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Applying economic instruments in developing countries: from theory to implementation

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ABSTRACT. The paper describes a number of developing country applications of economic instruments (EIs), focusing on how policy makers—mostly in Asia and Latin America—have addressed implementation problems. The informational and institutional demands of EIs can be as great as with regulations; in any event, the former are mostly used to complement not replace the latter. Consideration of political acceptability has conditioned both instrument design (e.g. grandfathering of tradable permits, non-compliance fees rather than simple pollution charges) and phasing of implementation (e.g. starting with local experimentation, setting low initial charge rates). With the advance of market-oriented economic reforms in the developing world, the policy and institutional environment should become more conducive to applying EIs; with greater political openness in many countries, the scope for involving the media, non-governmental organizations, and the public at large in environmental enforcement (e.g., through information disclosure programmes) should also increase.

1. Overview of issues

Economic instruments (EIs) encompass a rather heterogeneous toolkit of policies whose main defining feature is their reliance on markets and the price mechanism to internalize environmental externalities. In other words, these instruments 'leave decentralized agents their freedom of choice, of decision and of trade, while at the same time affecting the schedule of advantages and disadvantages associated with the consequences of those choices' (Godard, 1994).

The discussion of the relative merits of EIs as policy instruments is normally framed in terms of a contrast with the conventional approach applied in most countries since the inception of environmental policy, viz., a reliance on laws and regulations which dictate in some detail the measures which polluters must adopt under penalty of fines or other sanctions. This approach, which is loosely referred to as 'command-and-control' (CAC), has been criticized by economists on grounds of both static

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and dynamic inefficiency: the former because it requires compliance with the same standards by different pollution sources irrespective of their marginal compliance costs; the latter because it provides little incentive to technical improvement once compliance has been achieved. Seldom, however, is environmental policy designed solely to meet efficiency criteria, and seldom is policy choice strictly dichotomous. Normally, environmental policy seeks to strike a balance between environmental effectiveness and economic efficiency, broadly defined to include the administrative costs of implementation in addition to polluters' abatement costs. A combination of regulatory measures and economic instruments will often achieve the desired balance among different policy objectives more effectively than either alone. In effect, by mixing instruments policy makers are able to exploit the advantages of each.

Whatever the policy mix chosen, problems of implementation can arise for several reasons: (i) administrative complexity exceeding public and private sector institutional capacity; (ii) political resistance from those who perceive themselves to be adversely affected; (iii) possible inconsistencies with the existing legal framework; (iv) design flaws involving a mismatch between the type of instrument chosen and the nature of the problem targeted—e.g., when applying, an instrument requires close monitoring of polluters, but the large number, small size, and geographic dispersion of those polluters makes such monitoring unfeasible.

As governments acquire more experience in applying economic instruments, they are also learning more about the difficulties of implementing specific instruments. This applies as much to developed countries as to developing countries. Still, the literature dealing with problems of implementation and how they might be overcome is sparse. One of the first and most comprehensive discussions of implementation strategies for environmental taxes in developed countries is OECD (1996). For developing countries, where experience with such instruments is admittedly more limited, there has as yet been little comparable work.[1] This paper is intended to begin filling the gap. While the focus of discussion is principally on Asia, examples are also drawn from Latin American experience and reference is made to certain challenges facing economies in transition.

The paper is organized as follows. In the next section, following a brief review of the developed country (i.e., OECD) experience with applying economic instruments, the experience of developing countries is presented in greater detail, with an emphasis on implementation problems. Types of instruments discussed are: pollution taxes/charges,[2] product taxes/

[1] Panayotou (1994) contains some discussion of the specific conditions in developing countries that affect choice of policy instrument.

[2] The terms taxes and charges are used interchangeably here, though from an administrative point of view there can be an important difference between the two. Authority to levy a new tax normally resides with the Ministry of Finance (and may in some countries require prior legislative approval), whereas other ministries/agencies frequently enjoy some discretion to levy charges. Also, revenue from a tax normally accrue to the general government budget, whereas charge revenue is often 'earmarked' for specific uses.

charges, environmental funds, deposit-refund schemes, and tradable permits. Also considered are so-called suasive instruments (SIs), in particular, voluntary agreements and information disclosure programmes. In section 3, some general lessons are drawn regarding ways of coping with the more common implementation problems. Section 4 concludes.

2. Experience with applying economic instruments

Actual practice in applying EIs, while guided to varying degrees by theory, reflects the need to accommodate policy to the complexities of the real world. This may mean mixing elements of CAC, EIs, and/or SIs in order to realize multiple objectives; it may also mean modifying the design or phasing the introduction of a particular instrument to make it more administratively manageable or politically acceptable.

This review of the experience when applying EIs is meant to be illustrative rather than exhaustive. Moreover, while the primary concern is with implementation, this is not a systematic evaluation of the environmental effectiveness or cost efficiency of specific instruments in practice. There are very few *ex post* policy evaluations on which to report; most studies are *ex ante* estimates of hypothetical cost savings from adoption of a least-cost policy option relative to a proposed regulation or standard (see Hahn, 1989 for a summary of cost assessments of tradable permit schemes in the United States; also Tietenberg, 1990 for a review of cost studies of alternative air pollution control measures). In future, as experience with EIs accumulates, there will be scope for further research into their actual costs and other characteristics in comparison with the relevant alternatives. OECD (1997b) proposes one framework for conducting evaluations of various dimensions of EIs, both *ex ante* and *ex post*.

2.1 The OECD experience in brief

In OECD countries, experience with the application of EIs was rather limited until a decade ago. An early OECD review (1989) of EIs in 15 member countries (as of 1987) found examples of approximately 100 EIs in use, but the conclusion was that very few actually had any incentive effect and that, by and large, environmental management systems could still be characterized as dominated by command-and-control policies with some financial and economic add-ons.

A more recent look at OECD experience since 1989 (see OECD, 1994b) finds an increased reliance on EIs. Among the reasons given are:

a general tendency towards de-regulation of economic activity and greater reliance on markets and the private sector;
tighter budget constraints facing governments, which has stimulated an interest in instruments that are designed with 'built-in' compliance incentives and that can also raise revenue;
within the environmental arena, a growing concern with problems which do not lend themselves readily to CAC approaches—e.g., diffuse and mobile sources of pollution and global pollution problems; and
sharply increasing costs of pollution control as governments and their constituencies seek continued environmental improvements from what has

already become a rather strong baseline performance in many OECD countries.

An inventory of EIs in use in OECD countries as of the beginning of 1992 shows that product charges are most common (79 examples in 20 countries), followed by emission charges (70 examples in 21 countries). Other instruments follow at some distance: 33 examples of deposit-refund schemes in 16 countries, only 12 examples of tradable permit schemes, eight of which are found in the United States, and eight examples of enforcement incentives (Barde, 1994). Thus, as of 1992, a total of 169 EIs were being used in 23 OECD countries. Product charges and deposit-refund schemes were the two instruments which experienced the largest increase in frequency of use between 1987 and 1992 (considering those countries surveyed in both years). There is no tendency, however, in OECD, countries towards replacing the basic regulatory approach with a purely economic one. 'Economic instruments are complements mostly and substitutes only sometimes for other types of approaches' (OECD, 1994b, p. 187).

In contrast to the 1989 assessment, the most recent evaluation of EIs in OECD countries (OECD, 1997b) finds that they *have* had noticeable incentive effects; e.g., tax/charge measures have generally led to changes in emissions levels and not simply been absorbed as a cost by polluters. Possibly this more favourable assessment reflects the time lag in the behavioural response to new EIs, since an important element of the response is the renovation of the capital stock. As the study notes, however, detecting an incentive effect of EIs is not the same as proving that they are either more cost-efficient or more environmentally effective than the relevant alternatives.

2.2 Developing country experience with applying EIs

As yet, there has been no survey of the use of EIs in developing countries comparable to those done for OECD countries. Nevertheless, based on a limited review of the literature on environmental policy in the newly industrializing economies (NIEs) of Asia and Latin America (O'Connor, 1994, chapter 5; OECD, 1994a), a few observations can be made. While some countries have used EIs in certain applications since at least the mid 1970s, as in OECD countries the interest among NIEs in applying EIs has been gaining momentum in recent years. Thus far, the most frequent applications have been similar to those in OECD countries: emission/effluent charges appear to predominate, with product charges also fairly common. Deposit-refund schemes are beginning to operate in the higher-income NIEs, while there are only a few instances of operational tradable (or auctioned) permit schemes (Chile and Singapore being the outstanding examples). As in OECD countries, the basic policy framework in all these countries is a regulatory one and EIs are generally designed as complements to regulation. Some NIEs have been experimenting with the use of quasi-voluntary agreements and information disclosure, which are also attracting wider interest in OECD countries (though a few countries like Japan and The Netherlands have relied on the former for quite some time, and the United States has considerable experience with the latter).

Certain features of the institutional context in developing countries can give rise to policy implementation problems not commonly encountered in OECD countries. In some cases, this makes implementation equally problematic for a regulatory and an incentive-based approach. For example, limited information on baseline environmental quality and weak capacity to monitor pollution levels of specific sources hamper either approach. Russell and Powell (1996) argue that introducing and administering an environmental management system based on EIs will never be institutionally easier than one based on a regulatory approach and in most cases will be more difficult. The information intensity of certain EIs is a major reason given, but as discussed below there are more or less information-intensive EIs just as there are more or less information-intensive regulatory approaches. It remains true, however, that an EI designed to achieve static efficiency can be highly information intensive in the common case where polluters have different impacts on ambient environmental quality depending on location.

Weak enforcement of environmental regulations is also characteristic of many developing countries. While it has been suggested that greater reliance on EIs could help remedy this problem, that depends on the underlying causes. If it reflects a weak political commitment to environmental goals, this is unlikely to be changed by greater reliance on EIs—unless politicians (and the public at large) are persuaded that such a shift would significantly reduce the costs of achieving those goals. If, on the other hand, it reflects a principal-agent problem in which poorly paid government officials face weak (or even negative) incentives for strict enforcement, then providing enforcement incentives could help, provided they are backed by stiff penalties for accepting gratuities from polluters combined with a reasonable probability of detection.

Some features of the institutional environment in developing countries may favour the application of EIs (or SIs). With respect to EIs, environmental policy makers newly trained in the theory and design of EIs may enjoy greater freedom to experiment with their application than their OECD counterparts, who are more likely to be constrained by a long history of reliance on regulation and an entrenched bureaucracy accustomed to the old rules. As for SIs, in situations where government regulation is lacking or government enforcement capacity weak, civil society may assume a prominent role in what Pargal and Wheeler (1996) call 'informal regulation' through various forms of public pressure on and/or negotiation with polluters.

This section reviews several examples of EI/SI applications in developing countries, focusing on implementation issues and how they have been addressed. The examples cited come overwhelmingly from newly industrialised economies (NIEs), which reflects in part the coverage of the available literature but also the greater experience with EIs in those countries than in less-developed countries. It may be, as Russell and Powell (1996) have argued, that developing countries come to rely more on EIs the more their institutional capabilities come to resemble those in the OECD countries.

Emission/effluent charges

The early applications of pollution charges in developing countries usually took the form of non-compliance fees rather than simple charges applied to all emissions. This may be because of political concern over the 'double burden' associated with the latter. More recently, however, there has been a shift towards graduated charge schemes, with lower rates for within-standard discharges. This shift addresses one of the major weaknesses of non-compliance fees, viz., their failure to reward reductions in pollution beyond those dictated by standards.

Since the early 1980s, Korea has had a pollution charge scheme that initially operated as a simple non-compliance fee. At first, the Environment Administration (now the Ministry of Environment) could levy the charge only if the polluter continued to violate standards after having been issued an improvement order, but since 1986 the levy is automatic once emissions exceed the permitted level (Chung and Lee, 1992). Since the charge was introduced by an administrative act, it could be implemented with a minimum of delay. Ten air pollutants and 15 water pollutants are subject to the charge. Originally, the charge was based only on pollutant concentration, but in 1995 the charge formula was modified to include total load as well (OECD, 1997a). Moreover, polluters are now charged once pollution load exceeds 30 per cent of the threshold level. Thus, while not all pollution has a cost, neither is all pollution within standards costless (Rhee, 1994; OECD, 1997a). The amount of the charge varies with the location of the facility, the duration of excess discharges and the number of previous violations.

China has had a national system of pollution charges on air emissions, wastewater discharges, noise, solid waste, and radioactive wastes for the last 15 years (Potier, 1995; NEPA, 1996).[3] As in Korea, the charge is levied as a non-compliance fee, but only on the 'worst case' pollutant from a given source; it is based on both excess pollutant concentration (above standard) and total volume of wastewater discharge. As worst cases are successively cleaned up by a given source, the levy shifts across pollutants (Wang and Wheeler, 1996).

The system has been introduced in three phases, beginning in 1979 on an experimental basis in Suzhou city and then gradually extended to 27 provinces, autonomous regions, and cities directly under the central government. In 1982 it was extended nationwide. During the current phase, which began in 1988, the emphasis has been on reforming the system for allocation and use of charge revenues (see discussion of environmental funds below). The charge is backstopped by a fourfold system of penalties for serious violations of standards. In principle, charge rates are set at a level slightly above the average operating costs (including a depreciation factor) of pollution control facilities, to encourage broad compliance with standards. In practice, since they are not indexed to inflation, their real value has been eroded over time to the point where, at present, they

[3] Effluent discharges and waste gas emissions are the two main sources of charge revenue: they accounted for around 60 per cent and 30 per cent, respectively, of cumulative charge revenue between 1979 and 1995 (NEPA, 1996).

provide only a weak incentive for further pollution reduction. Recent efforts to raise the charge, however, have met with strong opposition from industry.

At the local level, environmental officials have considerable discretion in enforcement of the pollution charge scheme and indeed the intensity of enforcement (the actual revenue collected as a share of potential revenue) has varied widely across provinces and over time. As Wang and Wheeler (1996) point out, this does not mean that enforcement has been arbitrary. Geographic variation in its intensity reflects differing local conditions that are thought to affect local people's valuation of a clean environment (e.g., population density, per capita incomes, and average education levels).

In Malaysia, in the mid 1970s, the Department of Environment (DOE) introduced a permitting system for palm-oil mills which incorporated features of an effluent charge in that the licensing fee could be varied according to the quantity of processing waste. With the rapid expansion of palm-oil production during the 1970s, this industry soon developed into the largest source of water pollution in the country (Vincent, 1993). The system was built on effluent standards which were progressively strengthened over four years, starting from 5000 parts per million (ppm)—i.e., one-fifth the level in untreated palm-oil effluent—and declining to 500 ppm. The gradual phase-in was designed to give industry time to construct treatment facilities and acquire experience in operating them. The licensing fee consisted of two parts: a flat administrative (processing) fee of M$100 (roughly US$42 at 1975 exchange rate) and a variable effluent-related fee. For releases into a watercourse, the latter fee was set at M$10/tonne of BOD load discharged (up to the standard). An excess fee of M$100/tonne BOD was levied on discharges above the 5000 ppm standard. In effect, then, there was a two-part effluent fee, similar to the newly modified Korean charge. The Malaysian government reserved the right to grant a partial or full waiver of the effluent-related portion of the fee to those mills conducting research on new treatment methods. Beginning in the second year, the standard became not only more stringent but mandatory, backed ultimately by the threat of license cancellation.

In the first two years of the programme, the pollution load from palm-oil mills fell from 15.9 to 2.6 million population equivalents and by 1989 the population equivalent was less than 1 per cent of its level at the inception of the programme, despite the fact that palm-oil production was at a record high. The CAC aspect of the programme became dominant over time, partly because the effluent-related fee was gradually eroded in real terms by inflation. The DOE acted on its threat to sanction seriously non-complying mills. In 1979 it suspended the license of one mill and, between 1981 and 1984, it took legal action against an additional 27 mills. The costs to industry of the programme were not negligible, with cumulative expenditures on building and operating treatment systems having reached M$100 million by 1984. Those costs, however, were mostly shifted on to palm-oil growers: they could not be passed on to consumers in a highly competitive world market, while individual mills exerted considerable local market power over neighbouring growers.

Product taxes and tax differentiation
Under certain conditions, a product (or input) tax may be a suitable substitute for an emission/effluent charge, viz., where (i) consumption of the taxed product is closely correlated with pollution levels, (ii) the price elasticity of demand for the product is high, and (iii) substitutes are less polluting. The advantage of a product tax over a pollution tax is the lower monitoring and enforcement costs. The disadvantage is that, even where (i) is satisfied, the product tax only induces reduced consumption of the product (or input) but does not induce lower pollution per unit of consumption.

The case of leaded versus unleaded gasoline is one where all three conditions apply. In an effort to phase out use of leaded gasoline, governments in many OECD countries as well as a number of other countries have used differential taxation of leaded and unleaded gasoline, generally in combination with some regulatory measure. The cases of Thailand and Taiwan are illustrative. In the former, soon after the release of a study (USAID, 1990) which estimated that by the age of seven Bangkok children collectively suffer a loss of up to 700,000 IQ points as a result of elevated blood lead levels, the Thai government took steps to encourage substitution away from leaded gasoline in motor vehicles. Beginning in May 1991, the government introduced unleaded gasoline at a slight discount relative to leaded gasoline, subsidizing the former from a surtax on the latter. It also introduced a regulation requiring that all cars sold in Thailand from September 1993 be equipped with a catalytic converter. As a result of the two measures, the market share of unleaded gasoline has risen steeply in recent years (to roughly half the market).

Beginning a few years earlier, Taiwan followed a similar price differentiation strategy, with comparable results. Taiwan has not, however, required mandatory installation of catalytic converters but relies instead on a requirement that all new vehicles use unleaded gasoline and comply with new emission standards[4] (O'Connor, 1994). Monitoring reports by the Taiwanese Environmental Protection Administration (EPA) indicate that the average lead content in ambient air in Taipei decreased by more than 50 per cent from 1989 to 1992 (Pan, 1994). Combining price differentiation with an emission standard can mimic the effects of an emission charge (Eskeland, 1994): the former discourages the polluting activity (in this case, the burning of leaded gasoline) while the latter should make the activity cleaner (by inducing a reduction in the lead content of leaded gasoline).

Evaluating the relative contributions of the regulatory measure and the tax measure to the reduction in lead levels poses a difficult research challenge, one common to the many other instances where CAC and EIs are employed in combination.

Environmental funds
A number of developing countries and economies in transition have established environmental funds to finance certain environmental expen-

[4] One possible drawback of standards—whether technical or emission standards—that apply only to new vehicles is that they provide an incentive to delay the scrapping of older, usually more polluting vehicles.

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ditures. Such funds are best viewed as transitional mechanisms to mobilize financing to tackle an accumulated backlog of environmental problems. They can also play a useful role where capital for environmental investments having a high social return cannot be raised through established financial institutions.

There are several possible ways of financing such funds: a contribution from the general government budget; revenue from pollution charges; foreign donor support; private contributions; or some combination. Earmarking of financial resources to subsidize environmental investments is controversial, and the pros and cons need to be weighed in each specific case. If a pollution tax is set high enough to achieve the government's environmental quality target, then subsidizing pollution reduction measures would not be justified. In practice, however, pollution taxes have often been set too low to have an adequate incentive effect. Pollution abatement subsidies may reinforce the tax incentive, but at the possible expense of unduly encouraging investment in the polluting activity (Baumol and Oates, 1988, chapter 14).

Insofar as the polluter-pays principle is not yet widely accepted, and pollution charges are thus viewed as revenue raising rather than incentive devices, earmarking may increase the charge's political acceptability, irrespective of whether the revenue is rebated to industry or used to finance public waste treatment facilities. If instead it were added to general government revenue, polluters might object that they were being asked to bear more than their fair share of the financing of public goods provision.

From an efficiency perspective, earmarking has one important drawback. Setting aside revenue in a separate fund insulates those resources from competition among alternative uses, with the risk that the earmarked revenue would continue to be spent on environmental projects even if social rates of return were to fall below those on other projects. The result could be the creation of excess waste treatment capacity, as appears to have occurred with the water pollution levy in The Netherlands (OECD, 1996). More importantly in a developing country context, scarce resources can be diverted from uses that have a higher social priority.

In the event that charge revenue is used to replenish the environmental fund, there could be an incentive compatibility problem if the charges are levied as non-compliance fees and if those responsible for fee collection are also responsible for fund management. In effect, they would have an incentive to encourage continued non-compliance to ensure a steady revenue stream. In the case of Korea's environmental fund, established simultaneously with its pollution charge, this problem is partially addressed in that only one-third of the funding comes from the non-compliance fee with the remainder coming from a government budgetary allocation and from interest income. Also, the devolution of fund management to a semi-governmental body, the Environmental Management Corporation (EMC), separates responsibility for fee collection from that for fund management.

China has had a system of environmental funds linked to its pollution levy scheme for many years. The revenues collected in a given jurisdiction are deposited in a local environmental fund managed by a designated bank. Until 1988, the revenues from the levy were largely allocated as grants to subsidize pollution control measures, but since then there has been a shift towards greater reliance on loans. Roughly 80 per cent of the funds are lent or given to enterprises for pollution control investments, with the remainder going to local environmental agencies to finance the capital and operating costs of the charge scheme—e.g., the purchase of monitoring equipment and analytical instruments, the hiring and training of additional staff. Charge revenues have been a major source of financing for pollution control investments: from 1982 to 1986, they accounted for almost 30 per cent of pollution control expenditures in the steel industry; during that period the industry's rate of compliance with discharge standards rose from one-third to 60 per cent.

Deposit-refund schemes

These have been implemented in a few higher-income developing countries, notably Korea and Taiwan. In Korea, the Ministry of Environment initially proposed an ambitious deposit-refund programme, covering a wide range of products—food and beverage containers, pesticide containers, batteries, tires, lubricant oil, plastics, and certain domestic appliances (e.g., televisions and washing machines). In the end, the list was considerably shortened following objections from the Ministry of Trade and Industry, and deposit rates on those items covered are generally very low. For example, the deposit rate on PET (polyethylene terephthalate) bottles, depending on volume, ranged from 3 to 7 won in the mid 1990s (less than 1 US cent at the then prevailing exchange rate). Rhee (1994) argues that a reason for the low deposit rates (as well as low emission charge rates) is the government's concern to control inflation, but the strong influence of industrial interests on government policy is an additional factor. The result is that there is little incentive for waste recovery, and refunds claimed represent only a tiny fraction of deposits collected. In 1994, only 8.6 per cent of funds collected from deposits on beverage containers were returned. Thus, the government has plans to raise the deposit rate by the year 2000 to 65 per cent of actual collection and treatment costs (OECD, 1997a).

In Taiwan, the deposit-refund scheme has been more effective. Since 1988, Taiwan has been implementing a recovery/recycling scheme for several types of solid waste, including PET bottles, glass bottles, aluminium cans, waste paper, used tires, lubricant oils, mercury cell batteries, and pesticide containers. A deposit-refund scheme to support the recycling effort is to be introduced in a step-wise fashion, beginning with PET bottles. There are some 104 manufacturers of PET bottles in Taiwan making some 260 million bottles a year. Each is required to submit a recycling and disposal plan to the provincial or municipal authorities. Members of the industry have formed a foundation which administers a joint recycling fund to cover costs of collection and recycling of the bottles, with the fund replenished from a levy on the sale of each bottle (Chien, 1991). The deposit rate per PET bottle has been set at NT$2.00 (around US$0.08), with a portion of this (roughly one-quarter) refunded upon delivery to the recycling plant by one of the 23 salvaging companies that recover the bottles from some 14,000 collection locations. The Taiwanese EPA set a target recycling rate of 50 per cent for the first two years (1989–90) and 60 per cent by the fourth year. As of the third year, the accomplished recycling rate had only reached 41 per cent, but by the fourth year (1992) it jumped to almost 80 per cent (Pan, 1994), comparing favourably with rates in OECD countries.[5]

In short, the deposit rate in Taiwan appears to have been set at a level yielding strong incentives for recovery and recycling. The wide distribution of collection points and the development of a sizeable salvaging industry have also contributed to the scheme's success. The focus initially on one product presumably made the system more manageable and made possible incremental learning that should facilitate the extension of the scheme to other products.

Tradable (or auctioned) permits

To date, there has been rather limited experimentation with tradable permits in developing countries, though the experience with permit trading in the USA has stimulated a growing interest. Two examples stand out, however: Chile's 20-year experience with tradable water rights (which Mexico also introduced in 1992), and Singapore's use of permit auctions for chlorofluorocarbon (CFC) import/use and for motor vehicle ownership rights. A key issue in any tradable permit scheme is the initial allocation of property rights (or permits). In almost all cases, a variant of 'grandfathering' has been used to validate the rights of existing market participants (or polluters), though the Singaporean auction scheme described below imposes a levy even on the grandfathered permit holders. Other essential features of a permit trading scheme include: (i) a reliable database on baseline emissions (or consumption, or catch); (ii) transparent and simple trading rules; and (iii) an accurate monitoring, record-keeping, and reporting system.

In Chile, reform of the centralized system of water allocation occurred in parallel with market-oriented economic reforms that included trade liberalization. A more decentralized water allocation system (together with re-privatization of land ownership) was seen as vital if Chilean agriculture was to respond flexibly to the new market opportunities created by those reforms. As Rosegrant and Gazmuri (1994) note, shifting from administrative to market allocation of water implies transferring significant amounts of power from the government to water users, while also relieving the government of large investments in water infrastructure and operation and maintenance costs that are also shifted to users. A potentially large environmental benefit of water markets is the incentive they provide for greater water conservation, with the additional benefit of reduced investment requirements for constructing new water infrastructure that in itself

[5] For example, at roughly the same time, the PET bottle return rate in The Netherlands and Sweden was 90–100 per cent, in Denmark 80–90 per cent, and in Australia 62 per cent (OECD, 1994b). It should be borne in mind that return rates and recycling rates are not always identical; moreover, definitions of recycling rates can differ for a variety of reasons (see OECD, 1992:83 for a discussion of definitional issues).

can cause significant environmental disruption. On the other hand, large transfers and releases of water may alter temperature and flow conditions in ways that adversely affect fish and wildlife.

An important outcome of Chile's water policy has been the purchase of agricultural water by urban water suppliers without having to buy land or expropriate water from farmers through state intermediation. Farmers normally sell small portions of their rights while maintaining agricultural production with efficient on-farm irrigation technology for orchard and vegetable crops. Market allocation of water has also stimulated efficiency improvements in urban water and sewage services, since water and sewage companies can no longer expect a virtually free supply of water. Rosegrant and Gazmuri estimate that at a minimum $400 million in new infrastructure would be required to generate the incremental water that has been saved through efficiency gains. By eliminating broad subsidies that benefited better-off farmers and urban consumers, the government has freed up resources to provide targeted subsidies for poor urban water users and small farmers.

Singapore's CFC permit auction scheme began in the late 1980s, after the ratification of the Montreal Protocol.[6] Each quarter the national consumption quota (as defined under the MP) was allocated among importers and users, half on the basis of historic consumption (grandfathering) and half through a sealed-bid tender. Importers and users were required to register to participate in the bidding process, with each firm specifying the amount of its demand and its bid price. Bids were then ranked by price, with the lowest winning bid price (i.e., the one just exhausting the stock) serving as the unit permit price. That price was then charged on the full national allotment, including the pro-rated half. Initially, there was a steep increase in permit prices (caused in part by stockpiling), providing users with a strong incentive to adopt conservation measures, substitutes, and alternative technologies. As a result, CFC demand fell sharply. The auction procedure enabled the government to appropriate a sizeable share of the scarcity rents, which it then used to subsidize recycling services and the diffusion of information on alternative technologies (O'Connor, 1991). Since the decline in demand depressed the permit price, the government accelerated the phase-out schedule in an effort to support the price and maintain the incentive to continued demand reductions.

Singapore has also devised a vehicle ownership quota system designed to limit the growth in supply of private automobiles. The Vehicle Quota System (VQS) was introduced in May 1990, following essentially the same principle as the CFC quota system. Under VQS, anyone wishing to own a vehicle (except for buses and emergency vehicles) must have a certificate of entitlement (COE). Those vehicles already registered at the inception of the system were assumed to have a COE (another instance of grandfathering), while anyone wanting to buy a new one is required to bid for a COE in monthly tender exercises. Each bidder must indicate the amount he or she is willing to pay for the right to own a vehicle in a particular cat-

[6] Following the developed country phase-out schedule, Singapore's net imports of CFCs should have fallen to zero by now; Singapore does not produce CFCs.

egory. Bids are ranked from highest to lowest; each successful bidder pays a COE price equal to the lowest successful bid price. The COE is valid for ten years from the date of registration of the vehicle, after which the COE must be renewed at the prevailing price defined as the 12-month moving average price of the COE in that vehicle category. By mid 1992, the COE price premium for standard cars had risen to the range of S$20,000 (roughly US$12,000) and for a Honda Civic LS the COE price represented one-quarter of the total sale price. When the price of an ownership permit is added to other price-augmenting measures—an import duty, registration fee, additional registration fee, and annual road tax based on engine capacity—the final cost of car ownership in Singapore is 4.5 to 5 times the landed vehicle cost (Fan, Menon, and Olszewski, 1992).

The effective functioning of the CFC quota auction depends on a sufficiently large number of bidders to provide adequate safeguards against collusive bidding. Another desirable feature, from an efficiency standpoint, is that the benefits from reducing consumption are independent of who makes the reductions (i.e., unit emissions of a given CFC cause the same amount of environmental damage no matter who emits). Given Singapore's geographic concentration, roughly the same applies to the vehicle ownership entitlement scheme. Few other countries share this characteristic. In Thailand, for example, a vehicle driven largely in Bangkok has a very different effect on congestion and pollution from one driven mostly in a rural district or a small provincial capital. Thus, an auction would have to be localized, but preventing the registration of vehicles in one locality for use in another would pose serious enforcement problems.

Suasive instruments: voluntary agreements and information disclosure

Suasive instruments (SIs) rely on voluntary compliance by polluters, motivated either by the threat of adverse or the prospect of favourable publicity. Environmental education and awareness raising are key elements of any policy designed around SIs, since without an informed public such reputational incentives would be weak or non-existent. If, on the other hand, consumers are willing to act on their environmental preferences—e.g., by choosing to buy products with an 'eco-label' or boycotting the products of firms known as serious polluters—this may induce firms to improve environmental performance.

Voluntary agreements have a long history in local environmental policy making in Japan (Haga and Yano, 1992) and they are also commonplace in The Netherlands (Suurland, n.d.). Among developing countries, Indonesia has one of the richest experiences with this approach. Formalizing a firm or industry's commitment to reduce pollution in a voluntary agreement (whether with government, a non-governmental organization, or a citizens' association) may reinforce the credibility of that commitment. One of the attractions of such agreements is that, assuming reputational incentives and/or private enforcement efforts are effective, they place only modest demands on government's own monitoring and enforcement capacity. They are not, however, equally applicable to all types of environmental problems—e.g., they can involve high transactions costs when many small polluters are involved and they may fail to internalize fully external costs when these fall on third parties (see O'Connor, 1994, pp. 134–135).

In Indonesia, pollution reduction agreements are part of a programme, known as PROKASIH (or Clean River Programme), begun in mid 1989 and focused on cleaning up the most heavily polluted rivers (Woods *et al.*, 1992). Major pollution sources along the 20 dirtiest rivers were originally targeted, but presently some 34 rivers are covered by the programme. Letters of agreement have been negotiated between provincial governors and company directors. While participation in the programme is not voluntary, the letters of agreement are not legally binding. Though the terms vary, in general enterprises commit themselves to cutting effluent concentrations and loads in half within an agreed timeframe. While certain details of the agreements were not made public, the government did on occasion use publicity to influence participants' compliance. For example, a public announcement in 1991 by the then Population and Environment Minister of the names of companies violating their agreements appears to have redoubled compliance efforts.

As of 1994, 1,405 establishments were participating in PROKASIH. Along 18 out of 34 rivers, participating plants have significantly reduced their pollution loads (in terms of BOD), while for nine rivers average daily pollution load has increased since 1990–1991 (Afsah, Laplante and Makarim, 1995).[7] While the data show sharp BOD reductions in the initial years of the programme, more recent indications are that BOD loads have been rising again, raising questions about the sustainability of such an approach in the face of strong growth pressures.

Since mid 1995, the PROKASIH programme has been superseded by a programme known as PROPER[8] (Programme for Pollution Control, Evaluation and Rating), which revolves around a scheme of colour ratings applied to firms (with five hues from black to gold) based on their environmental performance. Performance evaluation is repeated at intervals, firms are re-rated, and each time the ratings are disclosed to the public. Preliminary indications are that the programme has been quite effective, at least in improving the performance of the heaviest polluters (those rated black or red). Of the 187 plants rated initially in June 1995, 115 were given a red colour code and six black; by September 1996 the number of red firms had fallen to 87 and of black firms to only one (Tietenberg, 1997, based on World Bank data).

[7] Data reported by Afsah *et al.* (1995) on individual PROKASIH plants suggests that the bulk of pollution reductions have originated in a small number of establishments.

[8] PROPER appears to have been inspired by the US national 'right-to-know' legislation, the Emergency Planning and Community Right-to-Know Act (EPCRA) of 1986, which requires that certain types of establishment report their annual toxic chemical releases and transfers to the US Environmental Protection Agency (EPA), which in turn makes a toxic release inventory (TRI) available to the public. In turn, PROPER has inspired a similar programme in the Philippines known as EcoWatch, launched in 1997.

An important difference between a system of strictly voluntary agreements and right-to-know legislation such as exists in the United States are the information disclosure requirements and costs. In the case of the US toxic release inventory (TRI), the EPA estimates that US companies spend US$346 million a year just to monitor and report their releases, before any investments are made in pollution control. In the case of voluntary agreements, how much polluting firms are willing to pay for self-monitoring will depend on the strength of reputational incentives, and the incentive for accurate information disclosure will depend in turn on the cost to the community of independent verification of environmental performance. If an entire industry's reputation is at stake, however, another influence on individual performance may be peer pressure exerted, for example, through an industry association.

3. Addressing obstacles to policy implementation

This section sets out some suggestions for addressing the most common implementation problems encountered when applying EIs. In some cases, the suggestions apply to regulatory or mixed policy instruments as well. Of the four obstacles to effective implementation mentioned above, the two that recur most often in the examples just reviewed are strong political opposition and weak institutional capacity, so those are the primary focus here.

3.1 Building political support

Identifying potential winners and losers. A transparent and participatory policy-making process provides a mechanism for *ex ante* preference revelation. The strength of support for, or opposition to, a proposed measure can be gauged and, if necessary, the policy redesigned before actual implementation. A policy-making process which does not permit such *ex ante* debate is more prone to *ex post* obstruction of implementation by potential losers.

What may matter most to those polluters whose costs will be increased by a particular policy is the perception of fairness in the design and administration of the policy in question. With regulations or permit trading schemes, the practice of grandfathering has been the most common means of winning support from incumbents, who represent a far more clearly defined interest group than potential new entrants into the affected industry or activity. In the case of charge schemes, political expediency has often dictated an initial reliance on non-compliance fees; at a later date, such schemes can be modified to incorporate a levy (possibly at a lower rate) on discharges within standards, in this way ensuring some dynamic incentive effect.

The degree of industry opposition to a new pollution tax is likely to depend on the firms' ability to shift the tax towards either consumers or suppliers and thus, *ceteris paribus*, is likely to be stronger the more competitive the industry's product and input markets. Under those circumstances, a system of tradable permits (with grandfathering of the initial allocation) could prove more politically acceptable than one of charges. Also, the more market participants, the easier it should be to

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ensure a reasonably competitive permit market. A more general advantage of tradable permits over charges—one of special relevance in some developing economies and economies in transition—is that the prices of the former adjust automatically to inflation whereas the latter do not. Other factors—notably information on the characteristics of abatement cost and damage cost functions—also need to be considered in choosing between a price-based instrument like a tax and a quantity-based instrument like tradable permits (Weitzman, 1974).

Clearly defined fiscal objectives. Public support for EIs is apt to be greater where taxpayers are well-informed about the government's intention in introducing a new pollution tax. If the tax is intended exclusively as an incentive device, then a full rebate to the taxpayers (e.g., the polluting firms) could effectively neutralize opposition. In that case, however, the rebate formula needs to be designed so as to reward those firms that do most to reduce pollution.[9] Assuming the government intends to raise revenue, then earmarking the tax revenues for environmental expenditure can enhance political acceptability, but it also reduces budgetary flexibility. Where large surplus revenues are not anticipated, this may not be a serious constraint. One way to introduce an element of flexibility would be to stipulate that, above some amount, any additional revenues would revert to the general budget.

Where government prefers to maintain full flexibility to reallocate revenues from the pollution tax, its broader fiscal objectives should be spelled out. For example, does the government intend to maintain overall revenue neutrality, reducing distortionary taxes proportionately? If so, even without revenue earmarking, the government may find that other expenditures need to be cut if the pollution tax should, by raising the return to pollution control investments, cause a diversion of the fixed revenue pool away from competing uses. In the end, this effect could be offset by a rise in total revenue via the growth effects of a less distortionary tax structure (the so-called 'double dividend'; see Goulder, 1995). To trace through and quantify the indirect effects of a proposed new eco-tax measure with significant revenue implications clearly requires a general equilibrium analysis.

Localized experimentation. The government may decide that the introduction of a new policy measure like an emissions charge at the national level involves an unacceptable degree of political risk. In that event, local experimentation could be a less risky way of testing public reaction as well as determining the effectiveness of the instrument and identifying possible implementation problems. Given uncertainties about the elasticity of response to a new charge, some fine-tuning of charge levels is apt to be required, and it is apt to be less costly to do this locally than nationally.

[9] Sweden's NO_x emission charge works this way. The tax, levied on large power generators, is proportional to emissions while the rebate is proportional to electricity produced. Thus, a power plant whose emissions per unit of electricity generated are above average has an incentive to reduce its emissions intensity (OECD, 1996).

Encouraging local experimentation with new instruments could also induce a healthy competition between localities conducive to policy innovation. The more successful approaches could then be replicated by other localities facing similar environmental problems.

3.2 Policy reform and the development of institutional capacity

Many developing countries are currently engaged in ambitious programmes of structural reform that include, among other things, domestic market deregulation, privatization, trade liberalization, and the gradual phase-out of various price subsidies. Not only do these measures have potentially important environmental implications in and of themselves—e.g., the reduction of energy, water, and fertilizer subsidies—but they also alter the broad policy and institutional framework within which environmental policy is made and implemented. Put simply, the removal of price distortions and the promotion of more competitive market structures should improve the chances that EIs will yield their potential efficiency benefits. Also, as reforms make economies more open to foreign trade and investment, an environmental policy centred on EIs should provide stronger incentives than one centred on CAC for domestic industries to exploit new opportunities to acquire both cleaner and more efficient production technologies from abroad.

Most reform programmes involve a redefinition of government's role *vis-à-vis* economy and society, the net result of which is usually a sharper focus on a more limited range of tasks, including environmental and other regulatory functions. If weak capacity of the environmental regulatory body reflects generalized weakness of public institutions, this takes time to remedy. In the meantime, and within limits, the institutions of civil society (e.g., civic associations, non-governmental organizations) may be able to compensate through 'informal regulation'. To be effective, however, this approach depends on the public's timely access to accurate information and capacity to interpret and react to that information, neither of which can be taken for granted in many developing countries. Also, at the very least, government must have a tolerant attitude towards freedom of information, even if it does not play an active role in its generation and dissemination.

If a lack of resources is the main constraint on the environmental agency's effectiveness, then a narrowing of government's remit may liberate some resources for use in environmental management. At the same time, however, fiscal policy reforms normally involve shrinkage of government deficits, in which case an environmental programme that employs EIs could prove attractive insofar as it enables at least partial self-financing. This would not hold if the implementation of EIs turned out to be far more costly than the implementation of a regulatory scheme with broadly similar environmental benefits. There is as yet little empirical evidence on the relative implementation costs of different instruments in developing countries.

Phased implementation of EIs offers both public and private institutions, both the policy enforcer and those expected to change their behaviour, a grace period for learning and adjustment to the new rules. For example, a regulatory programme based on non-tradable discharge permits can be the first step on the way to a tradable permit scheme. If the former cannot be implemented effectively, it seems doubtful that the latter can. Similarly, a pollution charge scheme can be implemented in phases, with charges set initially at low levels and serving principally to raise revenue, then gradually escalated (preferably in accordance with a pre-announced schedule) to have a progressively stronger incentive effect.

4. Conclusion

The experience of both OECD countries and developing countries suggests that EIs have overwhelmingly been used as complements to rather than substitutes for regulatory instruments of environmental policy. This seems unlikely to change, as policy makers generally aim at multiple objectives for which a mix of instruments is usually more effective than any one alone. The informational and institutional demands of EIs can be as great as with regulations, though with both there are ways of reducing those demands—albeit at a cost. Considerations of political acceptability have conditioned both instrument design (e.g., grandfathering of tradable permits, non-compliance fees rather than simple pollution charges) and phasing of implementation (e.g., starting with local experimentation, setting low initial charge rates). With the advance of market-oriented economic reforms in the developing world, the policy and institutional environment should become more conducive to applying EIs; with greater political openness in many countries, the scope for involving the media, non-governmental organizations and the public at large in environmental enforcement (e.g., through information disclosure programmes) should also increase.

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