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Applying Economic Instruments in Developing Countries:
From Theory to Implementation

by

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ABSTRACT

A growing number of developing countries are experimenting with the application of economic instruments (EIs) to environmental management. As they do so, the developed country experience offers some guidance on what implementation problems to expect, but developing countries are also learning by doing. Whether in developed or in developing countries, EIs have generally played a supporting role to environmental regulations, not taken their place. The same weak monitoring and enforcement capacity which has hampered much environmental regulation in developing countries also reduces the effectiveness of economic instruments. Still, as in developed countries, probably the single greatest obstacle to wider use of EIs has been political. Economic instruments make explicit the "polluter pays principle" which had hitherto remained at best implicit. Leaving subsidies aside, pollution taxes and charges have been among the most widely used instruments. Usually, though, they take the form of non-compliance fees, which can be seen as a political expedient to mute industry protests over the "double burden" of charges on residual pollution. Earmarking of charge revenues for environmental expenditures has also been common. Effective monitoring remains a constraint on the application of emission-based charges, but environmental excise taxes and tax differentiation have proven easier to implement. Permit trading or auction schemes remain the exception in developing countries, as in developed countries. Singapore has been a pioneer of the permit auction as the United States has of permit trading. Interest is growing in developing countries, in part on cost grounds, in part because political opposition may be less than with taxes and charges, assuming broad consent to the initial permit allocation. Some degree of grandfathering of existing polluters has been the preferred means of buying such consent.

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 internalise environmental externalities and thereby make polluters pay. In other words, these instruments "leave decentralised 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 virtually 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 referred to as "command-and-control" (CAC), has been criticised by economists on grounds of both static and dynamic inefficiency: the former because it requires compliance with the same standards by all pollution sources, irrespective of marginal compliance costs; the latter because it provides little incentive to technical improvement once compliance has been achieved.

For the policy maker, the economic efficiency of a particular policy instrument is only one selection criterion, however important. Once other relevant criteria are introduced, then trade-offs must be considered, and EIs are no longer necessarily the preferred candidate. For instance, if minimising exposure to a highly toxic substance is the policy objective, strict CAC measures -- including perhaps an outright ban on production/sale of the substance -- may be preferred over a policy which would discourage use through a seemingly prohibitive product tax. Another consideration is that what is optimal in theory may be of limited relevance if there is little possibility of implementing the optimal policy.

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 OECD countries as to developing countries. Still, the literature dealing with problems of implementation and how to overcome those problems is sparse. One of the first and most comprehensive explorations of implementation strategies for environmental taxes in OECD countries is OECD (1996). For developing countries, where experience with such instruments is admittedly more limited, there has as yet been virtually no comparable work[1]. This paper is intended to begin filling this gap.

Problems in implementation of environmental policy often reflect deficiencies in instrument design. Those deficiencies can be of a few sorts. One occurs when the instrument is overly complex, requiring sophisticated technical capabilities on the part of those charged with its implementation. Another occurs when instrument design is insensitive to the political dimension of successful implementation -- i.e., the impact on and likely reaction of powerful interest groups. Implementation problems may also arise from a weak legal and institutional framework. For instance, certain instruments presuppose a clear assignment and enforcement of property rights; others require an effective tax administration; still others may rely on a well-

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

functioning financial system. These legal/institutional prerequisites do not always exist in developing countries. Thus, instrument selection needs to be informed by an assessment of the institutional or other parameters which affect implementation. Another sort of implementation problem occurs when there is a mismatch between the instrument chosen and the nature of the environmental problem to be addressed: 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 prohibitively expensive if not impossible.

The set of instruments available to policy makers is an open one. Seldom is environmental policy choice dichotomous: either an emission standard or an emission tax. Various combinations of instruments are possible and very often regulatory measures are combined with economic incentives in a way which seeks to exploit the advantages of each. The flexibility of instrument design and the possibility of creating "hybrid" instruments are distinct advantages where policy makers aim to achieve a balance among competing objectives -- e.g., efficiency, effectiveness and equity (see O'Connor and Turnham 1992).

Besides CAC and EIs, a third type of instrument should be mentioned. Suasive instruments (SIs) rely on voluntary compliance by polluters, motivated either by the threat of adverse or the prospective of favourable publicity. Environmental education and awareness raising are key elements of any policy designed around SIs. If people are willing to act on their preferences for a clean environment -- e.g., by choosing to buy products with an "eco-label" or boycotting the products of firms known as serious polluters -- then this may be enough inducement for firms to improve environmental performance. Formalising a commitment to reduce pollution in a voluntary agreement (whether with government, with a non-governmental organisation, or with a citizens' association) may reinforce the credibility of that commitment. If SIs have certain advantages over other types of instrument -- particularly in the modest demands they place on government enforcement -- they also have drawbacks, viz.., they can involve high transactions costs when many small polluters are involved and are wholly unsuitable for certain types of pollution (e.g., non-point source) (see O'Connor 1994, pp.134-135).

This paper proceeds as follows: in the next section we briefly review the experience with the implementation of economic instruments in both OECD countries and developing countries. The types of instruments considered are: pollution taxes/charges[2], product taxes/charges, deposit-refund schemes, and tradable permits. Also, the related practices of earmarking revenues from EIs and creating specialised environmental funds are considered. In section 3, a general framework is presented for evaluating alternative instruments in terms of their ease of implementation, and in the final section some suggestions are advanced for addressing the more common implementation problems. The question of how environmental policy affects competitiveness in an open economy is also briefly considered.

[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 frequently enjoy some discretion to levy charges for specific purposes.

2. *Experience with Applying Economic Instruments*

Actual practice in applying EIs departs in virtually all cases from pure theory, reflecting the need to accommodate instruments to the complexities of the real world. In some cases this means mixing elements of CAC, EIs and/or SIs in order to realise multiple objectives; in other cases it means modifying the design or phasing the introduction of a particular instrument to make it more politically acceptable. As a rule, environmental policy makers operate in a "second-best" world.

This review of the experience with 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 such evaluations on which to report (with the exception of some cost assessments of tradable permit schemes in the United States, cited in Hahn 1989). In future, as experience with EIs accumulates, there will clearly be need of further research into their actual costs and other characteristics in comparison with the relevant alternatives.

2.1. *The OECD Experience*

The experience of OECD countries with applying EIs has been studied more extensively than that of developing countries. In reviews of that experience, one recurrent question has been what are the conditions for (obstacles to) effective implementation of specific EIs. Since some of the same problems with implementation encountered by OECD countries could be expected to recur in developing countries should the latter adopt similar instruments, it is worth considering how OECD countries have addressed those problems.

It is noteworthy that, even 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 characterised 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 more important 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;

-- Also, 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, 8 of which are found in the United States, and 8 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 command-and-control approach with a purely economic one. "Economic instruments are complements mostly and substitutes only sometimes for other types of approaches" (OECD, 1994b, p.187).

Pollution charges and charge-rebate schemes

Implementation problems with pollution charges are of two sorts: political and technical. The political problem can arise from three sources: general opposition from taxpayers to tax increases; opposition from the polluters, whose costs would normally be increased; opposition when the charge is perceived to be inequitably distributed -- e.g., if low-income households pay proportionally more. The first source of political opposition (and to a degree the second) can be addressed through designing a tax that maintains revenue neutrality.

Revenue neutrality. In its broader connotation (i.e., substituting eco-taxes for distortionary ones while holding total revenue constant), revenue neutrality may be a way of reducing taxpayer-voter opposition to a new charge. Another sense of revenue neutrality is to redistribute the revenue from the tax (levied, e.g., on emissions) to all those paying the tax, but in proportion to their production (Godard 1994). In this way, the externality is internalised in a way that transfers revenue from more- to less-polluting enterprises but without redistributing revenue away from the polluting industry per se. This approach is used by Sweden, for example, in the case of its nitrous oxide (NO_x) emission charge on large heat and power generators. The tax is proportional to emissions while the rebate is proportional to energy or electricity produced. Thus, a power plant whose emissions per unit of electricity generated are above average has an incentive to reduce its emissions factor. The scheme is revenue neutral with respect to the affected industry as a whole so, in theory, it should generate little political opposition.

Addressing equity concerns. In OECD countries that have experimented with eco-tax measures, distributional impacts have been a major political issue. Thus, policy packages have often included an explicit compensation mechanism aimed at adversely affected groups (OECD, 1996, chap.4). In analysing distributional impacts of a pollution tax/charge, the following need to be considered: (i) its effective incidence (who ultimately pays the tax?); (ii) the relevant baseline for comparison (the *status quo* or an alternative instrument to achieve the same environmental goal?); (iii) the dimension of distribution that is of particular interest -- e.g., across income groups, across types of firm/industry, across regions, etc.; (iv) what is to be done with the additional revenue from the tax?

Much of the empirical evidence on the distributional impact of environmental taxes in OECD countries relates to carbon taxes (see Poterba 1991 for the USA, Pearson and Smith 1991 for the UK, and Scott 1991 for Ireland); studies of impacts on different income/expenditure groups suggest varying degrees of regressivity. While the results from these studies cannot be assumed to hold for developing countries, it is plausible that the regressivity of a carbon tax would be even greater in developing countries. This could be the case, for example, where the poor rely for cooking and heating on coal while the wealthy prefer gas and oil (with their lower CO_2 emission factors)[3].

Carbon taxes are probably not on the near-term political agenda of most developing countries, so these studies are of limited relevance. Other types of environmental taxes in OECD countries have been studied less, but there has been some research on the distributional effects of motor fuel taxes (cff. Poterba, 1989, for the USA; Johnson, McKay and Smith, 1990, for the UK). Opposing tendencies appear for the two countries -- regressive incidence for the USA and progressive for the UK -- which probably reflects in part the relative availability of substitutes to private motorised transport.

Where the burden of an eco-tax falls unevenly on different income or other social groups, the question of whether and how to compensate the losers arises. The likelihood that compensation will be paid would appear to be greatest where the impact is sizeable and clearly traceable to that policy, and where the adversely affected group is readily identifiable and relatively cohesive. A pollution tax with small distributional effects which are in any case difficult to detect and widely diffused may not generate strong political resistance. Political considerations aside, on normative grounds governments may wish to offset the regressive incidence on low-income groups of a pollution tax (e.g., the carbon tax mentioned above) through some form of compensation. Two possible forms of direct compensation would be (i) an equivalent reduction in the regressive incidence of another tax (or combination of taxes), and (ii) a direct transfer payment to low-income households.

Besides compensation, another means of addressing the distributional impact of a pollution tax would be to grant certain exemptions. This is more appropriate for industries which can demonstrate special hardship -- in the case of a carbon tax, for example, energy-intensive industries that compete in world markets. In the case of Sweden's carbon tax, the government chose to put a cap on tax payments to ease the burden on certain industries (OECD 1994c). Several objections have been raised to such exemptions however: (i) they constitute an effective subsidy of energy- and hence pollution-intensive industries; (ii) for a given national emissions target, a subsidy on certain sectors would require a higher tax on others; (iii) the measure could evolve into a more-or-less permanent form of protection for the exempted sectors (Pearson and Smith 1991).

The virtue of simplicity. Very often, governments devising charge schemes must be prepared to sacrifice technical elegance in the interests of ease of implementation. Take the example of solid waste. All but three OECD countries have instituted waste collection charges for municipal solid waste. Generally the charge for households is levied on a flat rate while

[3] If the carbon tax were also levied on motor fuel, then the consumption patterns of different income groups would also need to be considered; it seems likely that in most low income countries poor households are less dependent on motorised transport than in OECD countries.

industry is often charged by quantity as well. On environmental and economic grounds, a charge should also take into account type/characteristics of the waste, but such differentiation would be exceedingly costly to apply at the level of the individual waste generator. Some countries grant rebates for waste properly disposed of by the generator. In Denmark, this rebate scheme has at times caused debate about what constitutes waste, suggesting the need for clearly defined eligibility criteria.

The United Kingdom (UK) government has recently advanced a proposal to levy a tax on all landfilled waste (see *Financial Times*, 17 May 1995, p.8). The environmental objective is to promote incineration, recycling and waste minimisation; the fiscal objective is to maintain revenue neutrality by lowering social insurance taxes by an equivalent amount. Within a decade the target is a 10-per cent reduction in the proportion of waste which is dumped in landfills (currently around 70 per cent); a further 10-per cent reduction is targeted for the second decade of the tax. While some have argued that a general waste tax would be more effective in minimising waste, the advantage of the landfill tax is the ease of administration, since it can be collected at a small number of well-identified sites. Questions have also been raised about the assumed environmental superiority of incineration over landfilling and fears have been voiced by local authorities over a possible increase in illegal dumping. Objections have also been raised to the choice of tax base: the tax is to be calculated as a percentage of the waste disposal charge rather than on the basis of weight. Such an *ad valorem* tax, it is argued, would penalise most the users of the better-engineered (hence more expensive) landfills, diverting waste to cheaper sites with fewer environmental safeguards (e.g., to stop leaching or trap gas). A weight-based tax would make sense, however, only if the environmental damage caused by the waste were a function of weight alone, which it clearly is not. Administratively, the *ad valorem* charge is simpler to implement since records on disposal costs already exist; to charge on weight would require the installation of scales (or weighbridges) at all landfill sites.

Gradual escalation. Experience suggests that charges are more likely to gain political acceptance if they are raised gradually from low levels, one reason being that existing polluters are allowed more time to adjust to the charge. Alternatively, if the government decides to initiate the tax at a high rate, then pre-announcement of the measure should enable existing polluters time to adjust their production processes or take other measures to reduce their future tax liabilities. Also, to avoid sudden large changes in tax rates that reduce the predictability of the investment climate, the government may wish to start with low charges and then gradually raise them towards the "optimal" level (i.e., the one where the environmental goal is reached). Finally, polluters may be reluctant to accept higher charges until they are convinced that the charge revenues are being put to effective use to raise environmental quality.

Transparency in revenue use: a limited role for earmarking. The acceptability of a pollution charge may depend on the public perception that the revenues are being reinvested in environmental improvement or, alternatively, that the charge is not being levied simply for general revenue-raising purposes. For instance, in Denmark, the allocation of revenue from a charge on milk cartons to the general budget was opposed when it appeared that the charge did not have the desired incentive effect. Earmarking -- or deciding in advance that some or all of the revenue from an eco-tax will be used for certain environmental purposes -- may increase the tax's political acceptability. If a pollution tax is set at a high enough level to achieve the environmental quality goal, then earmarking revenues for pollution reduction measures would not be justified. In practice, however, governments seldom set pollution taxes high enough to have the desired incentive effects on their own, in which case earmarking can aid in achieving the desired environmental outcome.

The strength of earmarking is also its weakness. By setting aside some amount of revenue in a separate fund for a specific use, it thereby insulates those resources from competition among alternative uses. Where environmental investments have traditionally been underfinanced because of their low estimated rates of return (based on standard cost-benefit analysis), such earmarking may be a useful interim measure. The risk is that the earmarked revenue would continue to be spent on environmental projects even though rates of return based on an extended cost-benefit analysis (i.e., factoring in environmental costs and benefits) would not justify the investments. Also, earmarking of revenues for environment might set a precedent for other types of expenditures.

A special case of earmarking is the levying of user charges for certain environmental services. In this case, the charge collected is considered payment for a service rendered, and the revenue is used to cover the costs of provision. If the revenue collected exceeds the amount necessary to provide an adequate level of service, then the issue arises once more of whether that additional revenue should be earmarked. The water quality management programmes in three European countries -- France, Germany and the Netherlands -- employ systems of effluent charges where the charge revenues are used to finance investments in waste water treatment and other water quality improvements. In the case of the Netherlands, the charge rate has risen steeply over time. As a result, the charge has gone from being principally a cost-recovery device for water treatment to being an incentive device to lower discharges. At the same time, since the revenue is earmarked, and pollution loads have fallen with some lag, the result has been overinvestment in treatment capacity: it is estimated that public treatment plants have surplus capacity of about 20 per cent (OECD 1996).

Product taxes and tax-subsidy schemes

Product taxes are indirect instruments levied on products associated with pollution rather than on pollutants per se. This makes them easier to administer, since no metering of actual discharges is required. Moreover, governments usually have an administrative system already in place for the collection of excise taxes, so the institutional demands of a product tax are modest. Also, with an excise tax, there is greater scope for choice than with an effluent/emission tax of where to levy the tax. By definition, the latter should be levied at source. With an excise tax, however, it is possible to levy it at different points in the production and distribution chain. The initial incidence of the tax -- which depends on where it is levied -- may be very different from its effective incidence -- which depends on relevant supply, demand and substitution elasticities. What matters in choosing where to levy the tax is how easy it is to collect. In general, the fewer the taxpayers the easier collection should be. Thus, with the US tax on chlorofluorcarboms (CFCs), it was decided to levy the tax on CFC producers and importers, who are far fewer in number than CFC users (Barthold 1994).

The most common sorts of environmentally-motivated product taxes/charges in OECD countries apply to fertiliser, lubricant oil, CFCs, and product packaging (OECD 1994). Other products sometimes subject to an environmental tax are batteries, tires, disposable razors and disposable cameras. Product taxes can be combined with subsidies or rebates. Alternatively, revenues from an environmental excise tax can be placed in an earmarked fund to finance clean-up. Germany has long had a tax-subsidy scheme for oil products. A tax on lubrication oil is used to finance a subsidy paid to firms which accept and dispose of waste oil properly, with the subsidy based on the average cost of disposal. Judging from the high percentage of waste oil that is properly disposed of, the scheme appears to have worked effectively. Besides its CFC tax, since 1980 the United States has levied an excise tax on hazardous chemicals to fund the Environmental Protection Agency's hazardous waste site clean-up programme (known as Superfund). Also, since the 1989 Exxon Valdez oil spill, it has levied an additional excise tax on petroleum and petroleum products to fund the Oil Spill Liability Trust Fund (Barthold 1994).

For the most part, environmental excise taxes have been employed either as user fees (e.g., to cover the costs of treatment and disposal after use), as recycling incentives, or as insurance premia in a mandated risk pooling scheme like the oil spill fund. Much less frequently are they employed as Pigovian taxes designed to internalise the costs of environmental damage in prices. For that, the amount of the tax would need to be linked to some estimate of the environmental damage caused by the product's use or improper disposal. A few examples do exist: in the United States, the amount of the CFC tax is a function of the particular product's ozone-depletion potential (a rough measure of the degree of environmental damage it causes); in Sweden, a tax on diesel fuel is differentiated by environmental characteristics (sulphur, aromatics, etc.), with a rebate given on "environmental diesel" (OECD 1994b).

Deposit-refund schemes

Deposit-refund schemes may be instituted voluntarily by industry or mandated by government, though most are now mandatory. While deposit-refund could potentially be applied to a number of products to encourage recovery/recycling, they are still mainly applied to glass, plastic and metal beverage containers. One important consideration in deposit-refund schemes is the share of the deposit fee in the total product cost. Experience suggests that for beverages (e.g., beer and soft drinks) where packaging costs are a relatively high share of total price, return rates tend to be high; for wine and liquor, the deposit rate is relatively low in relation to product cost and return rates are correspondingly low.

Another consideration is the relevant substitution options for the product(s) covered by the deposit-refund scheme. For instance, a deposit fee on new returnable bottles may encourage use of recycled bottles, but it may also divert supply to one-way containers made of materials not subject to any charge or deposit-refund scheme. In such case, it would be advisable to combine the deposit fee with a product charge on one-way packaging containers.

Still another issue in implementing a deposit-refund scheme is compensation to retailers for the handling costs of returned packaging. Those costs can be substantial, and a system not ensuring handling cost recovery is not likely to generate strong support from retailers. Where a separate collection network needs to be organised for the recovery of recyclable materials, the extent of coverage (and thus the convenience of use) of the network is an important factor conditioning recovery rates, and the costs of establishing and maintaining the network need to be considered when setting charge rates.

Tradable permits

Permit trading schemes are quantity-based instruments, which limit overall levels of pollutants in a defined pollution-control area. Thus, their environmental outcome is more certain than for price-based instruments like charges and taxes. Public opposition to such schemes may also be greater, however, if they are perceived by environmental groups and/or the public at large as giving industry a license to pollute. Thus, in the case of the United States, which has pioneered use of this type of instrument, reference is made to emission credits rather than pollution permits and an attempt is made to de-emphasise the explicit nature of the property right (Hahn 1989). Industry support, on the other hand, could be greater for a permit trading scheme than a pollution charge, if for example the government chooses to allocate a portion of permits to existing firms free of charge or for a nominal fee. In this way, some of those firms could stand to profit from subsequent permit trades, depending on how effectively they can reduce their own emissions[4]. Thus, "grandfathering" of at least a portion of the property rights (or permits) has been a standard feature of permit trading schemes in the United States (including those for SO_2 and NO_x emissions under the 1990 Clean Air Act).

The US SO_2 emission trading programme has been underway for three years, long enough to make a preliminary evaluation of its effectiveness. While the bulk of SO_2 permits are allocated to power companies, a small number are auctioned once a year by the Chicago Board of Trade to set a price for them. According to one such evaluation (see *Financial Times,* 6 May 1996, p.15), the programme has far exceeded expectations in achieving rapid emissions reductions. In 1995, the 110 most polluting power plants emitted only 5.3 million tonnes of SO_2 , well below the government ceiling of 8.7 million tonnes. Arguably, in terms of environmental effectiveness, such a quantity-based system is far superior to a tax on energy consumers[5]. The ultimate objective is to cut emissions to half their 1980 level by 2010. By one USEPA estimate, the cost to industry so far of the emission reductions has been US$2.5 billion, or roughly half what it would have cost under a traditional regulatory programme. One indication of the effectiveness of the programme is that the price of a permit has been plummeting, from US$132 per tonne in early 1995 to US$68 in early 1996; when trading was launched in 1993, the expected permit price was $1500.

Permit trading schemes should incorporate clearly defined, time-specific emission ceilings, so that polluters are not left in doubt about which emission reductions qualify for emission credits and which do not. Monitoring, record-keeping and reporting are key elements of a permit trading scheme, and a reliable database on initial emissions is needed to get the scheme started.

[4] The alternative would be to auction off the permits to the highest bidder(s), in which case the government would appropriate any rents created by the induced scarcity of supply of "rights to pollute".

[5] The Chief Executive of the UK Association of Electricity Producers is quoted as saying: "The usual cry is for taxes on customers to discourage them from using electricity. That is like trying to control the problem from the end of a very long rope and there is no guarantee that it will exert the necessary pull".

2.2. *Developing Country Experience with Applying EIs*

There may be factors specific to developing countries which give rise to policy implementation problems not commonly encountered in OECD countries. A shortage of skilled personnel is perhaps the most common handicap facing developing countries. While this is a general problem, it could have more serious implications for a policy built around CAC measures (where government officials should have detailed technical knowledge of the industries to be regulated) than for one relying on EIs (though the importance of detailed technical knowledge to designing effective EIs should not be underestimated). On the other hand, it is possible that other factors enhance the potential for applying EIs in developing countries. For example, it has been observed that EIs are more likely to be accepted where there are not entrenched bureaucracies committed to defending the prerogatives they have come to enjoy under a CAC system. This advantage would clearly apply only if the developing country is just beginning to establish an environmental management system. Also, it may be that pollution charges that affect industries differentially and thus induce structural adjustments can be accommodated more readily in a dynamic economic environment already characterised by rapid structural transformation. This section reviews several examples of EI applications in developing countries, wherever possible elaborating on implementation experience and problems. Similarities and differences with the experience in OECD countries are noted.

As yet, there has been no survey of the use of EIs in developing countries comparable to those done for OECD countries (see OECD 1989 and 1994b). Nevertheless, based on a review of the literature on environmental policy in the newly industrialising economies (NIEs) of Asia and Latin America (cf. 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 more broadly has been growing in recent years. Thus far, the most frequent applications have been similar to those in OECD countries -- i.e., emission charges seem to predominate, with product charges also fairly common. Frequently, the emission charges in the NIEs take the form of non-compliance fees rather than straight charges on all emissions. Deposit-refund schemes are beginning to operate in the higher income NIEs, while in very few instances are tradable permit schemes operational (Singapore being the outstanding example of this approach thus far). 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. Also, some NIEs have been experimenting with the use of voluntary agreements and other suasive instruments, which are also attracting wider interest in OECD countries (though a few countries like Japan and the Netherlands have relied on them for quite some time).

Emission/effluent charges

Since the early 1980s, Korea has had an emission charge system (applicable to both air and water) which operates as a non-compliance fee, thus combining elements of CAC and EIs. Since 1983, the Environment Administration (now the Ministry of Environment) has had the authority to impose an emission charge on facilities exceeding mandated emission standards. Initially, the charge could be levied only if the polluter should continue 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. The charge is based only on pollutant concentrations. It varies with the location of the facility, the duration of excess discharges and the number of previous violations. The charge rate has historically been set rather low, in some instances falling below the operating costs of a pollution treatment facility (Chung, 1991), with the result that polluters have been known not to operate their treatment plants at the risk of detection. A familiar problem with non-compliance fees is that, while if set at proper levels they discourage violation of standards, they do not reward overattainment. Moreover, the use of concentration alone as the basis for the fee can encourage dilution without any reduction in total pollution load. Thus, there has been discussion in Korea about the merits of shifting to a straight emission charge that would tax all emissions, not just those above standard, and would combine concentration with quantity in the charge formula (Rhee 1994).

In Malaysia, in the mid-1970s, the government 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 waste discharged. 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 (see Vincent 1993). The system was built on effluent standards which were phased in over four years, starting from 5000 parts per million (ppm) -- which is 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. In practice, the licensing fee consisted of two parts: a flat administrative fee and a variable effluent-related fee. For releases into a watercourse, the fee was set at M$10/tonne of BOD load discharged. The government continued to wield the threat of revoking a mill's operating license if it did not comply with the standard, though this was not done in the first year in order to allow firms a grace period for adjustment to the new policy. In effect, a firm had the option of paying an excess license fee of M$100/tonne for discharges above the 5000 ppm standard. Also, the 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, with the threat of license cancellation now operative. 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 one 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 fixed effluent-related license fee was gradually eroded in real terms by inflation. When tested, the government threat to close down delinquent mills proved credible. 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. The costs, however, were mostly shifted onto palm-oil growers, since they could not be passed on to consumers in a highly competitive world market whereas individual mills exerted considerable market power over neighbouring growers.

An effluent charge system for total suspended solids (TSS) from the coal processing industries of Bihar, India, is proposed in Kumar and Sherif (1995). The authors note that the present effluent standard (100 mg/litre) is widely violated because of firms' high abatement costs and government's weak monitoring and enforcement capability. While detailed firm-level cost data are not available, the authors estimate marginal abatement cost functions based on information regarding each firm's volume of effluent discharge, initial pollution load and size of particles. Those firms with an initially low pollution load, other things equal, have relatively high marginal costs of further reducing the load. Also, those firms whose effluent has large average particle size have higher marginal abatement costs, since they must undertake an additional processing step (flocculation). The total abatement costs of universal compliance with the uniform effluent standard are compared with those of two alternatives: (i) a uniform effluent charge levied on effluent concentration in waste water such that total abatement is the same as that associated with the enforcement of a common standard; (ii) a two-tiered effluent charge, with a higher rate on those firms whose waste water has TSS concentrations above 150 mg/litre and a nominal charge on the rest[6]. Both (i) and (ii) yield significant abatement cost savings relative to the strict regulatory approach, while (i) generates somewhat larger revenues for government. The authors note that, by ploughing the charge revenues back into upgrading monitoring and enforcement, the government can deal with a central practical weakness of the CAC approach, viz., the limited institutional capacity at local level to implement the policy effectively.

China has had a system of pollution charges, most assessed as non-compliance fees, for the last fifteen years (Potier 1995; NEPA, 1992). The charge is levied on both the quantity and the concentration of discharges. 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 nation-wide. The system is backstopped by a fourfold system of penalties for serious violations of standards. In principle, the charges are to be set at a level slightly above the average operating costs (including a depreciation factor) of pollution control facilities, to encourage broad compliance with standards. The charges are assessed on air and waste water discharges as well as on noise, solid waste and radioactive wastes[7]; they cover a range of parameters, with higher charges levied on more toxic pollutants (NEPA, 1992, p.11).

During the third phase, which began in 1988 and continues to the present, the emphasis has been on reforming the system for allocation and use of charge revenues. Until that year, revenues were largely allocated as grants to subsidise pollution control measures, but since then there has been a shift towards greater reliance on loans. The revenues are deposited in an earmarked local environmental fund managed by a designated bank. Roughly 80 per cent of those revenues are then lent or given to enterprises for pollution control investments, with the remainder going to environmental agencies to finance the capital and operating costs of the charge programme -- 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.

Since charges are relatively low and have been eroded in real terms by inflation, they provide relatively weak incentives on their own to reduce pollution. Also, at the local level, environmental officials are amenable to influence by industrialists intent on avoiding the charge. Perhaps one of the greatest weaknesses of the scheme is that state enterprises are permitted to pass on the charge costs in higher prices or entitled to a tax rebate to offset the charge.

[6] The rationale for (ii) is that the uniform charge falls most heavily on those firms which are already closest to complying with the effluent standard and whose marginal abatement costs, therefore, are most likely to exceed the charge.

[7] Waste water and air emissions are the two main sources of charge revenue: they accounted for around 50 per cent and 24 per cent, respectively, of cumulative charge revenue between 1986 and 1994; NEPA 1992 (N.B. The data in the Tables have been updated to 1994 by hand.)

Among the improvements being considered is a gradual increase in the charge rate, which is strongly opposed by industry. Also, the practice of extending loans from the environment funds only to those enterprises that have paid charges is to be phased out.

In a local experiment begun in 1992, the city of Chongqing in Sichuan province introduced an SO_2 tax which, because of the difficulty of measuring actual emissions, was levied on the amount and sulphur content of the coal burned by each industrial enterprise. At the same time, another city, Yichang, in Hubei province, introduced a system of SO_2 discharge permits, with charges levied on emissions exceeding the permitted level. In the initial phase, there was gross underreporting of emissions by factories, estimated through plant checks and tallies of water and coal consumption at three-fourths of total emissions. Consequently, the charge rate was substantially increased; whether or not as a direct result, atmospheric SO_2 concentrations have fallen significantly. Assuming the problem of monitoring can be resolved, it would appear a relatively small step from an SO_2 permit system to the introduction of permit trading.

Thailand has been studying the feasibility of introducing a waste water and sewage charge for the Bangkok Metropolitan Area (BMA) (see Phantumvanit *et al.* 1994). The charge would differentiate between households and small industries on the one hand and large industries on the other, with the former paying a surcharge on their water bills based on metered water consumption and the latter being charged according to a formula that includes both the flow volume and the concentration of BOD discharges in waste water. The charge on BOD is structured as a non-compliance fee, levied only on discharges exceeding standards. Limiting the pollution-linked charge only to large polluters (defined as those discharging on average 5 kg. or more of BOD per day) reduces the monitoring burden while adding the waste water charge onto households' water bills simplifies collection and administration. Nevertheless, there are certain problems which can be anticipated. First, the general public is not accustomed to the notion of paying a waste water (or sewage) treatment fee. Assuming that the additional charge does not pose a sizeable financial burden to the average urban household, the general reaction to the charge may be muted. There may still be an equity problem, however, if the charge on poor households' water consumption represents a disproportionate share of their income. This could be the case even if their average consumption is roughly equal to that of rich households; in practice, it might even be higher since water consumption is closely correlated with family size, and average poor household size exceeds that of wealthy ones. Another, perhaps even greater problem for implementation is how heavily polluting industries might react. If the fee adds significantly to their costs (as it might for some textile firms, for example), they could be expected to oppose it vigorously. If, on the other hand, the fee is set so low as to have no noticeable effect on their operations, its incentive value would be nil, though it could still be useful as a cost recovery device for common waste water treatment. The proposed charge scheme in Thailand is analogous, then, to the user charges applied in France and Germany in the sense that its principal function would be to recoup the operating (and part of the investment) costs of municipal waste water treatment facilities.

Product taxes and tax differentiation

An emission charge system can be costly to administer, especially if it relies on monitoring actual emissions rather than on standard industry emission factors (or coefficients). Moreover, in the case of air pollution, a system of emission charges based on actual emissions monitoring is still not practical for mobile sources. In principle, such a system could be designed (see Eskeland 1994), using a presumptive charge based on a standard emission factor for each model and year[8] (e.g., grammes emitted per litre or per kilometre) and multiplied by the distance travelled since the previous mileage reading. A partial rebate could be made if the periodic vehicle test reveals actual emissions well below the emission factor. However, the heavy administrative and technical requirements of such a system would make it extremely costly if not impossible to implement. Thus, Eskeland proposes a combination of a regulatory instrument (an abatement requirement) and an indirect EI (a product tax on gasoline) to mimic the effects of an emissions charge. An emission charge achieves two results simultaneously: discouraging the polluting activity and providing incentives to make that activity cleaner. Mandated abatement requirements achieve the second objective, while the gasoline tax accomplishes the first. One problem is that the demand for automobile use appears in many countries to be rather inelastic with respect to the price of gasoline, so the tax might have to be quite high to have a noticeable effect on kilometres travelled.

Another approach to addressing air pollution from mobile sources -- one applied widely in OECD countries but also in Taiwan and Thailand -- is tax differentiation between leaded and unleaded gasoline. 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, subsidising 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 tax differentiation and regulation, the market share of unleaded gasoline has risen steeply in recent years (to roughly half the market). In Taiwan, a similar price differentiation strategy has been used with similar 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[9] (O'Connor 1994). Monitoring reports by the Taiwanese EPA indicate that the average lead content in ambient air in Taipei decreased from 0.46 g/m in 1989 to 0.18 g/m in 1992 (Pan 1994).

A differential tax on leaded gasoline needs to be reviewed regularly since, as demand for and production of unleaded gasoline increases, at some point the relative price of the two should reverse, weakening the rationale for the tax differentiation.

Environmental funds

A number of developing countries and economies in transition have established environmental funds to finance certain environmental expenditures. The pros and cons of earmarked funds were discussed above for OECD countries, and the same considerations apply for other countries. Such funds are best viewed as transitional mechanisms to mobilise financing to address an accumulated backlog of environmental problems. They can also play a

[8]Actual emissions could still be expected to vary considerably within a given model/year class due to differences in maintenance, driving conditions, etc.

[9] One possible drawback with standards -- whether technical or emission standards -- that apply only to new vehicles is that they may provide an incentive to delay the scrapping of older (and generally more polluting) vehicles.

useful role where capital for environmental investments cannot be raised through established financial institutions.

There are several possible ways of financing such funds: a contribution from the general government budget; a donor contribution; revenue from pollution charges; or some combination. In the event that charge revenue is used and that such charges are levied as non-compliance fees, this can provide the wrong incentives. In effect, if those responsible for fee collection are the same as those responsible for fund management, they would have an incentive to encourage continued non-compliance. Otherwise, their source of funds would dry up. This incentive problem is at least partially addressed in the Korean scheme. The Korean environmental fund, established in 1983; is financed one-third from a non-compliance fee with the remainder coming from a government budgetary allocation and from interest income. 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. Thailand also established an environment fund, in 1991, with the entire initial capital of US$200 million contributed by the government (largely through a transfer from the oil price stabilisation fund). Subsequently, a foreign donor made an additional contribution. In both the Korean and Thai cases, the funds are used primarily as a source of cheap credit for firms investing in pollution control technologies and for operators of public environmental facilities like sewage treatment plants. The Thai fund, however, is under-utilised, apparently because of the rather strict and time-consuming procedures for loan application.

Deposit-refund schemes

These have been implemented in a few higher income developing countries, notably Korea and Taiwan. In Korea, the Ministry of Environment 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 bottles ranges from 3 to 7 won (roughly 0.5 to 1 US cent), depending on volume. Rhee (1994) argues that a reason for the low deposit rates (as well as low emission charge rates) is the government's concern to keep inflation low. The strong influence of industrial interests on government policy seems as likely an explanation. The result is that there is little incentive for waste recovery, and refunds claimed represent only a tiny fraction of deposits collected.

In Taiwan, the deposit-refund scheme has been more effective in practice. Since 1988, Taiwan has been implementing a recovery/recycling system for several types of solid waste, including PET (polyethylene terephthalate) 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 Environmental Protection Administration (EPA) set a target recycling rate of 50 percent for the first two years (1989-90) and 60 percent by the fourth year. As of the third year, the accomplished recycling rate had only reached 41 percent, but by the fourth year (1992) it jumped to almost 80 per cent (Pan 1994), comparing favourably with rates in OECD countries[10].

In short, the deposit rate in Taiwan appears to have been set at a level yielding strong incentives for 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 may have provided valuable lessons when the government begins to extend it to other products.

Tradable permits

To date, there has been little experimentation with tradable permits in developing countries. Permit trading schemes depend on the creation and enforcement of new sorts of property right. Where more familiar types of property right are only vaguely defined and/or weakly enforced, it may be premature to think of introducing new ones. On the other hand, the growing experience with permit trading in the USA has begun to stimulate interest in developing countries.

Among non-OECD countries, Singapore has been a pioneer in this approach to environmental policy, using competitive tenders for permits to import and use ozone-depleting substances (specifically, CFCs regulated under the Montreal Protocol) and for permits to own an automobile. In the case of CFCs, each quarter the national consumption quota (as defined under the MP) is allocated among importers and users, half on the basis of historic consumption ("grandfathering") and half through a sealed-bid tender. Importers and users must register to participate in the bidding process, in which each firm specifies the volume of CFCs it would like to purchase and its offer price. Bids are then ranked by price and the lowest winning bid price (i.e., the one which clears the market) serves as the quota price for the full allotment of CFCs, including the pro-rated half. There was a steep increase in quota prices during the first few rounds of bidding (caused in part by stockpiling). With the price hike, firms faced a strong incentive to adopt conservation measures and substitute technologies, which has resulted in a swift and sizeable reduction in CFC demand. The auction procedure has enabled the government to appropriate a sizeable share of the quota rents, which it has used to subsidise recycling services and the diffusion of information on alternative technologies (O'Connor 1991). Since the decline in demand depressed the quota price, the government accelerated the phase-out schedule in an effort to support the quota price and maintain the incentive to continued demand reductions.

Singapore has also devised a vehicle quota system designed to increase the price of ownership of a motor vehicle. 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 price of owning a car in Singapore is 4.5 to 5 times

[10]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 1994. 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.

the landed vehicle cost. 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 (i.e., grandfathered in), 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. Bids are ranked from highest to lowest; each successful bidder pays a quota premium 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 which is set as the 12-month moving average price of the COE in that vehicle category. By mid-1992, the quota 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 (Fan, Menon, and Olszewski, 1992).

The effective functioning of the CFC quota auction depends on a sufficiently large number of bidders to provide adequate insurance against collusive bidding. Another condition that is clearly met is that the benefits from reducing consumption are independent of who makes the reductions (i.e., a unit of emissions causes the same amount of damage wherever it occurs). It is doubtful whether the vehicle ownership entitlement scheme would work as well in most other countries as in Singapore, since few other countries have the same geographic concentration of motor vehicle use. In Thailand, for example, a vehicle sold to a Bangkok driver has a very different effect on congestion and pollution from one sold to a driver in rural Nakon Ratchasima or in Khon Kaen.

Suasive instruments: voluntary agreements and community "right to know"

If the concern of policy makers is with speedy implementation of control measures at minimum administrative cost, then certain suasive instruments (SIs) should be considered, notably voluntary pollution control agreements. There is a long history of the use of such agreements as an instrument of local environmental policy in Japan (see Haga and Yano 1992 and O'Connor 1994). Among developing countries, Indonesia has one of the richest experiences with this approach.

In Indonesia, pollution reduction agreements are part of a tightly focused programme, known as PROKASIH (or Clean River Programme), begun in mid-1989 to clean 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 between provincial governors and directors of those companies have been drawn up and signed. 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 loadings in half within an agreed timeframe. 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 9 rivers average daily pollution load has increased since 1990-91[11] (Afsah, Laplante and Makarim, 1995). In provinces which have demonstrated the best results, agreements have been renegotiated upon expiry with more stringent conditions or, in the event the conditions were not

[11] 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.

met, governors have issued warnings to the industries concerned. The government has also used publicity regarding participants' performance to good effect. In 1991, following a public announcement by the Population and Environment Minister of individual firms' records of violation of their agreements, the government agency monitoring the programme (BAPEDAL) was inundated with inquiries from industry about regulations and standards. While the data clearly shows sharp BOD reductions in the initial years of the programme, more recent indications are that BOD loads have been rising again, raising doubts about the sustainability of such an approach in the face of strong growth pressures.

Another form of suasive instrument (build on a CAC foundation) is the assignment to the public of the legal right to know about the pollution generated by industrial enterprises. Such community right-to-know legislation exists in the United States. The Emergency Planning and Community Right-to-Know Act of 1986 requires that certain types of manufacturing facilities 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. The high visibility this system gives to the pollution streams of specific firms has increased managers' incentives to improve environmental performance. In consequence, 1,000 firms signed on to a programme administered by EPA's Office of Pollution Prevention and Toxics under which they committed themselves to a 50-per cent reduction in emissions of 17 high-priority chemicals by 1995. The legislation has also been a factor in encouraging some industry associations to launch industry-wide environmental management programmes. The Chemical Manufacturers Association launched a Responsible Care Programme in 1989[12], using toxic release data to track performance of its member companies and to assist in the identification of best practice cases which can set an example for other members.

While the author is not aware of be any comparable legislation on the books of developing countries, Martin (1992) argues that the approach may be suitable for developing countries, where resources are limited for intensive monitoring and enforcement by a centralised agency. US experience suggests, however, that complying with the law is by no means costless: 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. The EPA must also administer the law and somehow ensure minimal accuracy of reporting.

A system of strictly voluntary agreements may be less costly than right-to-know legislation, but it is also likely to be less consistently effective. For, while both rely to some degree on publicity (whether positive or adverse) as an enforcement device, with voluntary agreements the public revelation of participants' pollution records is somewhat serendipitous, while with "right-to-know" legislation disclosure of the record is assured. Indonesia's PROKASIH falls somewhere in between, in that participation in the programme is not voluntary, and there is a regular system of monitoring of environmental performance of participants. The next step would be to incorporate a provision for regular public disclosure, so that participating companies know their environmental performance will become public record.

[12] In this case, another factor was a series of chemical accidents (most notoriously at Bhopal, India) which raised fears in the industry that the government would impose much harsher regulations if it did not seize the initiative and discipline its own members.

3. *A Framework for Analysing Implementation Problems*

Normally, in assessing alternative environmental policy instruments five criteria need to be weighed: environmental effectiveness, economic efficiency, equity, administrative feasibility, and political acceptability[13]. These are not always easily separable, however. For example, administrative complexity raises the economic costs of implementing a policy. Likewise, if the compliance costs are very large, or if the distribution of those costs is seen to be inequitable, the policy may face strong political resistance. Most often, problems of policy implementation arise because of inadequate regard for either administrative feasibility or political acceptability, or both. Another possible cause of such problems is the mismatch between the type of instrument chosen and the nature of the problem targeted. Still others are weaknesses in the legal and institutional framework.

3.1. *Political considerations*

Policy makers have their own interests which need to be factored into an analysis of the political process, as do the members of the bureaucracy ultimately responsible for policy implementation (Hahn 1989). Other relevant interest groups include polluting industries and/or consumers, pollution victims, environmental groups, and the voting and taxpaying public.

Until recently at least, political support for EIs has been lukewarm, including in most OECD countries, which goes a long way towards explaining their infrequent use. Politicians and administrators accustomed to having direct control over individual polluters may be reluctant to cede that for a system in which they simply set targets and let industry decide how to meet them. Somewhat less explicably, polluting industries have also tended to favour regulations over EIs. First, regulations generally have a greater *status quo* bias than EIs. For example, new pollution sources are frequently subjected to stricter standards than existing sources, a practice referred to as "grandfathering". Also, the choice of emission or effluent standard is often dependent on the existing abatement technologies or those expected to be available in the near future. The practice of "grandfathering" may also be employed with some EIs -- e.g., in the initial allocation of permits under a permit trading scheme -- as a way of validating the prior claims of incumbents, but it is less common with pollution charges. Another consideration is that, unlike standards, pollution charges are perceived as imposing a "double burden" on a polluting industry since it must invest in abatement technology and still pay the charge on its residual emissions. This is not the case with tradable permits, however, assuming they are not auctioned but grandfathered[14].

3.2. *Technical/informational problems*

A key advantage of EIs over CAC is the former's informational parsimony. This results from the fact that the use of economic instruments forces polluters to reveal their willingness to

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[13]Kumar and Sherif (1995) list a number of criteria for evaluating a proposed policy instrument: these make a helpful checklist:

(i) Does the policy achieve the stated goals?
(ii) Is it cost effective?
(iii) Does it provide incentives for research and innovation for better pollution control and pollution prevention technologies?
(iv) Is it administratively feasible to monitor and enforce?
(v) Is it publicly transparent?

[14] There is still an opportunity cost to holding permits, since a company foregoes the profit it could make if it were able to reduce pollution at a marginal cost below the permit price.

pay (WTP) for the right to pollute and thereby their marginal abatement costs (MAC). A uniformly applied regulation or standard does not depend on polluters' WTP or MAC. If, however, regulators sought to design an efficient system of regulation in the sense that the standard would be varied across polluters in such a way as to equate their MACs, the task of acquiring all the necessary cost information and keeping it up-to-date would be enormous.

While informational economies are a decided advantage of EIs, some EIs are much more information-intensive and technically complex to administer than others. A tradable emissions permit scheme, for instance, requires detailed monitoring of emissions and tracking of permit trades. In the United States, where this sort of scheme is most frequently used, the sulphur dioxide (SO_2) permit trading scheme relies on an annual spring auction held at the Chicago Board of Trade (home to the commodities exchange) to reveal permit prices. While such exchanges may be less well established in many developing countries, organising an auction is not likely to pose as great a challenge as tracking the bilateral permit trades between individual enterprises and monitoring their actual emissions. By contrast, a product-based pollution charge could be relatively easily administered in many countries, assuming they have a good track record in the collection of excise taxes. The information revealed by pollution charges is not entirely costless, however, since a trial-and-error process whereby charge rates are repeatedly adjusted until the appropriate level of charge is reached introduces uncertainty which can adversely affect investment and other decisions of industrial enterprises or others subject to the charge.

3.3. *Mismatch of instrument to the problem at hand*

The instrument chosen needs to be well-tailored to the characteristics of the pollution problem at hand. Factors which need to be weighed include:

(i) the nature of the pollutant (whether acutely toxic, bioaccumulative, carcinogenic, etc.) and of the dose-response function (whether linear, exponential, subject to threshold effects, etc.);

(ii) the "demographic" characteristics of pollution sources (a few large sources or many small sources; geographically concentrated or dispersed sources; stationary or mobile sources);

(iii) the "economic" characteristics of the polluters (whether operating in competitive or non-competitive industries; whether facing hard or soft budget constraints; whether facing similar or widely differing abatement costs);

(iv) the availability of economical substitutes/alternatives to the polluting products or processes; the elasticity of demand for the products of polluting industries.

Point (i) bears on the expected environmental benefits from close monitoring and strict enforcement, while point (ii) bears on the technical feasibility and costs of such monitoring. For example, where small doses of a substance can prove highly toxic to plant and/or animal life, the benefits in terms of damage costs avoided may well justify very close monitoring and strict enforcement source-by-source. However, if the sources are numerous and widely dispersed, the costs of individual source monitoring may be extremely high. In this case, a policy objective would be to find an instrument that reduces the need for such monitoring without increasing significantly environmental risk.

Another example is that of mobile source air pollution. Technical standards applied at the point of production may be an environmentally effective control strategy, but that depends heavily on the rate of turnover of the motor vehicle stock or, assuming that rate is low, on a complementary programme of retrofitting older vehicles with new control technologies. In any case, it is almost certainly not the most cost-effective strategy. While emission charges could be a cost-effective measure (at least in terms of abatement costs), the complexities of implementation are daunting and render this approach impractical for the time being. On the other hand, a product charge on the major polluting input (gasoline) can be easily administered (since most governments already levy gasoline taxes) and the efficiencv loss is relatively small. The environmental effectiveness could be enhanced by some degree of differentiation between types of gasoline -- e.g., leaded versus unleaded -- and between gasoline and diesel fuel, according to pollution effects.

The effectiveness of an emission charge or a product tax as incentive devices depends critically on relevant elasticities. If, in the preceding example, the elasticity of demand for gasoline is low, then the government would need to set a very high excise tax in order to have a noticeable impact on consumption levels. This might well make such a tax politically unacceptable. With limited substitution possibilities elasticities may well be low in the short run; allowing time for development of substitutes and alternatives and behavioural changes, elasticities can rise quite significantly. Pearce (1990) reports short-term demand elasticities for gasoline in Mexico in the range of -0.1 to -0.2; in the long run, however, they rise to over -1.0.

3.4. *Legal and institutional barriers*

Aspects of the legal framework which need to be considered in designing environmental policies are:

-- the system of allocation and enforcement of property rights, and

-- the system of negotiation, execution and enforcement of contracts.

Institutional factors to be considered include:

-- the effectiveness of public institutions and in particular of those institutions charged with management of the environment or, in the case of eco-taxes, with the collection of tax revenues;

-- the level of development of market institutions.

Property rights are important determinants of the degree of stewardship which individuals or communities are willing to exercise over the environment. Thus, open access resources -- whose exploitation is not governed by any clear and enforceable system of property rights -- are particularly susceptible to degradation. Property rights -- whether vested in individuals, communities or the state, whether defined by law or by custom -- are important preconditions for the enforcement of any environmental policy, whether based on CAC or on

EIs. The introduction of controls or incentives designed to affect use of environmental resources rests on some definition, whether explicit or implicit, of property rights in those resources. Take, for example, the case of air pollution control measures. If the government introduces standards for nitrogen oxide or sulphur dioxide emissions, it is challenging *de facto* property rights in air quality. Whereas previously the dominant right had been that of the polluter to use the atmosphere as a waste sink, the regulation re-defines the dominant property right to be that of the public to breathe clean air. The ability of the government to give this re-distribution of property rights practical force depends very much on the degree of political support it enjoys for controlling air pollution.

In this example, the property rights remain implicit and their reallocation can occur only through the political process. In other words, there is no market in such property rights. Certain economic instruments -- notably, tradable permits -- rest on a more explicit definition of property rights in environmental resources and their allocation through markets. Owners of pollution rights who do not want to exercise those rights can sell them to others who do. The total available supply of those rights is restricted, however, which is why they possess economic value.

A key difference between a tradable permit scheme and a CAC approach is that in the former the initial assignment of property rights becomes a separate decision variable, unlike in the regulatory case where the initial assignment of rights is dictated by the output and pollution levels of existing firms. In a tradable permit scheme the government can choose an allocation rule which does not simply validate the *status quo*. For example, by auctioning off the supply of pollution permits, it can ensure that they go to those with the greatest willingness to pay, extracting the economic rents in the process. Those incumbents unwilling to make a sufficiently high bid could lose out to newcomers or even to environmental groups interested in withdrawing permits from use. It is understandable, then, why existing enterprises might prefer a strict regulatory approach to one involving permit trading or, in the event of trading, would prefer an initial permit allocation grandfathered according to historical output or capacity levels.

Another factor which can impact on choice of instrument is the degree of public versus private ownership or control over productive assets. Where public-sector enterprises represent a sizeable presence in polluting industries, the potential for conflict of interest arises. Under a regulatory approach, it is desirable that an independent agency be vested with the necessary legal authority and enforcement powers over state enterprises; the ministry managing a particular enterprise cannot be considered an impartial enforcer since its primary objective is to meet production (or revenue) targets. In the case of EIs, applying pollution charges may be largely ineffectual if state-owned polluting enterprises can recoup any added costs from the government budget or financial institutions (Panayotou 1994). One interim solution could be to use a suasive instrument in the form of "environmental plan contracts" negotiated between the agency responsible for the environment, the ministry in charge of specific public enterprises, and the managers of those enterprises. The contracts would specify measures to be adopted to reduce pollution, performance targets to be achieved, and penalties for non-attainment. Including environmental performance among the criteria used to calculate incentive pay for state enterprise managers and workers could reinforce such contracts.

Institutional capabilities of the government to implement specific types of policies need to be considered. Trained personnel capable of designing and administering a programme of regulatory enforcement based on detailed technical and/or performance standards are not likely to be readily available in many developing countries. Government ministries and agencies with sizeable influence and resources (notably the finance ministry and other economic ministries) are more likely than the environment agency/ministry to attract the small numbers of highly educated and qualified people. Even if a dedicated and competent core of personnel heads the environment agency, it is likely to have difficulties executing the agency's mandate without adequate laboratory facilities and support personnel to carry out an effective monitoring programme. Moreover, low salaries of government enforcement agents can also contribute to weak or partial enforcement. This litany of institutional weaknesses in developing countries has been recited many times over.

Are EIs an answer to these weaknesses? They can be, but the implementation of EIs has certain institutional prerequisites of its own. For instance, collection of an emissions charge depends on a reasonably effective tax administration, and it also requires monitoring capabilities if charges are to be linked, however loosely, to actual emissions. Tradable permit schemes make more onerous institutional demands, since they require an institutional machinery for issuing permits, recording trades and checking them against actual emissions. Product charges place perhaps the fewest additional demands on government's institutional capacity.

More generally, the proper implementation of EIs presupposes the existence of reasonably well-functioning markets and relatively undistorted prices. In many developing countries, and perhaps especially in economies-in-transition, prices of certain natural resources, energy and other productive inputs are subsidised. Thus, before introducing Pigovian taxes or other economic instruments, price reform is a prerequisite. While such taxes are designed to internalise marginal environmental costs (and user costs in the case of non-renewable resources) in product prices, very often those prices fail even to reflect fully the marginal costs of supply (see Pearce and Warford 1993, Chapter 7). This suggests that resource pricing reform could yield not only environmental gains but also allocative efficiency gains (a "no regrets" policy). In practice, of course, removal of energy, water or other input price subsidies can be highly charged political issues, especially if the equity effects are not adequately addressed. Also, removing distortions in one market may not have the desired incentive effects if distortions remain in other markets. For example, a rise in energy prices which exerts pressure on industry to invest in new energy conserving technologies may not prove effective if they lack access to credit. Similarly, the impact of the removal of water subsidies to farmers depends critically on whether the prices of their produce are also liberalised.

Other institutions may be needed to reinforce the role of markets in the implementation of EIs. Not all the information needed by a polluting industry to be able to respond rationally to a pollution charge is provided by the market itself. One important piece of information is what types of abatement technology are available at what cost. While a local supplier industry and market may develop for such technologies, before it does access to abatement technology information is likely to be limited and costly. Government can provide a valuable service by collecting and disseminating such information to local industry, but other institutions like industry associations or research institutes may be in a position to perform that function more effectively.

4. *Addressing Obstacles to Policy Implementation*

The preceding discussion highlights a number of similarities in the policy implementation problems facing OECD countries and developing countries. Political opposition to greater reliance on EIs may be an obstacle in both groups of countries, though perhaps moreso in OECD countries where entrenched industrial and bureaucratic interests are apt to be stronger. Problems stemming from a weak legal and institutional framework are apt to be more serious in developing countries. Both groups of countries face similar challenges in matching the policy instrument(s) to the specific characteristics of the problem and should be able to learn from each other's experience in this regard.

Following is a distillation of some of the obstacles to and problems in implementation of environmental policy which have been touched upon in the preceding pages, with a few observations on how they have been or might be addressed. The purpose is to stimulate thought and discussion, not to provide a blueprint to policy makers. It should be clear that some of the problems are of a quite general nature, applying to regulatory as well as to EI-based approaches, while others are more common with one approach than with the other.

4.1. *Ensuring political support*

Political transparency. The political economy of environmental policy is the subject for a book. The above examples point to the importance of finding out the likely winners and losers from a particular policy before implementation. A transparent political system provides a mechanism for preference revelation before policy implementation through public disclosure of information, public hearings, and public debate. This can help policy makers anticipate the sorts and sources of resistance they might face to a proposed policy and to take those into account in revising policy design. Of course, policy compromises may have a cost in reduced economic efficiency and/or environmental effectiveness, but that may still be better than the *status quo*. A policy making process which does not permit such *ex ante* debate on policy is more prone to *ex post* obstruction of implementation by vested interests.

Transparency of policy impacts. Transparency can involve risks as well. It has been argued, for example, that one reason EIs have not been more widely endorsed -- despite their efficiency properties -- is that they make the costs of pollution control (and the distribution of those costs) more apparent than is the case with CAC measures. For example, while the consumer may ultimately pay the bill for environmental regulation of fossil fuel-burning power utilities, the effect is not as clear as if the government were to impose a tax on the sulphur content of fuel. Thus, the consuming public may object more vigorously to the tax than the regulation, even if the consumer welfare loss is smaller with the tax.

Transparency in defining 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) should neutralise opposition. Assuming the government also intends to raise revenue, then earmarking the tax revenues for environmental expenditure enhances transparency, but it also reduces government 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 total flexibility to reallocate revenues from the tax/charge, 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 less distortionary via an eco-tax. This points to the need for a general equilibrium analysis of any proposed environmental tax with significant revenue implications.

Localised 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. In this way, it could be fine-tuned before an attempt is made to replicate it at the national level. It seems highly likely, in most cases, that emission charges or other pollution taxes would need some fine-tuning, unless governments are exceptionally well-informed about the shapes of abatement cost functions and demand curves. In some cases, a uniform national policy would not be appropriate since the problem itself is localised -- e.g., photochemical smog in a particular city. Nevertheless, other cities facing similar problems could clearly derive useful lessons from a local policy experiment and avenues for transferring those lessons need to be opened where they do not already exist (e.g., through a council/conference of municipal or provincial environmental policy makers).

Gradual escalation. The rationale for phasing in gradually a new measure -- whether stricter standards or emission charges -- is to allow incumbents operating with old technology time to depreciate their investments. In one sense, those with the oldest technologies might be thought to pose the least problem, since they are most likely to have depreciated their equipment fully. On the other, the fact that they have not been upgrading their technology would seem to suggest competitive weakness and, hence, vulnerability to additional cost burdens. In many instances, existing facilities are treated differently from new facilities in the application of new regulations (e.g., given longer to comply, subjected to less strict standards), which may make political sense but also discourages the early retirement of old plant and equipment. In applying a new pollution charge, there is no rationale for distinguishing old from new sources, since the efficiency of the instrument is a direct function of the degree of variation in abatement costs across facilities -- and that depends in no small measure on their relative ages (hence generational differences in technology employed). Still, it may be advisable to introduce the charge at a low rate, even if that initially mutes the incentive effect. The mere fact of the charge may represent a significant policy breakthrough where users and polluters of resources have grown accustomed to their being available free of charge. It is important for the government to make clear at the outset that future rate increases can be expected and, preferably, to commit itself to a preannounced schedule for such increases.

4.2. *Removing distortions and building institutions.*

Policy sequencing. Market and policy failures are still fairly commonplace in both the developed and the developing world and, arguably, they are especially serious in formerly centrally-planned economies. Remedying those failures involves a process of structural reform that extends well beyond environmental policy but which nevertheless can have major implications for the environment. In particular, the elimination of price distortions in markets for natural resources and energy can go some way towards encouraging their more efficient use. Such resource pricing reforms should therefore be an early priority.

Economies in transition cast in bold relief some of the policy challenges facing other developing countries. For example, such economies start from a situation of more pervasive price distortions and, as prices are liberalised, they may remain in disequilibrium for some time. Also, it takes time before new entry dissipates market power and reduces price distortions in many newly liberalised markets. Moreover, an underdeveloped system of property rights can create uncertainties and encourage short-term resource mining. Thus, even as the government loosens its control over market prices and resource allocation decisions, it may need to tighten its grip initially on the regulatory levers. In any event, experience would suggest that, even if the eventual environmental policy objective is to rely more on EIs, building a solid legal and regulatory foundation may need to assume priority in the short run. Where pollution damage is already severe, there may be an economic rationale for an initial focus on stricter regulation, since the marginal benefits from tighter controls are likely to be high while, precisely because of the low initial abatement effort, marginal abatement costs are apt to be low. Later, as pollution levels are substantially reduced -- and the marginal benefits of further reductions decline while marginal costs rise more steeply -- the advantages of EIs become more apparent.

A preventive strategy. What is a suitable policy approach for those developing countries which find themselves in the enviable position of not having degraded their environments to the point where they are left with few alternatives to draconian control measures? In other words, if a government were in a position to develop an environmental strategy based on foresight rather than hindsight, what would it look like and what place would EIs occupy within it? As the economy grows, the challenge is to prevent significant deterioration in the environment and there is good reason to believe that a programme based on EIs is best suited to that task. For, a key feature of EIs is the built-in incentives that they provide to the continued search for ways to reduce pollution levels, whether through changes in product mix or in process technology, through end-of-pipe treatment, or by other means. It is the continued impetus to pollution-reducing innovation which should ensure the sustainability of the growth process. There is, moreover, a major difference between the country with foresight and the country with hindsight with regard to their abatement cost functions. If it is indeed the case that preventive measures are on average less costly than remedial ones, that cleaner process technologies are cheaper than end-of-pipe treatment technologies, then the country with foresight should be able to achieve and maintain the same environmental quality level as the country with hindsight but at lower cost. Much of the evidence on the cost advantages of prevention over end-of-pipe treatment is anecdotal and further research is desirable in this area.

4.3. *Applying economic instruments in an open economy*

As both developed and developing economies have become more open and trade-dependent in recent years, the impacts of domestic policy on international trade competitiveness assume greater importance in policy debates. Tax policy in general and environmental taxes in particular are central topics of debate. Variables that may influence the effectiveness of environmental policy and its trade impacts include: whether the country is a small economy that has no influence on international prices or a large one where a tax might affect international terms of trade; whether affected sectors produce homogeneous or differentiated products;

whether substitution possibilities and possibilities for technological innovation are strong or weak; the mobility of factors of production; whether trade agreements allow countries to use border tax adjustments[15] and other trade measures to reduce adjustment costs; and whether other countries introduce similar measures (OECD, 1996).

Effects on competitiveness. In an open economy, introducing pollution control measures -- whether regulations or pollution charges -- which raise costs to domestic producers of tradable may place them at a competitive disadvantage vis-à-vis foreign producers not faced with the same cost burden. This is especially so where domestic producers are strictly price-takers in international markets. Where factors of production (notably capital) are mobile, the imposition of environmental regulations or taxes on domestic industry may also cause a shift of investment -- notably in polluting industries -- towards less heavily regulated/taxed locations. The competitiveness effects of environmental policy is a paramount political concern in both OECD members and developing countries.

It is not simple to determine *a priori* which is likely to have a greater effect on competitiveness -- a regulation or an economic instrument. It depends on the specifics of the case and on the type of economic instrument. For instance, even though a pollution tax might involve a lower social cost than a comparable regulation, it could still have a more adverse effect on competitiveness because of the "double burden" it imposes on polluters. Effects on competitiveness will also depend on the fiscal impact of the tax -- e.g., if a pollution tax were combined with a reduction in the marginal tax rate on capital income or in social charges, its effects would be different than if it were an add-on to existing taxes.

Empirical studies of the impacts of existing environmental policies on competitiveness generally show a weak one. Only in a handful of industries are environmental control costs a high enough share of total costs to have had a major impact on competitiveness. While it is the case that the growth of certain polluting industries has been much faster in developing countries (generally assumed to have weaker environmental standards) than in developed countries, this is true of many less polluting industries as well and does not necessarily reflect a response to differential environmental standards. Other factors like differences in labour and land costs, access to raw materials and markets are often more significant explanations. Nevertheless, it is worth bearing in mind that these studies examine the effects of current environmental policies and not of "optimal" policies that would fully internalise environmental externalities.

Even if actual impacts of environmental policies on competitiveness are small, they may still be perceived to have large impacts and may therefore pose a political challenge. The case of the US CFC tax illustrates the importance of such perceptions (Barthold 1994). While the law authorising the tax is non-discriminatory between domestic supplies and imports of CFCs, it authorises the imposition of a derivative products tax on imports containing or made with CFCs (on the argument that foreign producers of such products should not enjoy a cost advantage by virtue of their access to a cheaper supply of CFCs). To tax all such imports, however, even when the amount of CFC involved is minuscule, adds significantly to the administrative burden of implementing the tax. Normally, a device to reduce administrative costs is to grant a *de minimis* exemption allowing the customs or other revenue-collecting authority to forego collection of inefficiently small amounts of a tax. In this case, however, the

[15] Border tax adjustments involve the application to imports of domestic taxes and the remission of domestic taxes on exports of like products; they are used to neutralise the competitive effects of the domestic tax.

law specifically prohibits the granting of any *de minimis* exemptions for electronics products made with CFCs, reflecting the political importance attached to the competitiveness of the US electronics industry.

Reconciling environmental and trade policies. The preceding example also raises the issue of a possible conflict between a country's environmental objectives and its trade policy objectives. For instance, countries that are members of the WTO or that aspire to become members need to consider whether a particular environmental tax is consistent with the rules of that organisation. For instance, while the use of a border tax adjustment on US imports of CFCs and of products containing CFCs would appear consistent with WTO rules (see OECD, 1995), the tax on products presumed to be made with CFCs is more problematical. From a technical point of view, it is very difficult to ascertain how a particular product has been made. More generally, border tax adjustments for domestic process taxes are of questionable WTO-compatibility.

Still, governments that are committed to implementing the polluter pays principle (PPP) must accept that this will mean higher costs to some industries. While gradualism in implementing PPP may reduce the magnitude of the adjustment costs to the economy, such gradualism implies a degree of government foresight that seems to be too often lacking. More often than not, environmental control measures are implemented under duress and as a matter of urgency only after a problem has become so serious as to defy all attempts at deferral.

Once governments accept the likelihood of having to impose costs on polluters, then the issue becomes one of choosing the instrument which achieves the desired environmental improvements, if possible at least economic cost, but also with the smallest possible adverse effect on competitiveness. Working to the advantage of trade-oriented economies is that they are in a favourable position to finance the import of new capital equipment incorporating cleaner technologies. With an environmental policy built around economic instruments rather than CAC, domestic industries have that much more freedom (and incentive[16]) to take advantage of the range of environmental technologies available through foreign trade and investment.

[16]In the case of the wood pulp industry, for example, Wheeler and Martin (1992) find that trade-oriented economies show a significantly higher rate of adoption than closed ones of the cleaner thermomechanical pulping process.

REFERENCES

Afsah, Shakeb, Benoît Laplante and Nabiel Makarim (1995), "Program-Based Pollution Control Management: The Indonesian PROKASIH Program", World Bank, Washington, D.C., October (processed).

Barde, Jean-Philippe (1994), "Economic Instruments in Environmental Policy: Lessons from the OECD Experience and Their Relevance to Developing Economies", OECD Development Centre Technical Paper No. 92, Paris, January.

Barthold, Thomas A. (1994), "Issues in the Design of Environmental Excise Taxes", *Journal of Economic Perspectives*, Vol. 8, No.1, Winter 1994, pp.133-151.

Chien, E. (1991), *Working Towards Environmental Quality in the 21st Century*, Environmental Protection Administration, Government of Republic of China, Taipei, January.

Chung, Chin Seung (1991), "Environmental Issues and Policies in Newly Industrialising Open Economies: The Case of Korea", Prepared for the Workshop on Trade and the Environment, GATT Secretariat, Geneva, 10-11 June.

Chung, Chin Seung and Sandong Lee (1992), *Environmental Management in Korea*, Report prepared for the OECD Development Centre Project on Environmental Management in East Asia: Challenge and Response, Final Draft, October, Paris.

Eskeland, Gunnar S. (1994), "A Presumptive Pigovian Tax: Complementing Regulation to Mimic an Emissions Fee", *The World Bank Economic Review*, Vol.8, No.3, pp.373-394.

Fan, H.S.L., P.G. Menon, and P.S. Olszewski (1992), "Travel Demand Management in Singapore", *ITE Journal*, December, pp.30-34.

Godard, Olivier (1994), "Introducing Environmental Taxes in Economies in Transition: Conditions and Obstacles", in *Taxation and the Environment in European Economies in Transition*, Centre for Co-operation with the Economies in Transition, OECD, Paris.

Haga, Kenichi and Yuko Yano (1992), *Environmental Management in Japan: Challenge and Response*, Prepared for the OECD Development Centre, Paris, Autumn (processed).

Hahn, Robert W. (1989), *A Primer on Environmental Policy Design*, Fundamentals of Pure and Applied Economics 34, Harwood Academic Publishers, Chur, Switzerland.

Johnson, P. McKay, S. and S. Smith (1990), "The Distributional Consequences of Environmental Taxes", IFS Commentary No. 23, The Institute for Fiscal Studies, London.

Kumar, Ritu and Yasser Sherif (1995), "Economic Incentives for Pollution Prevention: A Case Study of Coal Processing Industries, Dhanbad, Bihar, India", IIED Discussion Paper DP 95-01, February, London.

Lave, L. and H. Gruenspecht (1991), "Increasing the Efficiency and Effectiveness of Environmental Decisions: Benefit-Cost Analysis and Effluent Fees", *Journal of the Air Waste Management Association*, 41:5, pp.680-693.

Martin, Paul (1992), "Community Right-to-Know: The Impact of Toxic Release Inventory Reporting in the USA", Environment Department, World Bank, Washington, D.C., 25 February (processed).

National Environmental Protection Agency (NEPA), People's Republic of China (1992), *Pollution Charge in China*, Beijing.

O'Connor, David (1991), "Policy and Entrepreneurial Response to the Montreal Protocol: Some Evidence from the Dynamic Asian Economies", Technical Paper No. 51, OECD Development Centre, Paris, December.

O'Connor, David (1994), *Managing the Environment with Rapid Industrialisation: Lessons from the East Asian Experience*, OECD Development Centre Study, Paris.

O'Connor, David and David Turnham (1992), *Managing the Environment in Developing Countries*, Policy Brief No. 2, OECD Development Centre, Paris.

OECD (1989), *Economic Instruments for Environmental Protection*, Paris.

OECD (1992), *Reduction and Recycling of Packaging Waste*, Environment Monographs, No. 62, Paris.

OECD (1994a), *Applying Economic Instruments to Environmental Policies in OECD and Dynamic Non-Member Economies*, OECD Documents, Paris.

OECD (1994b), *Managing the Environment: The Role of Economic Instruments*, Paris.

OECD (1994c), *The Distributive Effects of Economic Instruments for Environmental Policy*, OECD Documents, Paris.

OECD (1995), *Report on Trade and Environment to the OECD Council at Ministerial Level*, Paris.

OECD (1996), *Implementation Strategies for Environmental Taxes*, Paris.

Pan, Tin-Bai (1994), "The Use of Economic Instruments in Environmental Protection: The Experience of Taiwan", in *Applying Economic Instruments To Environmental Policies in OECD and Dynamic Non-Member Economies*, OECD Documents. Paris.

Panayotou, Theodore (1994), "Economic Instruments for Environmental Management and Sustainable Development", Draft paper prepared for the United Nations Environment Programme's Consultative Expert Group Meeting on the Use and Application of Economic Policy Instruments for Environmental Management and Sustainable Development, Nairobi, 10-12 August.

Pearce, David W. (1990), "Public Policy and Environment in Mexico", Latin America and Caribbean Country Department, World Bank, Washington, D.C. (processed).

Pearce, David W. and Jeremy J. Warford (1993), *World Without End: Economics, Environment and Sustainable Development*, Oxford University Press, Oxford.

Pearson, Mark and Stephen Smith (1991), *The European Carbon Tax: An Assessment of the European Commission's Proposals*, The Institute for Fiscal Studies, London, December.

Phantumvanit, Dhira, Quanruedee Limvorapitak and David O'Connor (1994), *Applying Polluter-Pays-Principle: Time for Action*, prepared for the 1994 Annual Conference, Thailand Environment Institute (TEI), Bangkok.

Poterba, J.M. (1989), "Lifetime Incidence and the Distributional Burden of Excise Taxes", *American Economic Review*, Papers and Proceedings, Vol. 79, pp.325-330.

Poterba, J.M. (1991), "Tax Policy to Combat Global Warming: On Designing a Carbon Tax", in R. Dornbusch and J.M. Poterba, eds., *Global Warming: Economic Policy Responses*, MIT Press, Cambridge, MA.

Potier, Michel (1995), "China Charges for Pollution", *The OECD Observer*, No. 192, February-March, pp. 18-22.

Rhee, Ho-Saeng (1994), "The Use of Economic Instruments in Environmental Protection in Korea", in *Applying Economic Instruments to Environmental Policies in OECD and Dynamic Non-Member Economies*, OECD Documents, Paris.

Scott, Sue (1991), "Carbon Taxes: Theoretical Considerations and Estimates of the Effects on Households", Presented to the Economic and Social Research Institute Conference on *Controlling Carbon Dioxide Emissions: The Economic Implications for Ireland*, ESRI, Dublin, November.

United States Agency for International Development (USAID) (1990), *Ranking Environmental Health Risks in Bangkok, Thailand*, Two Volume Working Paper, Office of Housing and Urban Programs, Washington, D.C., December.

Vincent, Jeffrey R. (1993), "Reducing Effluent While Raising Affluence: Water Pollution Abatement in Malaysia", Harvard Institute for International Development (HIID), Cambridge, Mass., 18 March.

Wheeler, David and Paul Martin (1992), "Prices, Policies and the International Diffusion of Clean Technology: The Case of Wood Pulp Production", Environment Department, World Bank, Washington, D.C., 7 January (processed).

Woods, Susan M., H. Harsono Amir, and Michael Bordt (1992), *Environmental Management in Indonesia: Challenge and Response*, Report prepared for the OECD Development Centre, Paris, September (processed).