorer efficiency. The Ringelmann scale came into widespread use in Western countries as a measure of air pollution in the early part of the century and was used in some contexts by the EPA until 1974 (Eastern Technical Associates, 1999).

[40] However, historically, EPBs have absorbed just over 30 percent of the fee revenues (Wang, Zhang, and Wu, 1997).

[41] The bulk of investment in pollution control comes from: new construction (36 percent from 1991-93); reconstruction of existing plants (9 percent); and from an urban services fee paid by enterprises (39 percent) (Wang, Zhang, and Wu, 1997).

per unit of emissions) and emissions.[42] But neither argument is altogether convincing. Regarding the former, the fee system was established at the same time as emissions standards and it is difficult to disentangle the impacts of the two instruments. Regarding the econometric studies, as discussed in the next section, there are reasons to believe that "effective fees" may be a poor indication of regulatory stringency.

Most analysts have argued that is it unlikely that emissions fees by themselves have been responsible for improved environmental performance. They point out that fees are well below marginal abatement costs for most firms, and therefore provide limited abatement incentives (Florig et al., 1995; Yun, 1997; Yang et al., 1997).[43] More damning, an extensive firm-level study of the incentive effects of emissions fees (including an econometric study) found that incentives created by fees were limited (Yun, 1997). The study found that emissions fees are so low compared to marginal abatement costs that in many cases, even polluters with abatement equipment already installed prefer to pay fees rather than pay to operate their abatement equipment.

Problems. As noted above, one of the main problems with China's fee system is that the fees are set too low, in part because they have been eroded by inflation. Though fees have been in use for 20 years, they have only been increased one time (in 1991).

A second problem is that, the level of fees aside, the structure of the fees curtails incentives to abate. Because firms only pay fees on emissions that exceed the legal standard, they have no incentives to reduce emissions below the standard. In addition, because firms only pay fees on the "worst case pollutant" they have no incentives to abate emissions of other pollutants.

As discussed at length by Yun (1997) and Wang et al. (1997), a third critical problem is that fees actually create strong perverse incentives for enterprises and EPBs to perpetuate non-compliance. For enterprises, the problem arises for two reasons. First, enterprises are allowed to count most fee payments as production costs.[44] Thus, fees lower enterprises' tax liabilities, a very important consideration given that tax rates on profits are on the order of 33 percent. Second, enterprises are usually able to recoup the lion's share of the fees they pay. As noted above, 80 percent of fees are returned to enterprises, ostensibly for investment in pollution control. But local EPBs simply do not have the resources or political will to closely monitor how enterprises use funds. Therefore, in many cases, fee revenue ends up being used for non-environmental purposes. According to Yun (1997), many enterprises view the fee system as 'depositing money in a bank' and often actually overpay fees in order to lower their

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[42] Dasgupta, Wang and Wheeler (1997) found that each one percent increase in effective air emissions fees leads to a decrease of approximately 0.3 percent in sulfur dioxide intensity, and a 0.4 to 0.8 percent decrease in particulate matter.

[43] However, Dasgupta, Huq and Wheeler (1996) found the opposite. They argue that early in the regulatory process, industrial emissions intensity is highly responsive to changes in the price of pollution because marginal costs are often quite low.

[44] Enterprise are allowed to count as production costs all fees except those on emissions that are above standard for the third consecutive year, fees paid by factories built after 1979, and fines for underreporting and for late payment (Yun, 1997).

tax liabilities.[45] Thus, the fee system actually creates incentives for enterprises to be out of compliance so that they can pay fees. Aware of the problem, in 1988 the state mandated that funds be returned to enterprises in the form of loans rather than grants. However, this rule has not been enforced. EPBs are able to subvert it by exempting enterprises from repaying loans or by simply not enforcing repayment.

The fee system creates parallel perverse incentives for EPBs to perpetuate non-compliance. EPBs depend heavily on fee revenues for financial support. For example, Yun (1997) found that EPBs derived 70 to 93 percent of their operating revenues from fee revenues. Hence EPBs, like firms, have strong incentives to maintain a steady flow of fee payments.

A related problem is that, leaving aside incentives for non-compliance, enforcement is often weak. Unprofitable enterprises are usually able to escape paying fees by appealing to local authorities. Enforcement is especially weak for small-scale enterprises, most notably China's eight million township and village industrial enterprises that account for about one half of total industrial output and that are the major source of pollution in rural areas. Many of these enterprises fall outside of the formal regulatory system altogether (World Bank, 1997).[46]

4.5 Poland

In Poland, as in most transitional countries, decades of central planning have left a legacy of serious environmental problems. Polish emissions per unit of GDP of sulfur dioxide, nitrogen oxides, and particulate matter are two to eight times average OECD levels. Air pollution is primarily caused by emissions from large stationary sources. Power plants are probably the worst polluters. Structural (as opposed to regulatory) factors that contribute to high levels of emissions include: the dominance of relatively dirty heavy industries such as metallurgy, chemical production, and mining; extremely high levels of energy intensity; and dependence on relatively low-quality indigenous coal and lignite (Adamson et al., 1996). In this section, we describe Poland's efforts to use emissions fees to control air pollution.

Key features. Poland's airborne emissions fee system was established in 1980. A decade later, in concert with other Eastern Block countries, Poland enacted new legislation which revamped the emissions fee system and also tightened complementary CAC regulations.[47] Like most of its neighbors, Poland has a hybrid fee/standard instrument: a

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[45] Because enterprises use fees to lower their tax liabilities, local tax collection bureaus have actually discouraged the raising of fees. In some extreme cases the they have even set a limit on the total amount of fees that can be assessed. (Yun, 1997).

[46] NEPA, China's National Environmental Protection Agency, plans a number of reforms to deal with some of the problems discussed above (NEPA, 1997). These include: extending fees to cover all emission instead of just those that exceed the standard; charging fees on all pollutants instead of one 'worst case' pollutant; raising fees above marginal abatement costs and indexing them to inflation; and putting stronger conditions on the use of fee revenues by firms. China has begun experimenting with "local environmental funds" to improve the allocation of fee revenues. These are semi-private investment companies that are administratively separate from EPBs and which therefore have the potential to be more objective about how funds are disbursed (Wang, Zhang, and Wu, 1997).

[47] For a review of pollution fee systems in Central and Eastern Europe, see Vincent and Farrow (1997).

"normal fee" is paid on all emissions below an emissions standard and a "penalty fee" up to ten times higher than the normal fee is paid on all emissions above the standard. In Poland, emissions standards are source-specific and are set by a permitting process. Permit applicants are required to submit an extensive environmental impact analysis and to have their applications reviewed by an independent government-approved expert (Anderson and Fiedor, 1997).

Fee rates are determined by Poland's national environmental ministry for 62 specific air pollutants and seven different types of evaporative air emissions. The relative levels of the fees are based on their presumed potential to cause environmental damage. The absolute levels of the fees are supposed to be determined by a number of considerations including ambient air quality guidelines and marginal abatement costs, but in practice are determined by political acceptability and revenue requirements (Anderson and Fiedor, 1997).

Fee rates are revised annually. Until the 1990, fees were quite low. However, since that time they have increased dramatically, some by a factor of 20. In 1995, the highest "normal" fee for air emissions was US $54,000 per ton for various hazardous air pollutants. That same year, the "normal" fees were $83 per ton for sulfur dioxide and nitrogen oxides and $44 per ton for particulate matter (Anderson and Fiedor, 1997).

Monitoring of emissions relies on self-reporting and emissions factors. Local regulators are supposed to verify self-reports. Firms are allowed to defer penalty fee payments for three to five years. Deferred payments can be waived if the firm is in compliance by the end of the deferral period.

Revenue. In 1994, regulators levied $246 in airborne emissions fees (both "normal" and "penalty" fees) of which 90 percent ($221 million) was actually collected. Fee revenue is distributed to a national environmental fund (36 percent), 49 regional environmental funds (54 percent), and 2,400 local environmental funds (10 percent).[48] The environmental funds disburse the revenue to polluters (in the form of subsidized credit and grants) and to regulators. Revenue is also used for public-sector pollution control infrastructure such as water treatment and coal washing facilities. Environmental funds account for nearly half of annual capital costs of all investment in Poland (Anderson and Fiedor, 1997).

Impacts. Until the fee system was revamped in 1990, it had no discernable effect on either pollution levels or revenue. Since then fees have clearly had an impact on revenue. Their impact on emissions is less clear, in part because the imposition of fees has coincided with the tightening of CAC standards and with drastic economic restructuring. There is little doubt that "normal" fees have been set too low to have had much impact on abatement. However, according to Anderson and Fiedor (1997), "penalty" fees do provide significant incentives to abate. Bates et al. (1994), appear to concur, arguing that in the early 1990s, emission fees stimulated emissions reductions, at least for particulates and sulfur dioxide.

[48] For nitrogen oxides, 90 percent of revenue goes to the national fund since nitrogen oxides pollution is considered to be a national and transborder problem.

Problems. Aside from the level of the fees, there are a number of important barriers to the effectiveness of Poland's fee system, all of which echo problems with China's system. First, monitoring and enforcement is quite weak. Regulators have a limited ability to verify self-reported emissions for the usual reasons: they are undermanned, underfinanced, and lack public support. In addition, their ability to levy and collect fees is limited by political concerns: many of the worst polluters are large politically powerful enterprises. Although enterprises have been closed down periodically for environmental reasons, most quickly resume operations. Limitations on monitoring and enforcement are apparent from the fact that in 1992, fully 40 percent of all registered air polluters (and 55 percent of water polluters) were operating without valid permits needed to calculate fees and were therefore effectively exempted from the fee system. Also illustrative is the fact that in 1992, only 20 percent of "penalty" fees levied were actually collected (Anderson and Zylicz, 1996). Second, notwithstanding extensive privatization, many large enterprises are state-owned and still operate under soft budget constraints. Since such firms can effectively count emissions fees as operating expenses, fees do not create strong incentives to abate. Many state-owned firms receive waivers and subsidies or simply violate regulations with impunity (Bates et al., 1994). Even private firms are allowed to count "normal" fee payments as production costs in order to lower their tax liabilities, a concession that erodes the incentive effect of fees (Anderson and Fiedor, 1997). Third, the use of fee revenue to subsidize regulatory activity and pollution control investment creates perverse incentives to maintain the flow of fee revenue (Bates et al., 1994). Finally, coal is widely used for residential heating. As a result, thousands of small fixed-point sources fall outside of the fee system (Adamson et al., 1996).[49]

5. CONCLUSION

The four case studies presented above provide a number of important lessons about the use of EI instruments in developing countries. In what follows, we consider the implications of these case studies for each of the three types of EI instruments discussed in this paper: emissions fees, tradable permits, and environmental taxes.

5.1 Emissions Fees

The effectiveness of emissions fees can be judged by their impact on the environment, revenue generation, and on regulatory administrative costs. In terms of the first criteria, Sweden's nitrogen oxides fee has been the most successful, having clearly reduced emissions. Opinions vary widely on whether and to what extent Chinese and Polish emissions fees have had a significant environmental impact. Emissions fees in all three countries have generated

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[49] Given the existence of these barriers, it is not feasible in the foreseeable future to rely exclusively on emission fees to achieve air quality objectives. Bates et al. (1994) recommend pursuing a 'mixed' regulatory system, including a ban on the use of dirty fuels by small sources in urban areas and a combination of increased emissions fees, tradable permits, and CAC regulation for large polluters.

significant revenue: roughly $80 million per year in Sweden from nitrogen oxides fees; $54 million per year in China from fees on a variety of air pollutants; and $221 million per year in Poland from fees on a variety of air pollutants. Information on administrative costs is incomplete. We know only that in the case of Swedish nitrogen oxides fees, administrative costs have been estimated at 0.2 to 0.3 percent of revenues, a figure that analysts consider quite low. Thus, while emissions fees appear to be an effective revenue generating mechanism in all three countries, they have only clearly had a significant environmental impact in Sweden.

Why have emissions fees in China and Poland not had a bigger environmental impact? Part of the answer has to do with the design of the fee system, in particular, the level and structure of fees and the means by which earmarked fee revenue is refunded. Regarding the level of fees, some fees in both countries are set well below marginal abatement costs so that firms generally prefer paying fees to investing in abatement. For example, nitrogen oxides fees in China and Poland are $280 per ton and $83 per ton respectively. By contrast, Sweden's nitrogen oxides fee--set to average marginal abatement costs--is US $5,200 per ton. In Southern California's RECLAIM program, a regional nitrogen oxides permit trading system, the 1997 price of permits (for discharge in 1999) was $1,800 per ton (South Coast Air Quality Management District, 1998). Thus, Swedish fees and the Californian permit prices exceed Polish fees by factors of 60 and 20, respectively. Purchasing power parity differences and the fact that marginal abatement costs in China and Poland are no doubt lower than in Sweden probably account for some of the differences, but certainly not all of them.[50]

Regarding the structure of fees, the two-tiered design of China's and Poland's emissions fees dampens incentives to abate. In China, fees below the emissions standard are zero and in Poland they are as little as one-tenth of the "penalty" fee rate. As a result, firms have little or no incentive to abate as long as they comply with the emissions standard.

Regarding the use of earmarked revenue, in both China and Poland, the mechanism for disbursing the fee revenue offsets (to some degree) the incentives to abate created by the fees. In both countries, the use of fee revenues to finance regulatory activity creates perverse incentives on the part of regulators to ensure that polluters remain out of compliance and continue to pay emissions fees. Worse, in China, tax regulations that permit polluters to count fee payments as production costs, coupled with minimal control over how polluters use refunds, create parallel perverse invectives on the part of polluters to remain out of compliance to continue to pay emissions fees.

But while design issues are partly responsible for the poor performance of emissions fees in China and Poland, weak monitoring and enforcement are also to blame. In both countries, environmental protection organizations are undermanned and underfinanced, and

[50] However, Chinese and Polish sulfur dioxide fees, also $280 per ton and $83 per ton respectively, are on par with the price of allowances in the U.S. sulfur trading program, which have varied between $100 to $200. However, US permit prices may reflect a glut in the permit market due to overinvestment in scrubbing capacity (Schmalensee et al., 1998).

many firms--especially large state-owned enterprises with soft budget constraints--are politically insulated from regulatory pressure. The importance of constraints on monitoring is starkly illustrated by the fact that in China, notwithstanding a system in which fees are ostensibly assessed on emissions of 20 different air pollutants, regulators commonly rely on a single out-dated and extremely crude measure of opacity to assess compliance and calculate charges, and in Poland 40 percent of registered polluters operate without permits needed to calculate fee rates.

A final factor that has contributed to the ineffectiveness of Chinese and Polish fees has been the fact that in both countries many sources--mainly small firms and non-industrial sources--fall outside of the fee system.

What lessons can be distilled from our case studies of emissions fees? First, fees need to be set high enough to have an impact on emissions and also need to be indexed to inflation. Unfortunately, relatively high fees present a number of difficulties in a developing country setting. Politically, it may be difficult to raise fees to the requisite level: firms are bound to complain that they must pay fees on emissions in addition to paying to abate, and that the high fees imperil their competitiveness. Also, high fees could result in pervasive noncompliance that might eventually threaten the legitimacy of the regulatory system. These do not appear to be insurmountable difficulties, however. All three of the countries we have examined have instituted measures that have reduced firms' compliance costs and made fees more politically palatable, albeit at some cost in terms of environmental impact. These include earmarking fee revenue for the use of the firms that pay them, exempting small and economically fragile firms from paying fees, using two-tiered fee structures to reduce regulatory costs for firms that meet emissions standards, and differentiating fees across firms by vintage and geographical location.

Second, earmarked fee revenue should be disbursed in a manner that preserves the incentive properties of emissions fees. The Swedish system appears to be a model in this regard. Fees are refunded to firms in proportion to output so that firms can always boost profits by cutting emissions per unit of output. In addition, unlike the Chinese regulators, Swedish authorities place no restrictions on how firms use earmarked funds, thereby avoiding the extremely problematic and costly responsibility of trying to monitor firms' expenditures in addition to their emissions. Finally, unlike Chinese and Polish authorities, Swedish regulators do not depend on fee revenues to finance their operating expenses.[51]

The last point merits some qualification. In countries where fiscal resources are in extremely short supply, using fee revenue to finance regulatory activity may be a virtual necessity, at least in the short run. In such cases, it may be possible to design institutional

[51] One aspect of the Swedish program that might present political difficulties is the economy-wide mechanism for collecting and disbursing fee revenue that effectively transfers funds from economic sectors with high emissions per unit of output to those with low emissions per unit of output. Those sectors that are net providers of funds would clearly be less apt to support such a system than those that are net recipients. One potential solution to this problem is create a number of smaller sector-wide mechanisms for the collection and disbursal of funds.

mechanisms to dampen perverse incentives created by this arrangement, for example by rewarding regulators who cut aggregate emissions and by creating environmental funds to separate the collection and disbursement of fees as in Poland. Still, if China is an accurate guide, policy makers should recognize that there is likely to be some trade-offs between the dependence on earmarked revenues to finance regulation and the effectiveness of that regulation.

Third, to the extent possible, barriers to the effectiveness of emissions fees created by pre-existing regulation should be removed or mitigated. An example is tax laws that enable Chinese and Polish firms to count most emissions fees as costs.

Finally, emissions fees clearly require some minimum level of credible monitoring and enforcement. Notwithstanding a perception that EI instruments are somehow market-based and are therefore less dependent on an effective public-sector regulatory authority than CAC instruments, our case studies clearly indicate that unless the institutional capability and political will exists to provide a minimum level of monitoring and enforcement, emissions fees will simply not have a significant environmental impact. Of course this is a facile point: it begs the question of how to improve monitoring and enforcement given constraints discussed in Section 2. Part of the answer has to do with building institutional capability and generating political will, topics that are beyond the scope of this paper. But another part has to do with the choice of monitoring methods. Of our three case studies, the only unambiguously successful fee program is in Sweden where CEMs are used to monitor nitrogen oxides emissions. Unfortunately, the cost of CEMs make them an unrealistic choice for most developing countries, at least in the short run. Recall that the annual costs of operating CEMs used in the Swedish nitrogen oxides program (excluding capital costs) is $40,000 per firm while the annual cost of the CEMs used in the US Sulfur Dioxide Program (including capital costs) is $125,000 per firm. Since second-best monitoring methods are the logical alternative, emissions fee systems should probably be restricted to those pollutants (e.g., sulfur dioxide) for which such methods are effective. Unfortunately, second best methods are not well-suited to monitoring particulate emissions, which according to conventional wisdom are the most harmful to human health.

5.2 Tradable Permits

The Emission Trading Program, the United States' oldest tradable permit program for air emissions, has had mixed success. Although it has reduced compliance costs relative to a pure CAC system, cost reductions have been smaller than expected. Moreover, most analysts agree that environmental impacts have been negligible. The key problems include design deficiencies (that mostly arise from an attempt to graft permit trading onto an underlying complex CAC system), uncertainty, and imperfect information. All of these factors raise transactions costs and restrict trading. By contrast, the newer Sulfur Dioxide Program has had strong environmental and economic impacts, in large part because it was consciously designed to overcome many of the problems that have plagued the ETP. For example, the Sulfur Dioxide Program substitutes for rather than complements existing CAC regulations, and places relatively few restrictions on either trading or abatement alternatives. On the face of it, the evolution from the ETP to the Sulfur Dioxide Program suggests that the success of any air permit trading scheme depends largely on getting program design right, and that armed with this lesson, developing country policy makers should be able to develop effective air permit trading programs. However, in our opinion, such a conclusion is unwarranted.

There are several aspects of the US experience that suggest that large-scale air permit trading in developing countries would be problematic. Most important, the success of the Sulfur Dioxide Program depends critically on a high level of monitoring, enforcement, and administration. The sulfur dioxide permit market is viable and robust because participants know that they will receive full credit for emissions reductions and that emissions limits implied by allowances will be strictly enforced. The effective monitoring and enforcement that underpins the program is due to investments and institutions that would be difficult to replicate in developing countries: monitoring depends on CEMs entailing annual expenditures of over $100,000 per firm (recall that program planners deemed second-best monitoring to be inadequate); enforcement of allowances, which is largely taken for granted, depends on effective regulatory institutions; and program administration is carried out by firms and by specially-created regulatory organizations costing millions of dollars per year. In the near term, such investments and institutions are probably beyond the reach of most developing countries. Importantly, a lack of monitoring, enforcement, and administrative capabilities is a far more critical constraint on permit trading than on emissions fees. Permit markets will simply not work absent these capabilities, while emissions fees can, as the China and Poland case studies illustrate.

Another cautionary lesson from the American experience is that, despite the fact that markets are exceedingly well-developed in the United States, transactions costs, uncertainty, imperfect information, and institutional rigidities have hampered the development of permit markets. In the case of the ETP, these factors (together with design problems), have been blamed for the continued thinness of the emissions credit market, and in the case of the Sulfur Dioxide Program, they have been blamed for the low trading volumes in the first several years of the program. Transactions costs, uncertainty, imperfect information, and institutional rigidities are likely to be far more severe in developing countries.

5.3 Environmental Taxes

The only evidence we have on taxes specifically designed to have an environmental impact are carbon and sulfur dioxide taxes in Sweden. Both appear to have had a some impact on emissions--although the impact of the former has been more significant and more clear cut--and to have raised considerable revenue.

The Swedish experience holds several lessons for developing countries. First, the legislative history of Sweden's carbon taxes clearly illustrates that levying taxes set high enough to have an environmental impact is likely to be quite difficult for political reasons. Sweden was able to impose relatively high carbon taxes on industry in 1991 only because it simultaneously cut income and energy taxes. Even so, industry's resultant lobbying efforts succeeded in not only in completely eliminating carbon taxes but also in lowering complementary energy taxes, thereby actually reducing real energy prices paid by industry relative to 1991 prices. In developing countries where the power of industrial lobbies often swamps that of environmental advocates, political barriers to highly visible environmental taxes are likely to be at least as important. Second, not surprisingly, the Swedish experience illustrates that taxes are relatively easy to administer. Sweden has used existing tax authorities to administer its environmental taxes. Marginal administrative costs are estimated to be in the range of one to five percent of total revenues. Third, tradeoffs between environmental impacts and fiscal revenues are significant. Revenue from Sweden's sulfur tax has fallen as the tax has dampened demand for high-sulfur fuels. Fourth, targeting environmental taxes is feasible. Sweden exempted industry from its carbon tax, and also exempted those using coal, oil and peat as feedstock instead of as fuel from its sulfur tax. It is not clear that developing countries would be able to replicate Swedish efforts to create incentives for end-of-pipe abatement by refunding tax payments to firms that install scrubbers since Sweden used CEMs to prevent firms claiming refunds from cheating. However, on the face of it there is no obvious reason why second-best monitoring could not be used for the same purpose.

5.4 Policy Prescriptions

We conclude this section with some final thoughts about the advisability of promoting emissions fees, permit trading and environmental taxes in developing countries. Regarding emissions fees, our case studies clearly indicate that they are a politically feasible regulatory alternative. It is far less clear that they can be an effective means of stimulating emissions reductions. Part of the problem seems to be that many existing emissions fees systems in developing countries have some critical design flaws (e.g., the level and structure of fees, and the mechanism for refunding revenue) that in theory can be corrected. But at least as important as these design flaws is a lack of institutional capability and political will to provide a minimum level of monitoring and enforcement, a problem that is not so easily corrected. This suggests that the appropriateness of emission fee systems will vary across countries, across regions within countries, and also across pollutants. Yet there are several aspects of the case studies presented above that argue in favor of promoting their establishment at a national level. First, once a fee system is in place, regulators may be able to raise fees and strengthen enforcement over time (as has happened in Poland), a strategy that may minimize resistance to establishing a high-fee system. In other words, even low fees with negligible impacts provide regulators with a "foot in the door." By the time high fees are politically feasible, marginal abatement costs may have risen making the costs savings that accrue to a fee system more attractive. Second, fee systems provide valuable revenue to finance regulatory activity and direct investment in environmental projects.

Regarding permit trading, we have argued that the success of US policy makers at developing trading programs, based in part on lessons learned about program design, is not likely to be easily replicated in developing countries. For most developing countries, emissions fee programs would appear to be a more realistic and appropriate policy alternative. The fact that (to our knowledge) there are no functioning air permit trading programs anywhere in the developing world supports this view.[52]

Finally, regarding environmental taxes, the one case study we have examined suggests that they do represent means of overcoming financial and institutional constraints on direct regulation; can raise significant revenue; and can be targeted. The principal barrier to their use appears to be the political difficulty of levying taxes that are high enough to have an environmental impact.

[52] Regulators are attempting to set up permit trading systems in at least four cities: Almaty, Kazakhstan; Santiago, Chile; Chorzow, Poland; and Mexico City, Mexico. So far these efforts have not been particularly successful. See Margolis, Trivedi, and Farrow (1995); Huber (1996); and Dudek, et al. (1992).

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