# TRADABLE PERMITS

## TRADABLE PERMITS – A MARKET-BASED ALLOCATION SYSTEM FOR THE ENVIRONMENT

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Environmental concerns are as old as Man, but tradable permits are a relatively recent innovation in dealing with these problems. Barely forty years have passed since the basic idea underlying tradable permits was stated by Coase (1960), who noted the reciprocal nature of harmful effects and suggested that their regulation might be accomplished as effectively and efficiently by a market as by the more conventional forms of regulation. Another decade would elapse before this insight was elaborated and applied specifically to environmental problems (Crocker, 1966; Dales, 1968; Montgomery, 1972). For another two decades, economists promoted tradable permits as a policy alternative, but the concept was generally regarded as impractical despite its theoretically attractive properties. Only in the last decade have tradable permits been implemented and declared a success, mostly in the United States, where they are still the exception, but also increasingly in Europe. An obvious question is whether the current enthusiasm for tradable permits reflects a passing fad or a more enduring trend. This article seeks to provide a perspective that will enable readers to answer that question.

### What is a tradable permit?

In its most general use, a tradable permit can be defined as a transferable right to a common pool resource. A common application is individual tradable quotas (ITQs) for fishing rights, which are granted in quantities to preserve the fishing stock and to avoid over-exploitation. In environmental applications, the common pool resource is air or water that does not contain concentrations of substances that harm human health or that degrade air or water quality in some manner. A narrower and more specific definition for environmental applications is then: a transferable right to emit a substance that can create pollution. Implicit in this definition, and in the concept of tradable permits, is the notion that some level of emissions does not create pollution, just as some level of fishing does not constitute over-fishing.

The permits that implement command-and-control regulations, what I will call conventional environmental permits, are a type of operating permit that specifies conditions concerning discharges that must be met for a particular facility to operate, or for a vehicle to be sold and operated. These permits typically cover a variety of emissions and they may set standards for each, perhaps limiting emissions to some relatively low rate per unit of input or output, or prescribing certain technologies or practices, which will have the same effect. They are attached to the facility or vehicle; they aim at controlling substances that can contribute to pollution; and they implicitly grant rights to emit the substance so long as permit conditions are met. Tradable permits differ from these conventional permits chiefly in focusing on a single discharge and being transferable. Transferability implies that the potentially polluting discharge can be identified and separated, or unbundled, from the underlying environmental permit. As such, transferability imposes specific requirements on tradable permit systems that are not necessarily required for conventional environmental regulation.

**A tradable permit focuses on a simple discharge and are transferable**

### Types of tradable permits

Tradable permits can be classified into three distinct forms – credit trading, averaging, and allowance trading – and distinguished by their relation to a conventional environmental permit.[1]

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[1] This typology is used and explained in greater detail with examples in Ellerman, Joskow, and Harrison (2003).

*Credit Trading* is the form closest to the conventional permit. A facility that does more than required to meet the conditions of its permit may get credit for its extra effort and that credit can be transferred to another facility that is thereby excused from fulfilling the condition of its permit in like amount. As the name implies, credit trading awards exemplary behavior and allows compensating regulatory relaxations of a common requirement. A distinctive feature of credit trading is certification, the process by which the regulator determines that credit-worthy activity has occurred and that the credit can be transferred. Certification has been a problem in that the regulator usually seeks to ensure that a facility will not receive credit for what it would have done "anyway," since granting credit in this case would lead to higher emissions by the firm to whom the credit is transferred. The transaction costs associated with certification have been high and have often overwhelmed the cost savings from the proposed trades. As a result, even when credit trading has been made a feature of environmental regulation, few trades have been observed. As noted by Shabman, Stevenson, and Shobe (2002), credit trading is an extension of conventional command-and-control regulation that keeps firm-level abatement decisions in the hands of the regulator.

**Credit trading rewards exemplary behavior, but requires costly certification**

*Averaging constitutes* a further step away from the underlying environmental permit in dispensing with certification. It can be seen as automatic credit trading in which parties that do better than required in their permits automatically receive credits that can be used by others without any question from the regulator whether the firm generating the credit would have reduced emissions anyway. The pre-existing standard about which emissions are traded is still in place, but in dispensing with certification, the regulator no longer attempts to make the abatement decision at the level of the firm. The common standard or technology is simply a reference point or benchmark about which differences are traded. Although averaging is a more precise term to describe what actually occurs, European terminology tends toward various formulations containing the term "relative," which imply trading around a limit relative to input or output instead of under an absolute cap as in an allowance system.

**Averaging is credit trading without certification**

*Allowance Trading* is also known as cap-and-trade, so called because of the absolute cap on emissions and the ability to trade emissions under the cap. Although a logical progression from credit trading and averaging, allowance trading is in several ways a radical departure. For one thing, the compliance requirement is entirely different. Instead of determining compliance by reference to a common standard and sanctioned or compensated deviations from it, firms are required to surrender a permit for *every* unit of discharge. Although the cap may be very constraining in the aggregate, no firm is expected to meet any specific standard. It must only obtain and surrender an allowance that can be readily bought or sold in the market. In effect, allowances have become essential inputs into production subject to the same marginal cost calculations as other inputs.

**Allowance Trding is radically different in that it must observe an absolute cap on emissions**

Two consequences flow from the allowance trading form of tradable permit. First, the regulator's task is not to specify an emissions standard, but a cap. This requires initial decisions concerning 1) an acceptable or optimal quantity of emissions and 2) the limits to trading, both spatially and temporally. Second, the rights to discharge are now explicit and must be allocated in some manner instead of being implicit and granted without question to the owners of the emitting facility.

These three forms of tradable permits can be seen as a progression from a centralized system in which abatement decisions throughout the economy are the sole province of the regulator to a more decentralized, "property rights" system in which firms take over the abatement decisions subject to the constraints of the cap and its spatial and temporal dimensions, which only the regulator can (and should) decide.

## Requirements for an effective system

As the most evolved form of a tradable permit system, allowance trading has prerequisites that differ in important aspects from what conventional command-and-control systems require. Some of the requirements of allowance trading are shared by averaging and credit trading systems, but not all or to the same extent. These prerequisites follow logically from the transferability of tradable permits and from the nature of allowances and the cap in allowance trading systems.

*Measuring emissions* is perhaps the most radical requirement of tradable permits for many, if not most, environmental programs do not determine compliance by the actual measurement of emissions. Compliance consists of installing and operating certain equipment, engaging in certain practices, or limiting certain inputs, all of which will reduce emissions, if enforced and implemented continuously. In contrast, tradable permit systems require measurement and continuous monitoring of the regulated emissions; otherwise there is no way to determine compliance or to define what is to be traded.[2] Although obvious, measurement is not always feasible and the growth in tradable permits is in part the result of changes in the ability to monitor, and the cost of doing so, that are associated with the late 20th century changes in information, control and sensing technology (Kruger, McLean, and Chen; 2000).

*Allocating emission rights* is a prerequisite of allowance trading only, although rights to emit are implicit in both credit trading and averaging, as they are in conventional environmental permits. Deciding who is entitled to receive these allowances is a matter of some consequence and great controversy.[3] Allocation involves a two-level decision, first, whether to auction the permits or grant them gratis to various entities, and then how to distribute the auction revenues or permits, as the case may be. Claimants for this rent have not been wanting and a considerable literature has developed on the optimal use of the scarcity rent created by the cap.[4] The pros and cons of various methods of allocation is well beyond the scope of this paper, but the fight over prospective rents – which combines unadorned rent-seeking with high principles of equity and efficiency – can be both an obstacle and a means of gaining consensus, as evidenced most recently in the negotiations surrounding the proposed EU Emissions Trading Directive (Council of the European Union, 2002). This controversy is largely avoided in credit trading and averaging because, ironically, the implicit assignment of the rent to the incumbent in the underlying command-and-control system of regulation is not raised and never challenged.

*Defining pollution*. All environmental regulatory systems presume some definition of pollution, but none are required to define it as specifically as cap-and-trade systems. Not only must the potentially polluting discharge be separately identified, but at least in theory the amount constituting pollution must be determined, as well as the spatial and temporal relation of discharges to the harmful effects. This requirement is faced by all environmental regulation, but the connection between emissions and the problem justifying the emission constraint is usually less direct. For instance, technology standards are prescribed not because they fit the problem but because they usually represent the "best" that can be done at the present, and that will contribute to the problem's solution, at the least, and perhaps eliminate it. While in theory the cap should be the level that will avoid harmful effects, an increasingly frequent solution is that the cap is set at a level that would be achieved if some "best" technology were to be required of all, or, especially in the case of greenhouse gas controls, at a level that is presumed to be a step in the direction of reducing emissions to some ultimate goal.

**Increasingly the cap is defined by best practice rather than environmental effects**

### Why tradable permits?

A fair question today in response to the attention being given tradable permits is: Why? Or alternatively: Why not taxes or conventional regulatory measures?

By far the more common policy instrument for achieving environmental goals is what has come to called command-and-control regulation, namely, the mandating of specific technology or other emission standards that are presumed applicable to all sources. The reasons for relying on conventional regulatory measures heretofore are easy to enumerate. Both taxes and tradable permits require emissions to be measured so that, if measurement is not feasible or it is costly, the only alternative is to prescribe the appropriate abatement technology or set of practices and to set up the enforcement regime that will lead to acceptably continuous application. Then, in the early days of modern environmental regulation, the sources of pollution were easily identifiable in being mostly large and stationary, which made it easier to pre-

[2] Credit trading could occur without measurement since the creditable reduction and the transfer depend entirely on regulatory determination. For instance, a regulator might allow a firm to meet a less stringent standard at one facility if it installs technology that is expected to reduce emissions more than required at another facility, without actually measuring emissions at either facility.
[3] When trading is allowed, the receipt of the right is distinct from its exercise. If allowances are freely granted, or "grandfathered," to incumbents, the recipient and the user are often the same, but the two functions remain distinct. In deciding to use a grandfathered allowance, the recipient-user is incurring an opportunity cost and effectively paying himself as *rentier* for the use of the permit. Were he not to use the permit, he could sell the permit and collect the rent as income.
[4] See, for instance, Harrison (1996), Goulder et al. (1999), and Dinan and Rogers (2002).

scribe appropriate abatement. Also, when faith in the capability of expert government agencies was greater than it is now, there seemed less reason to question this approach.

Those circumstances are increasingly less applicable on both sides of the Atlantic. The ability to measure emissions at relatively low cost has been greatly reduced by improvements in sensing and information technology. The big, initial pollution problems have been satisfactorily addressed, and the problems now facing modern post-industrial societies are far more complex and less obvious. Finally, experience and the rise of public choice literature has diminished confidence in the efficiency and equity of direct government intervention and led to a search for more effective, efficient and equitable approaches.

As market-based instruments, environmental taxes have the same efficiency attributes of tradable permits in leaving abatement decisions to firms, but they have been regarded as non-starters in the United States, and although more used in Europe, taxes are far from being the prevalent mode of environmental regulation. The reason for the apparent preference for tradable permits instead of taxes probably resides in the domain of political economy. For one thing, taxes appropriate to the state the scarcity rent that is embodied in tradable permits.[5] Moreover, the usual alternative to tradable permits is not an environmental tax but some form of conventional environmental regulation, which has the merit – from the standpoint of incumbents – of unobtrusively endowing them with the entitlement to the scarcity rent. The title is not as secure and it is not separable from the facility for which the environmental permit applies, but better an encumbered entitlement than none at all, or one that has to be bought. From this standpoint, tradable permits are worth considering, perhaps not so much because of their efficiency properties, but because they offer the possibility of unbundling the right from the facility and monetizing it directly.[6] If incumbent emitters had no voice in societal decisions, the choice of instrument would not be a matter of concern, but they do. In Europe, one should recall the frequent exemptions from energy or environmental taxes for energy-intensive industries, always because of "competitiveness" and what is invariably industry's willingness to accept equivalent, conventional, regulatory constraints that allow them to retain the scarcity rent. For these participants in the political system, taxes are the least preferred alternative and tradable permits are acceptable, even in cap-and-trade form, if the scarcity rent that the inefficient, default command-and-control system would award them, is not disturbed.[7]

**Although as efficient as environmental taxes, tradable permits allow the emitters to retain the scarcity rent**

### Whither tradable permits?

Two different approaches have been taken in adopting and implementing tradable permit systems. The first is what might be called the *de novo* approach whereby a new regulatory system is developed usually to deal with a new environmental problem, or at least one that is not dealt with directly by the existing system of environmental controls. The US Acid Rain Program and the proposed EU GHG Emissions Trading Programs are salient examples. These *de novo* programs invariably draw the most attention and their adoption is usually time-consuming and contentious for the very reasons that have been mentioned above. The nature of the environmental problem, the level of the cap, and the allocation of allowances are all likely to be matters of lengthy debate in any democratic society; however, once consensus is formed and a decision made, these programs can be implemented relatively quickly and effectively.

The other approach, which can be observed currently only in the United States, is one in which a tradable permit system supplants an existing conventional regulatory program. These programs arise when regulators realize that the goals of the conventional environmental program cannot be achieved, despite ample authority, usually because the specific targets of control are not as obvious as they were in the first wave of environmental regulation or because the economic and political costs of implementing the program as prescribed are too high, or even infeasible. Examples in the United States are the Northeastern $NO_x$ Budget Program

[5] The Swedish $NO_x$ emission tax is a notable exception that supports the point. The revenue from the tax on NOx emissions is returned to incumbents on a basis other than current emissions.
[6] The rents in conventional regulation are capitalized in the facility to which the permit is attached. This value accounts for a portion of the usual excess of the sale price over book value for many existing powerplants, refineries, and other industrial facilities.

[7] Perhaps, no better current example exists that the recent (December 2002) compromise concerning auctioning and grandfathering in the EU Emissions Trading Directive. Despite strong arguments in favor of auctioning, at least 90% (i.e., not excluding all) of the permits will be grandfathered, that is, distributed gratis to incumbents.

and the RECLAIM programs in the Los Angeles Basin, for both of which the caps are set at levels that would have been achieved, in theory, by the existing command-and-control systems. In recognizing the impracticability of the detailed regulation to reach these goals, the regulator opts to attain the environmental goal by abandoning the pretense of making firm-level abatement decisions. A notable feature of this path, which is implemented by regulatory agreement and not by legislation, is that the rights to emit are retained by the incumbents, as they would be, had the default command-and-control system been practicable. The end result is that the tradable permit system quietly supplants the default command-and-control system.

## A familiar analogy

The development of tradable permits recalls a similar, much earlier common pool resource problem that all societies have had to confront: land. Like clean air and water, land was once freely available for the taking, but the increase of human activity made it scarce and all human societies have had to devise institutions to allocate the scarcity. Over the centuries, societies of widely differing historical and cultural traditions have devised institutions to distribute the rights to the use of land, and the rents that go with them. For advanced industrial systems, hardly anyone questions that a decentralized system of private property rights provides a better allocation than any other practicable method of managing this scarcity. The initial allocation of these rights may have been coercive and unfair, but that ancient act is lost in the mists of history and no one really cares now, even though a significant portion of everyone's lifetime income is devoted to acquiring the right to call a small piece of the earth home. Until recently, private property rights in land were strongly contested by some and large societies have attempted to implement systems that would manage the scarcity through centralized allocation, but they succeeded only in proving the incapacity of such an approach. The question now is whether the current common pool resource problem, the environment, can be dealt with any more successfully by centralized methods. If not, we should not be surprised to observe a similar decentralized, property rights system for the environment.

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# Tradable Permits and Other Environmental Policy Instruments

## – Killing one Bird with two Stones

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Economists have long made the theoretical case for the use of tradable permits (TPs) as an environmentally effective and economically efficient means of addressing environmental externalities. This has been given increased empirical support with the successful introduction of a number of schemes in the United States over the last two decades, with the $SO_2$ Allowance Trading Program being the most visible recent example (see OECD 2002 for a discussion of some recent programmes). Moreover, a number of other countries have started to introduce TP systems as well, for a variety of different types of environmental impacts. In the area of $CO_2$ this has been given increased impetus with the endorsement of TPs within the context of the Kyoto Protocol, most particularly by the European Commission which has prepared a draft directive on GHG emissions trading.

Tradable permits are rarely "stand-alone" schemes

However, TP schemes are almost never introduced as "stand-alone" schemes. They co-exist with – and interact with – other environmental policy instruments with the same, or very similar, environmental objectives. A key public policy issue is, therefore, to evaluate when and whether it makes sense to use two instruments to hit one target. This paper seeks to examine this question by analysing some of the potential interactions between TPs and other environmental policy instruments. It does so with reference to four other types of instrument which frequently interact with TP schemes: direct regulations such as performance and technology standards, environmental taxes or charges, subsidies for abatement inputs or capital equipment, and voluntary policy approaches.

### Tradable permits and direct regulation

In many senses most TP regimes have emerged out of direct forms of regulation. The original American EPA Emissions Trading Program is the clearest example of such a case. However, even more recent TP schemes have been underpinned by pre-existing regulatory schemes. In some cases, this is primarily of importance for distributional reasons. For instance, under Los Angeles County's RECLAIM program for $NO_x$ and $SO_x$, permits were allocated *gratis* to firms according to estimated emissions that would have arisen under the regulatory system that it replaced.

In other cases the effects are much more far-reaching. This is particularly true of baseline-and-credit schemes in which credits for emissions reduced are the units which are traded, rather than permits for emissions actually generated. Under such schemes, it is important to be able to determine when an emission which would have otherwise been emitted is deemed to have been abated. Some notion of a 'baseline' level of emissions is, therefore, the point against which the credit is generated.

In most baseline-and-credit schemes the baseline is that level which would be emitted if the firm complied with the existing regulatory system. For instance, under the EPA's Clean Air Act's Emissions Reduction Credit Program credits are created when firms reduce their emissions below the level allowed by their operating permit (see Hahn and Hester 1989). Similarly, in the Swiss VOC permit trading program in the Canton of Basel in Switzerland which was initiated in 1993, credits were created for emission reduction below the emission performance standard ($75\ mg/m^3$)[1] (see Jeanrenaud 1999). And finally, under the American Lead-in-Gasoline trading program,

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[1] Although in practice very few credits have been created due to the stringency of the standard.

**Box : The Costs of Regulatory Constraints on Permit Trading**

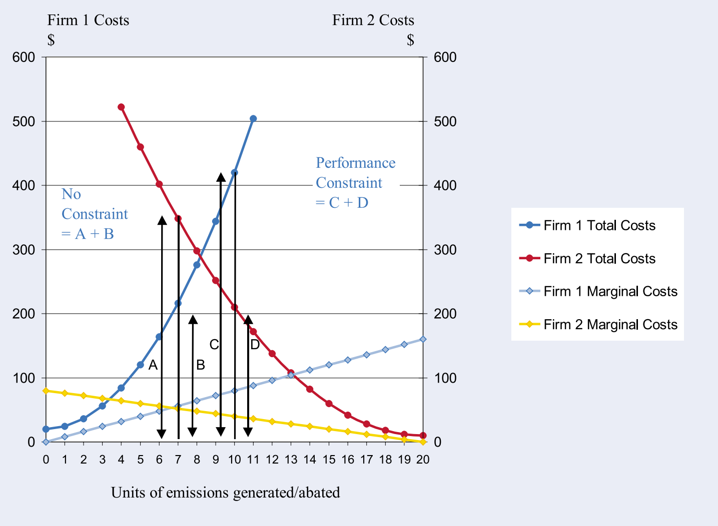
The costs of introducing a regulatory constraint (a minimum performance standard) on a firm within a permit trading system can be illustrated with a hypothetical permit trading market. Assume that prior to the introduction of any type of environmental regulation, two firms emit 40 units of a particular pollutant. The two firms have the following hypothetical total abatement cost (TAC) curves:

Firm 1: $TAC = 20 + 4\ EA^2$

Firm 2: $TAC = 10 + 2\ EA^2$

Where TAC is total abatement costs and EA is emissions abated. The figure below shows total and marginal abatement costs for Firm 1 (Firm 2) increasing from left to right (right to left) as levels of abatement rise along the horizontal axis. Upon the introduction of a TP system which caps emissions at 20 units, firms are allocated 10 permits each. Total costs will be minimised at the point at which marginal costs for the two firms are equal. This point is reached at the heavily shaded line to the left, when firm 1 buys approximately 3 units from firm 2, at a permit price of $52, and total abatement costs of $563 (the sum of the two arrows A and B).

Assume now that the regulatory authority decides to protect local environmental conditions in the jurisdictions where each of the plants are located by placing a regulatory constraint (such as a performance standard) of 10 units on firm 2. This might be a result of a concern that damages rise sharply above this level. In this case, the equilibrium is the heavily-shaded line to the right and total costs would rise to $630 (the sum of the two arrows C and D). Costs of compliance are, therefore, approximately 20% higher than in the case where permit trading is not restricted. Whether or not this results in improved economic efficiency depends upon the relationship between marginal damages of emissions from the two plants.




credits were earned if fuel was manufactured by refineries with a lower lead content than that mandated by regulatory limits. (See Stavins 2001.)

In other schemes, regulatory constraints are used to restrict the use of TPs in order to protect local environmental conditions. For instance, in the United Kingdom, the architects of the proposed programme for $NO_x$ and $SO_x$ trading have made it clear that the regime would have to protect local environmental conditions. However, it is not clear whether this would require the application of "Best Available Technologies" as mandated under the European Commission's IPPC Directive. This would severely restrict trading opportunities (see Palmer and Davies 2002).

Even under the American $SO_2$ Allowance Trading programme, some states have imposed regulatory constraints on the scope for trading in order to protect local environmental conditions. For instance, in Wisconsin, local air pollution regulations prevented generators from buying permits even though their marginal costs exceeded the prevailing permit price. In Illinois, the use of scrubbers was mandated (see Conrad and Kohn 1996 and Fullerton et al. 1997). In New York, the Department of Environmental Conservation filed a suit to force the EPA to use "deposition standards" to restrict the use of permits in environmentally sensitive areas (see Tietenberg 1995).

**Adding regulatory constraints increases costs, but also benefits**

What are the costs of such restrictions? Fullerton et al. (1997) estimated that applying minimum performance standards in the $SO_2$ program increases costs more than twofold. Farrell et al (1999) provide similar results for the American Northeast's NOx programme. (For a hypothetical numerical illustration see Box.) However, neither of these studies look at whether the benefits of constraining trade through regulatory requirements in order to protect local environmental conditions outweigh the increase in compliance costs. A single undifferentiated market would also be suboptimal, resulting in non-equalisation of marginal benefits and costs.

The key point is that because of the administrative cost of using one instrument to target the impacts directly in a differentiated manner which allows for marginal costs to equal marginal benefits for all emitters, a combination of policies is applied.[2] If applied efficiently this can be a 'second-best' policy option. Abatement cost minimisation for a given level of emissions is achieved through the use of the TP system, while still insuring against breaches of local environmental thresholds and other non-linearities in damage functions through regulatory constraints.

In other areas, the case for the retention of regulatory constraints is less evident. For instance, it has been proposed that the use of energy efficiency standards in the European Union's IPPC be retained even after the EU Emissions Trading Scheme for greenhouse gases has been introduced. While the objectives of the energy efficiency standards are broader than just climate change mitigation – indeed, their environmental objectives are manifold – it is clear that the retention of mandated energy efficiency standards may reduce the potential gains from trade within the Emissions Trading Scheme.

In some areas, regulatory constraints reduce the potential gains of emissions trading

This can be seen by examining a typical firm's objective function. The firm seeks to maximize profits, taking into account both production costs (PC) and compliance costs (CC). The latter are made up of both abatement costs (A) and permit use (P).[3] Capital (K), labour (L), and energy (E) are used both in production and abatement. The effect of the energy efficiency standard can be seen as a constraint on the firm's choice of factor inputs. In effect, the firm will not be able to use a ratio of energy use to output in excess of (E/Q)*. The maximization problem is, therefore:

$$\Pi = P^*Q - PC(K,L,E) - CC(A(K,L,E),P)$$

$$\text{s.t. } E/Q < (E/Q)^*$$

If (E/Q)* is less than would be the ratio chosen by the firm in the absence of the constraint, potential gains from trade will be lost. In effect, the firm will not be able to optimise its permit use. If this is not the case, then the performance standard is redundant. As such, the standard can only increase (or hold constant) compliance costs. Whether or not this cost is worth paying depends upon the efficiency of the standard in meeting the other environmental objectives for which it has been introduced.

## Tradable permits and environmentally related Taxes

There has also been considerable experience with the joint application of TPs and pollution taxes, particularly: as a means to reduce compliance cost uncertainty; and, as a means to capture windfall rents or tax revenue. The potential desirability of the joint application of taxes and permits (rather than using one or the other on its own) to reduce compliance cost uncertainty has been recognised for a considerable length of time. In particular, Roberts and Spence (1976) proved that the joint application of the two instruments was preferable in the presence of: A) non-linear environmental damages; and B) uncertainty concerning abatement costs. In effect, by delimiting the bounds of permit price uncertainty through taxes (and subsidies), the potential welfare losses from the regulatory authority either over-estimating or under-estimating marginal abatement costs can be reduced.

This has been dubbed the "safety valve" argument. By putting a cap on permit prices, regulatory authorities are able to convince risk-averse affected firms and households of the desirability of introducing a TP regime. In Denmark, the government explicitly used a "safety-valve" argument in setting the penalty at 40 DKK ($US 4.78)/ton of $CO_2$. In addition, some commentators have argued that the CFC tax in the United States was the binding instrument, and not the Ozone-Depleting Substances Program (see Stavins 2001).

It would, of course, be possible to achieve similar objectives within the TP system itself. For instance, under the $SO_2$ Allowance Trading program the government holds reserves of permits which it would release onto the market if the price were to reach $US 1,500 (see Tietenberg 1998). However, this has the disadvantage that the price can only be capped for as long as the reserve holds – excessive demand will eventually drive the price higher. Thus, the price effects are less certain, undermining the benefits in terms of reduced uncertainty. On the other hand, the environmental effects are more

[2] The usual economic case for the efficiency of marginal cost equalisation is really just a special case in which marginal benefits of abatement are equal across emission sources.
[3] Note that this is true even if permits are allocated gratis to the firm, since the firm will still face an opportunity cost for each and every permit surrendered.

certain with a permit reserve since under a tax-based price cap the government has no direct control over any unforeseen increase in emissions arising from the cap.

Another potential use of taxes in conjunction with TP regimes arises from the common use of gratis allocations of TPs rather than auctions. Whether this is done on the basis of historical emissions (grandfathering) or regulatory requirements or some other mechanism, firms will receive a windfall rent equal to the value of the permits allocated. In order to recover some of these windfall rents, taxes can be applied in conjunction with the TP regime. This appears to have been the motivation behind the use of the CFC tax in conjunction with the ODS Program in the United States. Initially set at $1.37/lb in 1990, it rose to $5.35 in 1995 (see Harrison 1999). This tax is paid on all CFCs sold and is complementary with the permit trading program. Thus, irrespective of the permit price, the tax has to be paid.

In a closely related vein, the desire of governments to retain at least some of the revenue from pre-existing environmentally-related taxes has also been a motivation for the joint application of taxes and TP systems. For instance, the United Kingdom's Emissions Trading Scheme for greenhouse gases co-exists with the Climate Change Levy which imposes a tax on coal, gas and electricity use on business, commerce and the public sector.

While the target groups of the two programs is somewhat different – with the ETS targeted upstream and the CCL downstream – the two policies interact in two ways. Firstly even for those downstream electricity users which are not themselves subject to the ETS, they will face price increases for electricity which are additional to the CCL. In addition, some coal and gas users will face a target under the ETS as well as be subject to the CCL (see Sorrell 2003). This results in double regulation, with externalities for at least some emissions from some sources being double-internalised.

## Tradable permits and subsidies

The use of environmentally-motivated subsidies in conjunction with of TP schemes is less widespread than the use of taxes or direct regulations with TPs, but there are still some important examples. Two areas will be highlighted. Firstly, financial subsidies are sometimes provided for improved environmental performance. In some cases, such subsidies are targeted at the level of investment (i.e. capital depreciation allowances for abatement technologies); in other cases they are targeted at specific inputs or outputs (i.e. tax exemptions on sales of renewable energy); and, in still other cases they are targeted much further upstream at technology development (i.e. public support for research and development in environmentally-benign technologies).

As long as the subsidies co-exist with a cap-and-trade system they will not undermine the environmental effectiveness of the TPs. However, they will not increase the environmental effectiveness either. Moreover, they will have effects on the distribution of impacts across firms and the economic efficiency of the system. For instance, under the $SO_2$ Allowance Trading Progam, the public utility commissions of some states have provided favourable tax treatment for capital expenditures on scrubbers relative to expenditures on permits, low-sulphur coal and other compliance strategies (see Bailey 1996).

Subsidies are used less frequently than taxes in conjunction with TPs

The effect of the subsidy will be to distort decision-making. Affected $SO_2$ emitters will be encouraged to purchase scrubbers in excess of the level which would be optimal. This will not improve the environmental effectiveness of the program in a global sense, but will merely drive down permit prices by releasing permits onto the market and encouraging other firms to use permits as a compliance strategy. It will also increase overall costs, above and beyond the costs associated with the direct financial implications of the subsidy.

Secondly, in other cases the relationship between subsidies and TP schemes is more direct. Indeed, perhaps the best-known combination between a TP system and the provision of subsidies is the United Kingdom's "sellers' auction" for $CO_2$ emission reductions under the UK Emissions Trading Scheme (ETS) (see Kitamori 2002 for a discussion). In a decreasing-price auction firms bid for government-provided subsidies against emissions reductions relative to their baseline emissions in 1998-2000. In total £215 million in subsidies will be provided in the period 2002–2006 (see DETR, 22/03/2002). In the first auction the price for allowances was £53.37/tonne.[4] The firm can sell any

[4] This is not equivalent to the market price for the cost of abatement of a tonne of carbon due to the annual nature of the commitment and other factors.

allowances for any reductions that it undertakes in excess of the amount for which it has bid.

While the ability to sell excess allowances is clearly characteristic of baseline-and-credit TP systems, the importance of the financial incentive for participation in the scheme is significant. In effect, the bidding scheme is perhaps best understood merely as a potentially efficient means of the allocation of subsidies. Instead of granting investment funds through detailed project applications on the one hand, or in a non-discretionary manner through undifferentiated subsidies on the other hand, firms are encouraged to reveal the true costs of abatement through the auction. However, the economic efficiency of the programme is dependent upon effective auction design, such that firms are not able to behave collusively in order to minimise reductions relative to the subsidies available.

**Tradable permits and voluntary approaches**

Voluntary approaches make sense only if the allocation of TPs is free. Even then there are problems.

Voluntary approaches to environmental policy can be integrated with TPs in two important ways:

- Adherence to TP systems by firms can be made voluntary through the use of ‘opt ins’; and,
- Emission reductions agreed to under voluntary agreements can be used as a means to allocate permits in a grandfathered TP scheme.

If the permits are auctioned, no firm would be likely to volunteer to be involved in the program in the absence of a regulatory threat or a financial inducement. In the case where permits are allocated gratis, the question is significantly more complicated since such schemes are characterised by strategic behaviour and financial uncertainty. Unlike under a mandatory cap-and-trade scheme the firm does not know what the ultimate “cap” will be, since this depends upon how many (and which) firms volunteer.

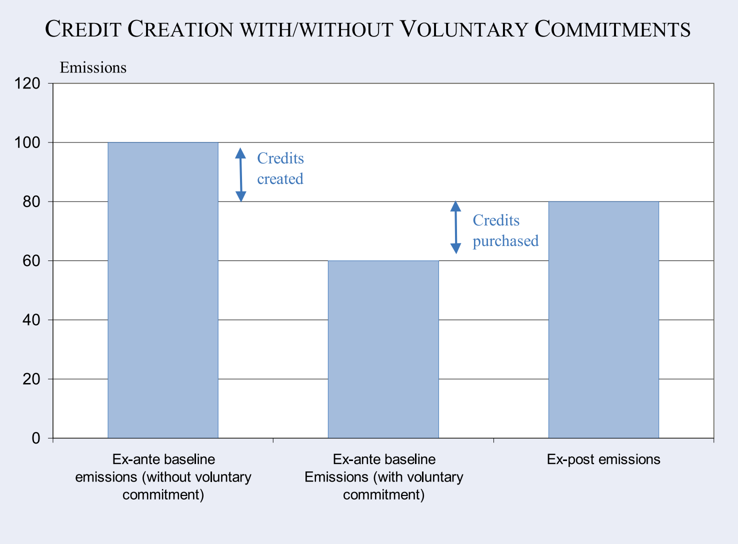
In effect, each firm faces a different expected benefit and cost schedule depending upon which other firms are involved. In some cases the net benefits will be positive and in some cases they will be negative relative to the case where they continued to adhere to some existing regulatory regime. It is possible that the distribution of costs and benefits is such that no firm will volunteer, even if it is in their collective interest to do so. Indeed, this is why the United Kingdom subsidised firms to participate in their ETS programme.

In most extant cases, however, voluntary adherence is only an option for a sub-set of firms, with most firms being mandatory participants. This is the case with the EPA’s $SO_2$ Allowance Trading Program. It is also the case with Pennsylvania’s $NO_x$ Allowance Retirement Program which is mandatory for fossil-fuel powered electric generating plants, but voluntary for others (Stavins 2001). Similarly, under RECLAIM it is possible for mobile sources and small point sources to volunteer to become involved (see Nash and Revesz 2000).

To a great extent allowing for voluntary adherence for some firms while preserving a core of firms for which the cap-and-trade programme is mandatory simplifies the decision for the firm since if the number of potential “voluntary” firms is small relative to the number of “mandatory” firms, the permit price can be taken as given. This also means that the regulator faces less uncertainty about the likely number of firms that are to be involved.

However, even in such cases voluntary adherence can raise concerns. The case of the $SO_2$ Allowance Program is instructive. Between 1996 and 1999 the percentage of emissions that were attributable to “opt-ins” was between 12% and 13% (www.epa.gov/airmarkets). However, Montero (2000) found that this »substitution« provision of the program tended to be taken up by those firms which were grandfathered emissions far in excess of actual emissions. An increase of one standard deviation in the firm’s allocation of permits relative to actual emissions increased the probability of “volunteering” from 32% to 84%. Indeed, the “adverse selection” effect dominated the effects of productive efficiency.

An important additional point relates to the treatment of existing “voluntary” commitments in the determination of permit allocations within TP schemes. In recent years, there have been extensive discussions in different programmes about the extent to which reductions achieved through formal “voluntary” approaches (negotiated agreements, etc…) should be included in the allocation of permits and in the evaluation of their baseline.




For instance, in the CEC's (2001) proposal for an allowance trading programme for GHGs it is stated that "the target set under the [negotiated] environmental agreements can serve as a useful basis for the allocation of allowances by Member States". This would, however, be politically difficult to achieve if the scope of the permit trading scheme is broader than the scope of the pre-existing agreement since firms which were not party to the agreement would benefit. More generally, this may raise the issue of »moral hazard«, making it exceedingly difficult for governments to negotiate agreements with firms in future due to the possibility of this affecting future permit allocations.

These ambiguities are even more important in credit-and-baseline schemes where credit creation is affected by the choice of the baseline. In some cases, the distinction may result in a switch from the firm being a net buyer rather than net seller of permits. This can be seen in the Figure, where for a given price of permits a firm shifts from being a net seller if voluntary commitments are not included to a net buyer if they are. In the Canadian Pilot Emission Reduction Trading Program, Trading Rule 2.4.3 states an emission reduction is surplus if it is not otherwise required of a source by current regulations or other obligations (e.g. a voluntary commitment). The precise meaning of a "voluntary commitment" was to be elaborated by a special Task Team. In their deliberations it was proposed that one required element for a "voluntary commitment" was that it includes a "negotiated agreement between an organization and the government and/or ENGO's such as a Memorandum of Understanding" (see Humphries 2000).

## Conclusions

In practice TP systems almost always co-exist with other environmental policy instruments. In some cases (i.e to protect local environmental conditions, to reduce compliance cost uncertainty, encourage additional abatement), a case can be made for their joint application. However, in other cases the secondary instrument will be at best redundant and at worst may result in increased administrative costs, increased economic inefficiency and reduced environmental effectiveness. Thus, whenever introducing a tradable permit system it is vitally important to understand the links with pre-existing policies. In some cases adjustments may need to be made to ensure complementarity. In other cases, it may be advisable to scrap the policy altogether.

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# TRADABLE PERMITS – TEN KEY DESIGN ISSUES

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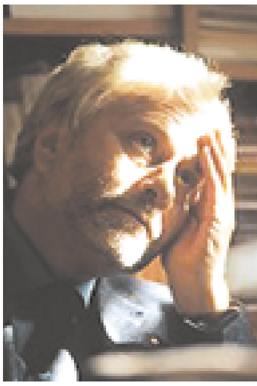

## Introduction

In this paper, we provide a guide for policymakers who consider using tradable permits as an environmental policy tool. Most of the issues we discuss are relevant both in the domestic and the international realm, although some have particular significance in one of the two areas.

In recent years, tradable permits (TP) have become rather widespread in use.[1] The table overleaf gives an overview of some of the numerous experiments, in particular in the US. There is a wealth of resources available that comment on the success of these programs (Stavins 2002). One noteworthy point is that the international experience is rather small. Europe has only relatively recently begun to develop such programs. For example, in Denmark, the Ministry of the Environment fixes annual emissions ceilings in the power generation industry, and leaves the actual allocation to the country's two power plant consortia. The UK allowed intra-firm trading of $SO_2$-allowances among large combustion plants from 1991 to 1997. But inter-firm trading was not allowed (Sorrell 1999).

The system in the Netherlands, where electric power producers face emissions standards for $SO_2$ and $NO_x$ but can comply through cost-sharing arrangements whereby plants with higher abatement costs are compensated, has resulted in intra-firm trading (Klaassen and Nentjes 1997). In Germany, the transfer of emission reduction obligations among firms in air quality non-attainment areas is allowed. The cost-savings have been estimated to be very limited (Schaerer 1994). The most recent experiment with market-based instruments is the UK Emission Trading Scheme, aimed at achieving the UK's commitment under the – yet to be ratified – Kyoto Protocol. Schneider and Wagner (2002) describe the program in detail. Since the first auction only took place in March 2002, and trading has been somewhat limited so far, it is too early to make an assessment concerning the success of the program.

What lessons can we learn from these programs, some of which have been "grand policy experiments" (Stavins 1998)? In this guide for policymakers, we focus exclusively on design and implementation issues and we draw on theoretical and empirical work on this question. Of course, there is no blueprint for the perfect system. It is our belief, however, that when tradable permit systems are used where they are appropriate, then heeding the lessons from the past increases the chances of the system leading to the desired outcome (in particular a cost-effective attainment of pre-set environmental goals). The balance of the paper deals with ten such key design issues.

## Trading of emissions versus inputs

In principle, we would want to regulate risks and impacts. However, it is quite difficult to trade risks directly. This is why policy typically moves one or two steps away from this level, leading to either *emission* permit trading or *input* permit trading. For example, a true $CO_2$ trading program would

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Part of this paper is based on Wagner and Schneider (2003).

[1] This should not obscure the fact, however, that tradeable permits are not the only game in town. In fact, important trade-offs with alternative environmental policy instruments need to be considered. For space reasons, it is not possible to adequately deal with these trade-offs here, and so we can only point the reader to the more extensive survey (Wagner and Schneider 2003) where questions like the optimal timing of environmental policy in the presence of significant uncertainties and irreversibilities and the relative merits of different policy instruments with respect to cost efficiency, environmental effectiveness, administrative practicability, dynamic efficiency and incentives for technological innovation, and political acceptability are discussed (Summary tables of the relative advantages and disadvantages can be found in the appendix of this paper). One particularly important insight developed recently in a number of papers (Abel et al. 1995; Arrow and Fisher 1974; Chao 1995; Dixit and Pindyck 1998; Hassett and Metcalf 1994; Kolstad 1992; Pindyck 2000) concerns the fact that policy decisions with respect to climate change are essentially irreversible and delay of action is possible. Under these conditions, waiting has optionality value; thus, the observed delay in climate policy implementations may at least partially be an optimal response to the prevailing uncertainties.

**Some selected tradeable permit systems (Stavins 2002)**

| Country | Program | Traded Commodity | Period of Operation | Environmental and Economic Effects |
|---|---|---|---|---|
| Canada | ODS Allowance Trading<br><br>PERT GERT | CFCs and Methyl Chloroform; HCFCs; Methyl Bromide<br>$NO_x$, $VOC_s$, CO, $SO_2$, $CO_2$ | 1993–1996; 1996–present; 1995–present; 1996–present; 1997–present | Low trading volume, except among large methyl bromide allowance holders |
| Chile | Santiago Air Emissions Trading | Total suspended particulates emission rights trading among stationary source | 1995–present | Low trading volume; decrease in emissions since 1997 not definitely tied to TP system |
| European Union | ODS Quotat Trading | ODS production quots under Montreal Protocol | 1991–1994 | More rapid phaseout of ODS |
| Singspore | ODS Permit Trading | Permits for use and distribution of ODS | 1991–present | Increase in permit prices; environmental benefits unknown |
| United Kingdom | Emissions Trading Program | $CO_2$ emissions | 2002–present | Unknown |
| United States | Emissions Trading under CAA | Criteria air pollutants | 1974–present | Performance uneffected; savings = $5–12 billion |
| | Lead Gasoline Phasedown | Rights for lead in gasoline among refineries | 1982–1987 | More rapid phaseout of leaded gasoline; $250 m annual savings |
| | Water Quality Trading | Point-nonpoint sources of nitrogen and phosphorus | 1984–1986 | No trading occurred because ambient standards not binding |
| | CFC Trades for Ozone Protection | Production rights for some CFCs, based on depletion potential | 1987–present | Environmental targets achieved ahead of schedule; effect of TP system unclear |
| | Heavy Duty Engine Trading | Averaging, banking, and trading of credits for $NO_x$ and particulate emissions | 1992–present | Standards achieved; cost savings unknown |
| | Acid Rain Reduction | $SO_2$ emission reduction credits; mainly among electric utilities | 1995–present | $SO_2$ reductions achieved ahead of schedule; savings of $1 billion/year |
| | RECLAM Program | $SO_2$ and $NO_x$ emissions among stationary sources | 1994–present | Unknown |
| | N.E. Ozone Transport | Primarily $NO_x$ emissions by large stationary sources | 1999–present | Unknown |

correspond to the first type; a carbon (content) trading program belongs to the second group. In general, the choice between the two depends on the degree of uniform mixing of the pollutant. For example, it would be problematic to have a sulfur-content trading program because $SO_2$ is a highly non-uniformly mixed pollutant – which is why the US has chosen to implement an $SO_2$ allowance trading program. Aside from this physical property, there is also an economic or political aspect: administrative feasibility. Clearly, the closer to the actual impacts regulation takes place, the more complex it gets. Taken together, these two factors suggest an important trade-off.

### Mandatory versus voluntary

Some observers have argued that a mandatory scheme is likely to be more environmentally effective. This is not necessarily true since significant emissions reductions may also be attained through voluntary participation. What is correct, however, is that mandatory schemes will in all likelihood be more cost-effective. Why? Under a voluntary scheme only entities that expect themselves to be sellers will join the scheme (even if they end up being buyers after all). In other words, there is a strong element of adverse selection involved, as has been shown for the case of the $SO_2$ program by Montero[2] (1999). Thus, abatement cost heterogeneity will be lower under a voluntary scheme, leading – for a given environmental goal – to lower cost-savings. Transaction costs for companies joining industrial opt-in programs have typically been high (Atkeson 1997).

### Absolute versus relative baselines

Typically, the difference between relative and absolute targets is argued to be as follows (Bode 2002): One limits total emissions to some absolute amount and may therefore limit "growth," while the other is presumed to impose less of a constraint on growth in output, albeit at the cost of some growth in emissions. As Ellerman (2002) points out, the U.S. experience with both systems does not provide much support for this distinction.[3] But it is not clear whether this experience is also relevant for the choice of baselines in climate change policy, for example. Indeed, one of the major components of the US Climate Plan announced in February 2002 is the concept of moving away from committing to a national emission cap by a specified date (such as is embodied in Kyoto) to a targeted rate of decline in emissions intensity of the economy. Kolstad (2002) argues that this part of the proposal does have some merit, on the grounds that it addresses the problem with the emissions cap approach of Kyoto that requires continual renegotiation of the caps as we proceed through time. It also does not have the (psychological and possibly real) effect of hindering growth for developing countries. Finally, an intensity target has the advantage of resolving some uncertainty, since other the absolute baseline significant cost uncertainty arises from a combination of uncertainty over how much an economy may grow by the time the commitment period arrives. The last word is still out on this issue.

Apart from this, two other reasons argue for using absolute baselines in national programs. The problem is that without a specified baseline, reductions must be credited to an unobservable hypothetical – what the source would have emitted in the absence of the regulation. Second, as was experienced with EPA's Emissions Trading Program, relative baselines create significant transaction costs by essentially requiring prior approval of trades as the authority investigates the claimed counterfactual from which reductions are calculated and credits generated (Nichols, Farr, and Hester 1996).

### Grandfathering versus auction

Almost all emission trading programs in action have started with grandfathered permits. For example, the most important emission trading program so far, the Clean Air Act amendments of 1990 dealing with $SO_2$ trading provide for annual auctions in addition to grandfathering – but such auctions involve less than three percent of the total allocation. Overall, the auctions have proven to be a trivial part of the overall program (Joskow, Schmalensee, and Bailey 1996). This is astounding since on the theoretical level, there seem to be compelling reasons for auctioned permits.

First of all, with perfect information and no transaction costs, trading will result in the economically efficient outcome independently of the initial distribution of permits (Montgomery 1972). Second, auctions are more cost-effective in the presence of certain kinds of transaction costs. Third, the revenue raised can be used to reduce other distortions (Goulder and Bovenberg 1996). Note also that while instruments such as tradable permits can create entry barriers that raise product prices, reduce the real wage, and exacerbate preexisting labor supply distortions, this effect can be offset if the government auctions the permits, retains the scarcity rents, and recycles the revenue by reducing distortionary labor taxes. Fourth, auctions provide greater incentives for firms to develop substitutes (see the section on technological progress). Fifth, due to the revenue raised by auctions, administrative agencies may have a bigger incentive to monitor compliance (Ackermann and Stewart 1985).

[2] However, the environmental effects must be kept in perspective. The number of allowances that could be considered excess amounted to only 3% of the total issued during 1995–1999 and the inflation of the cap during the time when these banked allowances will be used is only about 2%. Thus, these effects do not appear to have threatened the overall integrity of the allowance program.

[3] On the one hand, the consumption of coal has not been perceptibly reduced by the imposition of a cap on sulfur dioxide emissions. Rather, more low-sulfur coal is produced and a number of units have retrofitted scrubbers. On the other hand, the lead phasedown, which is the prototypical averaging (i.e. relative baseline) program, has not lead to more output of leaded gasoline.

Finally, grandfathering can lead unregulated firms to increase their emissions in order to maximize the pollution rights that they obtain if there is a transition to a market-based system (Dewees 1983). Overall, under almost any circumstances to be encountered in the real world, an auction of emission rights is preferable to grandfathering.

In addition to these considerations, questions of equity but also of dynamic efficiency will guide the treatment of new sources. Obviously, the decision will depend on the competitiveness of the market – the policy decision here is as much industrial policy as it is environmental policy.

**Allocations and efficiency in the international context**

Chichilnisky (1993) and Chichilnisky and Heal (1994) point out that the presumption that equal marginal abatement costs are the correct condition for efficiency is not strictly correct. The reason for this is that, simply, a dollar to a person in the developing world does not have the same welfare implications as a dollar to a developed world person. What matters are the real opportunity costs. Formally, the authors find that Pareto efficiency requires that the marginal cost of abatement in each country must be inversely related to that country's marginal valuation for the private good. This has strong policy implications: If richer countries have a lower marginal valuation of the private good, then at a Pareto-efficient allocation, they should have a larger marginal cost of abatement than the lower-income countries. With diminishing returns to abatement, this implies that they should push abatement further. Summarizing, the allocation of property rights in a tradable permit system is important if environmental quality has a direct impact on wellbeing and marginal valuations of private goods differ strongly across countries.

The main policy implication for the design of efficient permit trading programs concerns the allocation of rights. Even after choosing to go with tradable permits as the environmental policy instrument, we need to carefully use the degree of freedom left in terms of the distribution of property rights.[4] Whenever politicians bring up equity issues, economists are quick to point out that those have nothing to do with efficiency. For once it seems that politicians are right, if not in their reasoning.

**Banking and borrowing**

The US has had significant experience with programs that allow intertemporal trading, in particular banking. Two lessons emerge from this experience (Ellerman 2002): First, when allowed and coupled with a phased-in reduction requirement, banking will be used and it will accelerate the timing of emission reductions. Studies of the US Acid Rain Program also find that firms have learned very well how to optimally accumulate and draw down banks (Ellerman and Montero 2002). Second, the ability of banking to dampen allowance price fluctuations may be important when the spatial scope of the cap is limited.[5] In fact, this second point hints at the importance of a temporal safety valve that may allow agents to borrow in times of extraordinary demand. Of course, there is good reason to restrict temporal flexibility when the environmental problem is other than a stock pollutant.

**Market power and the design of emission permit markets**

In order for cost minimization gains to be fully realized, the emission trading market must work in a competitive manner. If some agents have the capacity to influence the transaction price of traded permits or can prevent the entrance of competitors by hoarding permits, efficiency losses may ensue (OECD 2001). For example, Hahn (1984a) shows that the deviation of abatement costs from the cost minimum is related to the extent to which the initial distribution of permits differs from the equilibrium distribution (and to the price elasticity of demand).

Another type of strategic behavior occurs if firms use the permit market to drive up rivals' costs (exclusionary manipulation). Note first that this can only occur if firms operating in the same industry also participate in the same permit market. Misiolek and Elder (1989) conclude that, surpris-

[4] Chichilnisky et al. (2000) concentrate on the first welfare theorem in markets in which agents trade, at a uniform price (that is, not at personalized Lindahl prices), permits to produce privately produced public goods. They take the total quantity of permits fixed by the government at a level consistent with Pareto efficiency. They show that the equilibria are nevertheless generally inefficient, due to the public good character of one of the traded goods. But the main surprise is that there exist certain allocations of rights to emit from which the market overcomes the »free rider« problem and achieves efficiency. This is a key characteristic of competitive markets for privately produced public goods.
[5] This was important to bring price levels back to normal in the RECLAIM NOx program in the US after the California electricity market crises in late 2000 and early 2001.

ingly, this may not necessarily have a negative impact on cost efficiency. It is unclear to what extent this result survives the inclusion of uncertainty. Experimental studies and anecdotal evidence from existing permit markets suggest that this is probably not a major problem – at least for domestic programs. On the international level, things may look different. As regards carbon trading, a particularly important danger seems to be that Russia and the Ukraine exert market power. In a first attempt to estimate the costs of such a situation, Burniaux (1999) finds that by 2010 the price of Assigned Amount Units (the term for emission permits that the Kyoto protocol uses) would be about 20 per cent higher than under the competitive scenario (for a discussion see OECD (2001)). Clearly, the best way to avoid such situations is for governments to devolve their assigned amounts to their legal entities and promote industry-level trading (Bader 1996; Hahn 1984b).

**Market efficiency, transaction costs**

If we want to rely on environmental markets to give us efficient results, we must be able to rely on them in providing informational or market efficiency first. One key to a smooth functioning of the tradable permit market is a low level of transaction costs.

Three potential sources of transaction costs in tradable permit markets can be identified: (1) search and information; (2) bargaining and decision (Dwyer 1992; Kohn 1991); and (3) monitoring and enforcement. Anecdotal evidence abounds regarding the prevalence of significant transaction costs in tradable permit markets. Atkinson and Tietenberg (1991) surveyed six empirical studies that found trading levels in permit markets to be lower than anticipated by theoretical models. On the other hand, it has been recognized that success stories like the EPA's leaded gasoline phasedown can partially be attributed to the program's minimal administrative requirements and the fact that the potential trading partners (refineries) were already experienced at striking deals with one another.[6] Transaction costs in the $SO_2$ market in the US – the most successful TP market – are now minimal. The lesson for policymakers is to make administrative procedures as simple as possible and to equip potential trading partners with means to efficiently communicate market-relevant information with each other.[7]

A final word is in order on the international realm. When governments themselves trade, transactions could be the result of bilateral bargaining where emission permits are not the only element of the transaction; in other words, governments will in general be motivated by other factors than strict economic ones. Prior notification by parties and, more generally, the establishment of specific exchanges has been advocated to promote competitive behavior (Bohm 1998). First experiments (Hizen and Saijo 1999) seem to indicate, however, that disclosure of contract information does generally not improve market efficiency. Similarly, trading through an exchange does not seem to improve significantly the efficiency of the trading regime as opposed to bilateral trading. These results are surprising and merit further investigation.

**Enforcement and management framework**

There are two aspects to an enforcement framework: One is the monitoring of compliance with the regulatory framework and detecting violations. The other is responding to violations in a way that ensures that it is always in the interests of participants to comply. Often, the first aspect is the simpler of the two. For example, for $CO_2$, since it is a mostly uniformly mixed pollutant, we do not have to monitor each and every source of $CO_2$ emissions, but can focus on the sales of the major distributors of carbon-based fuels. In fact, just from such sources, estimates of the consumption of various carbon-based fuels in each country are already available from data on production, import, export, and inventories.[8]

The enforcement poses much more serious problems, in particular in the international context. Malik (1990) demonstrates that with imperfect compliance, firms set the level of emissions such that marginal profits equal the permit price plus the expected fine. It can also be shown that if the marginal penalty of noncompliance is constant, tradable emission permits lead to less noncompliance than does regulation. With increasing margin-

[6] For an overview of quantitative empirical estimates across various programs, we refer the reader to Wagner and Schneider (2003)
[7] Not only the level of transaction costs is important. Stavins (1995) shows that when transaction costs are dependent on the volume traded, this may imply that the final equilibrium, and hence cost efficiency, is no longer independent of the initial distribution of permits (the precise result depends on the exact shape of transaction costs)..
[8] It should be noted that if the lives of quotas are not synchronized – if they specify a total of emissions over a multiyear life – matters could be more difficult.

al penalties (as a function of the violation), all firms will comply if the permit price below the expected per unit violation penalty. With decreasing marginal penalties, firms that decide not to comply will pollute more than under regulation. In sum, with imperfect enforcement, whether or not tradable permits meet the environmental goal depends on the structure of the penalty function.

With respect to market management more generally, the clear recommendation from economic theory is to allow market participants to fully exploit cost-saving opportunities and risk-management possibilities, for example through the use of derivatives (as they are already traded in the $SO_2$ and $NO_x$ allowance markets in the US). In addition to facilitating heding price risks, derivatives also help achieve market depth and liquidiy and so improve market functioning.

**Interaction between international and domestic policies and needs**

Sometimes it is argued that it does not matter how countries enforce given total emission levels domestically, as long as the allocation of quotas among countries is clear "...in principle, any domestic policy regime is possible." (Chichilnisky and Heal 2000). Hahn and Stavins (1999) deal critically with this important point, which has received surprisingly little attention in the literature on international environmental agreements.

They start from the observation that the Kyoto Protocol's greenhouse gas trading mechanism will lead to minimized costs if all countries use domestic tradable permit systems to meet their national targets and allow for international trades. Thus, the European Union's proposal to introduce a trading system within Europe to fulfull the requirements of Kyoto, indeed is very important for the overall performance of Kyoto's system. By contrast, political practice suggests that many countries will use non-trading approaches such as greenhouse-gas taxes or fixed quantity standards. Hahn and Stavins show that in these cases, achieving the potential cost savings of international trading requires some form of project-by-project credit program – like joint implementation. However, large transaction costs, likely government participation, and absence of a well-functioning market may be obstacles for this toute. Overall, there is an important trade-off between the degree of domestic sovereignty and the degree of cost-effectiveness.

A related question is how to link existing schemes, for example the Danish and the UK $CO_2$ schemes (Bode 2002). Again, as long as the abatement costs in separated trading schemes are different, the linkage of two schemes can result in increased overall cost-effectiveness. There will be equity considerations, however, since prices will change compared to the previous equilibrium. This may raise resistance by the loosing participants in advance of the linking of schemes. Bode (2002) discusses in detail how the linkage of schemes and differences in design features like those discussed in the present paper interact with each other. Obviously, there are also often difficult legal issues involved (Rodi 2002).

**Summary**

Tradable permit programs have been in use in the United States for a long time and are also on their way to becoming a very popular environmental policy instrument in Europe. This guide has aimed to highlight ten of the most important issues in designing a successful tradable permit program.

1. The choice of trading of emissions versus trading of inputs (e.g. $CO_2$ trading versus carbon content trading) depends on the degree to which the pollutant is uniformly mixed.
2. In most instances, mandatory schemes will be more cost-effective. They avoid adverse selection problems in participation.
3. Many arguments speak for the use of absolute baselines in national programs. We have also pointed out, however, that the concept of targeting a decline in $CO_2$ emissions intensity in the economy may have some merit.
4. The clear economic advice is to auction off permits instead of grandfathering them. Of course, political feasibility considerations will often make this impossible.
5. Initial allocations may be important for efficiency when there is a high degree of inequality in wealth between the trading entities, for example, in the international context.
6. Temporal flexibility should be allowed to as large extent as environmental effectiveness allows it.
7. The market management authority needs to be careful to avoid anti-competitive behavior on the market, although existing studies seem to indicate that strategic behavior on tradable

permit markets is not an important phenomenon.

8. Participating firms and other entities must have the ability to quickly communicate in order to keep transactions costs low.
9. Continual monitoring of compliance and enforcement of the "rules of the game" of a tradable permit program are essential ingredients in reducing uncertainty for market participants and to secure environmental effectiveness.
10. The design of national emissions programs in the presence of international agreements is difficult. Linking existing schemes inevitably produces losers who may need to be compensated.

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## Appendix

**Table A1:**
**Instruments of environmental policy and criteria to evaluate them**

| Dimension | Instrument Emission Charges | Tradeable permits | Regulation |
|---|---|---|---|
| Cost efficiency | + | + | – |
| Environmental effectiveness | – | + | + |
| Administrative practicability | + | + | + |
| Dynamic efficiency | + | + | 0 |
| Political acceptability | 0 | 0/+ | + |

"+" = high, "–" = low, "0" = neutral.

Source: Klaassen (1996), Wagner and Schneider (2003).

**Table A2:**
**Conditions affecting cost efficiency and environmental effectiveness**

| | Cost efficiency | | | Environmental effectiveness | | |
|---|---|---|---|---|---|---|
| | Charges | Permits | Regulation | Charges | Permits | Regulation |
| Uncertainty about costs | – | 0 | – | – | 0 | ? |
| Imperfect markets | – | – | ? | – | 0 | ? |
| Transaction costs | 0 | – | 0 | 0 | 0 | 0 |
| Imperfect enforcement | 0 | – | ? | 0/– | – | – |
| Discontinuous control | 0 | 0 | – | – | 0 | 0 |
| Cost-saving techn. Progress | – | 0 | ? | ? | 0 | 0 |
| Economic growth | 0 | 0 | 0 | – | 0 | – |
| Inflation | 0 | 0 | 0 | – | 0 | 0 |

"-" = negative impact; "0" = no impact; "?" = unknown.

Source: Klaassen (1996), Wagner and Schneider (2003).

# Tradable Permits with Imperfect Monitoring

Juan-Pablo Montero*

## Introduction

In recent years environmental policy makers have been paying more attention to tradable permits (or emissions trading) as an alternative to the traditional command-and-control (CAC) approach of setting emission and technology standards. A notable example is the 1990 U.S. Acid Rain program that implemented a nationwide market for electric utilities' sulfur dioxide ($SO_2$) emissions (Schmalensee et al., 1998; Ellerman et al., 2000). In order to have a precise estimate of the $SO_2$ emissions that are going to the atmosphere, the Acid Rain program requires each affected electric utility unit to install costly equipment that can continuously monitor emissions. Another example with similar monitoring requirements is the Southern California RECLAIM program that implemented separated markets for nitrogen oxide ($NO_x$) and $SO_2$ emissions from power plants, refineries and other large stationary sources.[1]

These and other market experiences suggest that conventional tradable permits programs are likely to be implemented in those cases where emissions can be closely monitored, which almost exclusively occurs in large stationary sources like electric power plants and refineries. At least this is consistent with the evidence that environmental authorities continue relying on CAC instruments to regulate emissions from smaller sources for which continuous monitoring is prohibitively costly (or technically unfeasible). Although CAC regulation for smaller sources does not directly target emissions either (the regulated source must install some required abatement technology or set its emissions per unit of output equal or lower than a certain emissions standard), some regulators believe that a permits program in which emissions are not closely monitored may result in even higher emissions than under an alternative CAC regulation because permits provide firms with more flexibility to choose output and emissions.

Thus, it appears at first that permits markets are not suitable for effectively reducing air pollution in cities such as Santiago-Chile or Mexico City where emissions come from many small (stationary and mobile) sources rather than a few large stationary sources. It would be prohibitively costly, for example, to require operators of central heating systems in residential or commercial buildings to install continuous emission monitoring equipment. Through annual inspections, however, the regulator could monitor boilers' combustion technology, fuel type, emissions rate and size, as he would precisely do under CAC regulation. But since the regulator does not observe the total number of hours boilers are operated during the year, he would certainly have imperfect estimates of boilers' actual emissions.

**May tradable permits be used to reduce air pollution in big cities?**

Rather than disregard tradable permits markets as a policy tool, I think the challenge faced by policy makers in cities suffering similar air quality problems is to find out when and how to implement these markets using approximate monitoring procedures similar to those under CAC regulation. While the literature provides little guidance on how to approach this challenge, it is interesting to observe that despite its incomplete information on each source's actual emissions, Santiago-Chile's environmental agency has already implemented a tradable permits market to control total suspended particulate (TSP) emissions from a group of about 600 stationary sources (Montero et al., 2002). Based on estimates from annual inspection for technology parameters such as source's size and fuel type, the regulator approximates each source's

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[1] It is worth noting that RECLAIM did not include a market for volatile organic compounds (VOC) in large part because of the difficulties with monitoring actual emissions from smaller and heterogeneous sources (Harrison, 1999).

actual emissions by the maximum amount of emissions that the source could potentially emit in a given year.

Motivated by Santiago's emissions trading experiment, in a recent paper I provide a theoretical and empirical evaluation of the advantages of tradable permits over CAC regulation under imperfect monitoring (Montero, 2003). The purpose of this note is to communicate the main results and policy implications of this study.

### Some theory

A model to evalute the effects of tradable permits vs. CAC regulations under imperfect monitoring

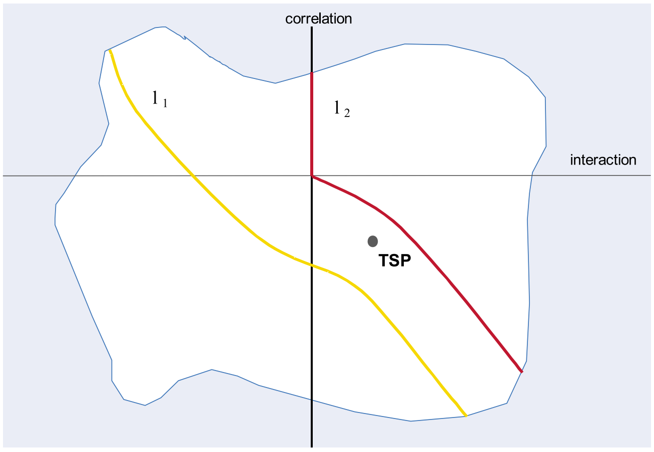
It is well known that, when emissions can be closely monitored, a tradable permits program can provide important cost savings over an alternative CAC regulation (Tietenberg, 1985). It is not clear, however, whether permits can still provide an important welfare advantage when emissions are imperfectly monitored. To answer this question I develop a theoretical model and I compare social welfare under the two (optimally designed) policies: technology (or emission rate) standard and tradable permits. Since the regulator is assumed to observe only the firm's abatement technology or emission rate but not its actual emissions, in order to implement the permits policy the regulator must use some proxy for emissions. For example, as in Santiago's trading program, he could proxy emissions by the emissions that the source would emit if it operated its production facilities without interruption throughout the year (sources in Santiago's program operate, on average, less than half of the time).[2]

The theoretical model provides important results that can be tested with the data. In fact, I find that permits policy provide firms not only with flexibility to choose production and abatement possibilities (the cost savings effect) but sometimes with incentives to choose socially suboptimal combinations of output and abatement; something that would not occur if emissions were accurately measured. The misalignment between private and social incentives occurs because the regulator neither observes emissions nor hours of operation (or output), so the permits policy can prompt changes in output that can lead to higher emissions. There are two cases in which the incentives misalignment can happen. The first case is when firms with relatively large output ex-ante (i.e., before the regulation) are choosing low abatement (i.e., when there is a negative correlation between production and abatement costs). The second case is when firms doing little abatement find it optimal to increase output ex-post (i.e., when there is a negative interaction between output and abatement).

While the cost savings effect is always positive (i.e., the permits policy is always cheaper than the standards policy), the correlation and interaction effects can be either positive or negative. When either one or both of these latter two effects are negative, the superiority of the permits policy over the standards policy is no longer evident. The size and sign of these three effects is an empirical matter that will ultimately depend on the cost structure of the specific industry (or group of sources) that is going to be regulated. Generated from simple but reasonable parameter values, the Figure provides an illustration of how the correlation and interaction effects affect the relative advantage of permits over standards. The permits policy is welfare superior for all those combinations to the right of line $l_1$. When there are no correlation and interaction effects the permits policy is unambiguously superior to the standards policy.

Because in deciding whether to use permits or standards, the regulator is likely to face a trade-off

INTERACTION AND CORRELATION EFFECTS




[2] It is important to explain that using as a proxy half of the maximum emissions would work equally well because the regulator would then adjust (i.e., increase) the number of permits accordingly. See Montero (2003) for more details.

between cost savings and possible higher emissions, it seems relevant to discuss the advantages of implementing a hybrid policy in which permits are combined with some (optimally chosen) standard. While the hybrid policy should not be inferior to either single instrument policy, I find that in many situations the hybrid policy converges to the permits-alone policy but it almost never converges to the standards-alone policy. In fact, for all those cost structures in which the correlation and interaction effects fall to the right of line $l_2$ in the Figure, the hybrid policy converges to the permits-alone policy, i.e., the inclusion of a binding standard would decrease welfare.

## Some empirical evidence

The theoretical results indicate that whether the permits policy provides higher welfare than a standards policy is an empirical question. I use the experience from Santiago's total suspended particulate emissions (TSP) trading program to answer this question. The TSP trading program, established in March of 1992 and effective since 1994, was designed to curb TSP emissions from the largest stationary sources in Santiago (industrial boilers, industrial ovens, and large residential and commercial heaters). Because sources were too small to require sophisticated monitoring procedures, the authority did not design the program based on sources' actual emissions but on a proxy variable equal to the maximum emissions that a source could emit in a given period of time if it operated without interruption.

The proxy variable (expressed in kg of TSP per day) used by the authority in this particular program was defined as the product of emissions concentration (in mg/m$^3$) and flow rate (in m$^3$/hrs) of the gas exiting the source's stack. Although the regulatory authority monitors each affected source's concentration and flow rate once a year, emissions and permits are expressed in daily terms to be compatible with the daily TSP air quality standards. Thus, a source that holds one permit has the right to emit a maximum of 1 kg of TSP per day indefinitely over the lifetime of the program.

Sources registered and operating by March 1992 were designated as existing sources and received grandfathered permits equal to the product of an emissions rate of 56 mg/m$^3$ and their maximum flow rate at the moment of registration. New sources, on the other hand, receive no permits, so must cover all their emissions with permits bought from existing sources. The total number of permits distributed (i.e. the emissions cap) was 64 percent of aggregate emissions from existing sources prior to the program. After each annual inspection, the authority proceeds to reconcile the estimated quasi-emissions with the number of permits held by each source (all permits are traded at a 1:1 ratio). Note that although permits are expressed in daily terms, the monitoring frequency restricts sources to trade permits only on an annual or permanent basis.

Because firms are not required to provide the regulator with information on production and abatement costs, to empirically recover the cost structure of the industry and test the advantages of the TSP program I apply the theoretical framework to information other than cost such as emission rates and utilization (hours of operation). The Table presents a summary of the data used in the empirical study for selected years. The first two rows show that the exit and entry of sources has been quite significant. By 1999, 36 percent of the affected sources were new sources despite the fact that they did not receive any permits.

The model is applied to emission rates and utilization

In order to comply with the TSP trading program, affected sources can hold permits, reduce emissions or do both. They can reduce emissions by either switching fuel (for example, from wood, coal, or heavy oil to light oil, liquid gas, or natural gas) or installing end-of-pipe technology such as filters, electrostatic precipitators, cyclones, and scrubbers. Sources do not gain anything, in terms of emissions reduction, by changing their utilization level (i.e. days and hours of operation), because by definition it is assumed to be at 100 percent.

The next rows of the Table show data on emission rates and utilization. The large standard deviations show that these variables vary widely across sources in all years. As the 1993 numbers indicate, sources' utilization was quite heterogeneous before the implementation of the program, indicating some potential for higher emissions under a permits policy. The Table also indicates that the emissions rates of affected sources has remained quite different across sources after the program became effective. This compliance heterogeneity

**Summary statistics for affected sources in selected years**

| Variable | 1993 | 1995 | 1997 | 1999 |
|---|---|---|---|---|
| No. of sources | | | | |
| Existing | 635 | 578 | 430 | 365 |
| New | 45 | 112 | 146 | 208 |
| Total affected | 680 | 690 | 576 | 573 |
| Emission rate (mg/$m^3$) | | | | |
| Average | 94.9 | 83.1 | 54.7 | 27.8 |
| Standard dev. | 88.1 | 77.8 | 43.0 | 18.5 |
| Max. | 702.0 | 698.2 | 330.7 | 108.2 |
| Min. | 1.5 | 1.5 | 3.6 | 4.6 |
| Utilization (%) | | | | |
| Average | 39.4 | 48.0 | 49.2 | 53.7 |
| Standard dev. | 30.3 | 31.5 | 31.8 | 32.3 |
| Max. | 100 | 100 | 100 | 100 |
| Min. | 0 | 0 | 0 | 0 |
| Total emissions (kg/day) | 7,051.9 | 6,320.9 | 3,535.0 | 1,665.0 |
| Total permits (kg/day) | 4,604.1 | 4,604.1 | 4,087.5 | 4,087.5 |

Notes: A utilization of 100 percent corresponds to 24 hrs of operation during 365 days. Utilization figures are based on most but not all sources. Information on utilization is not required for monitoring and enforcement purposes.

**Capturing the cost structure: permits are superior to regulation**

confirms that, contrary to what occurs under CAC regulation where all firms must either install the same abatement technology or comply with the same emission rate, permits provide enough flexibility for sources to comply in very different ways.

The last two rows of the Table show data on emissions and permits.[3] Although 1994 was in principle the first year of compliance with the program, trading activity did not occur until the end of 1996 because of evident enforcement problems. The emissions goal of the TSP program was only achieved by 1997 (total emissions below total permits). This was the year after which natural gas became available from Argentina at unexpectedly attractive prices so that many affected sources switched to this cleaner fuel leaving the cap of 4,087.5 permits largely unbinding. This is consistent with the fact that all TSP trading activity took place from the end of 1996 to the middle of 1998 with prices steadily declining from 17,000 to 3,000 US$/permit.[4] For these reasons, most of the empirical analysis is based on the 1997 data.

Using the data summarized in the Table, I then proceed to capture the cost structure of the group of sources affected by the TSP program. Econometric estimations indicate that while the interaction effect is positive (i.e. firms doing more abatement are also increasing output relative to sources doing less abatement), the correlation effect is negative (i.e. sources more heavily utilized are doing less abatement). These two effects almost offset each other. I find only a mild increase in emissions, if any, compared to what would have been observed under an equivalent standards policy. Furthermore, because cost savings are found to be substantial (explained by the significant heterogeneity in emission rates shown in the Table), the permits policy is found to be superior.

In terms of the Figure, the dot "TSP" provides a good illustration of the cost structure of the sources affected by the TSP program, which suggests some potential gains from implementing a hybrid policy. Preliminary estimates based on the 1997 data indicate that the combination of a slightly larger fraction of permits with an optimally chosen standard could add some extra 10 percent of benefits.

## Conclusions

When emissions cannot be closely monitored, the environmental regulator will inevitably face a tradeoff between abatement flexibility and output and abatement misallocation in deciding whether or not to implement a permits policy instead of a traditional standards policy. Because these misallocations can lead to higher emissions, I do find situations in which a standards policy can be welfare superior. However, when I used emissions and output data from Santiago's TSP emissions trading program to test for this possibility I found no evidence. Conversely, I found conclusive evidence that the production and abatement cost characteristics of the sources affected by the TSP program are such that the permits policy is unambiguously welfare superior because not only does it lead to significant cost savings but also to virtually the same aggregate emissions than under an equivalent standards policy.

[3] A few permits were retired from the market in 1997 as the authority revised the eligibility of some sources for receiving permits (Montero et al., 2002).
[4] Obviously, intra-firm trading has continued as new sources are coming into operation.

The superiority of the permits policy is due in large part to the fact that sources making larger emission reductions are also increasing their utilization relative to other sources. This behavior seems to be more general than one may think. Firms choosing abatement investments with proportionally large fixed/sunk costs (e.g., installing end-of-pipe technologies) not only make larger reductions but also enjoy lower ex-post marginal abatement costs (ex-ante marginal abatement costs should be similar at the margin), so their ex-post marginal production cost is relatively lower, and hence, their utilization relatively higher.[5]

In conclusion, the theoretical and empirical results discussed here make a strong case for the wider use of environmental markets even in those situations in which emissions are imperfectly observed. In the particular case of Santiago, these results suggest that using simple monitoring procedures it is possible and economically sound to expand the TSP trading program (which now is responsible for less than 5 percent of TSP emissions in Santiago) to other sources that are currently not regulated or subject to costly CAC regulation such as smaller stationary sources and industrial processes (both responsible for 27.0 percent of TSP in 2000), power diesel buses (36.7 percent), trucks (24.7 percent) and smaller commercial vehicles and cars (6.8 percent). A similar approach can also be used for regulated other pollutants such as $NO_x$.

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[5] The $SO_2$ trading system of the 1990 US acid rain program provides strong evidence on this as well. Affected sources retrofitted with scrubbers (end-of-pipe technologies that can reduce up to 95 percent of the emissions coming out of the stack) experienced a noticeably increase in utilization relative to affected sources that switched to lower sulfur coals or simply did not abate emissions (Ellerman et al., 2000, pp. 334–341).

# Emissions Trading with Greenhouse Gases in the European Union

Johann Wackerbauer*

The EU will allow $CO_2$ emissions trading at industry level from 2005

In the 1997 Kyoto Protocol, 38 developed countries (plus the EU) accepted legally binding reductions of greenhouse gas emissions of at least 5 percent over the period 1990 to 2008–12. The European Union has committed itself to an even higher reduction of 8 percent within this time framework. To provide flexibility, the Kyoto Protocol permits the transfer or exchange of emissions reductions among the signatory countries via so-called flexible mechanisms. Industrialised countries may transfer or acquire from each other emission reductions on a project basis through Joint Implementation (JI). The Clean Development Mechanism (CDM) allows emissions credits to be obtained from projects undertaken in developing countries. Finally, the Kyoto Protocol marks the creation of an international emissions trading (IET) system among the signatory states (Galeotti et al. 2001). Furthermore, the Kyoto Protocol allows a group of countries to have an aggregate target by setting up a bubble. The countries of the European Union did this in their burden-sharing agreement, the EU Bubble being the first of its kind. Because of this common commitment, emissions trading between the members of the European Union and between entities within those countries is regarded as "domestic action" (Egenhofer 2001).

In March 2000, the European Commission adopted a Green Paper on greenhouse gas emissions trading within the EU and launched a debate on the introduction of this market-based instrument. In October 2001, the Commission submitted a proposal for an EU greenhouse gas emissions trading system. In December 2002, the Council unanimously reached political agreement on a common position on the Commission's proposal. The proposal covers greenhouse gas emissions trading for the European Union at industry level which is in contrast to the Kyoto Protocol, which allows international emissions trading only at the state level. According to the EU scheme, the total quantity of greenhouse gas emissions will be limited and installations will be able to engage in Community-wide emissions trading. All installations covered by the scheme will have to apply for a greenhouse gas "permit" that requires adequate monitoring and reporting of emissions. Furthermore, to emit a certain quantity of greenhouse gases, operators must possess corresponding greenhouse gas "allowances", denominated in metric tonnes of carbon dioxide equivalent. The allowances will be transferable and may be traded between companies. The first trading period will be from 2005 to 2007, preceding the Kyoto Protocol's commitment period. In this first phase, only $CO_2$ emissions will be covered by the scheme. The next trading period will coincide with the Kyoto Protocol's commitment period of 2008 to 2012. Member states will allocate allowances in each trading period such that total emissions are not higher than if they were regulated by the IPPC Guideline (EU Commission 2001). The scheme will be applied to most of the significant greenhouse gas emitting activities that are already covered by the IPPC Directive as well as some installations not covered (see Table). According to the Commission, some 4,000 to 5,000 installations will be regulated by the Directive covering approximately 46 percent of estimated EU carbon dioxide emissions in 2010. The chemical sector is excluded because its direct emissions of carbon dioxide are less than one percent of the EU's total emissions, and the number of chemical installations in the Community, in the order of 34,000 plants, will increase the administrative complexity of the scheme. The waste incineration sector is excluded due to problems of measuring the carbon content of the waste material that is being burnt. However, carbon dioxide emissions from any on-site power and heat generating facility will be included if it exceeds the threshold of 20 MW.

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**Activities covered by the Commission's proposal**

| |
|---|
| Energy activities<br>⇨ Combustion installations with a rated thermal input exceeding 20 MW (excepted: hazardous or municipal waste installations)<br>⇨ Mineral oil refineries<br>⇨ Coke ovens |
| Production and processing of ferrous metals<br>⇨ Metal ore (including sulphide ore) roasting or sintering installations<br>⇨ Installations for the production of pig iron or steele (primary or secondary fusion) including continuous casting, with a capacity exceeding 2.5 tonnes per hour |
| Mineral industry<br>⇨ Installations for the production of cement clinker in rotary kilns with a production capacity exceeding 500 tonnes per day or lime in rotary kilns with a production capacity exceeding 50 tonnes per day or in other furnaces with a production capacity exceeding 50 tonnes per day<br>⇨ Installations for the manufacture of glass including glass fibre with a melting capacity exceeding 20 tones per day<br>⇨ Installations for the manufacture of ceramic products by firing, in particular roofing tiles, bricks, refractory bricks, tiles, stoneware or porcelain, with a productions capacity exceeding 75 tonnes per day, and/or with a kiln capacity exceeding 4 $m^3$ and with a setting density per kiln exceeding 300 kg/$m^3$ |
| Other activities<br>⇨ Industrial plants for the production of<br>(a) pulp from timber or other fibrous materials<br>(b) paper and board with a production capacity exceeding 20 tonnes per day |

Source: Commission of the European Communities, COM(2001)581, Annex 1.

## The compatibility of emissions trading with traditional environmental instruments

Theoretical considerations suggest that emissions trading has considerable economic advantages over the use of other instruments to combat greenhouse gas emissions and meet the Kyoto target. Evidence also suggests that the more widely emissions trading is applied, the higher the economic benefits. This contrasts with the actual situation of environmental policy resting on "traditional" instruments, i.e. regulation, voluntary agreements, and taxation.

In principle, there are two categories of environmental policy instruments serving the control of greenhouse gas emissions: Direct regulation, also referred to as command-and-control instruments, on the one hand, and instruments providing incentives for climate-friendly behaviour, also referred to as market-based instruments, on the other. Standards on specific emissions or energy efficiency are examples of the first category, taxes and subsidies, but also tradable permits belong to the second category. With respect to climate change policy, voluntary agreements and project-based instruments have to be added to this list. Voluntary agreements are not easy to classify: Insofar as they imply a commitment to the reduction of greenhouse gas emissions, they resemble a regulatory instrument. Insofar as financial incentives are given to individual firms for joining the voluntary agreement, they approach market-based instruments. In fact, they cannot be clearly assigned to one of the two categories but rather resemble a corporatistic approach (Remings et al. 1996). Last, but not least, project-based instruments are new investments in technical projects that have environmental advantages. Being voluntary, they cannot be regarded as command-and-control instruments; because of existing incentives for minimising the cost of reducing emissions, they are more or less market-based.

**Market-based instruments are superior to command-and-control instruments**

Economic theory shows that market-based instruments are superior to command-and-control instruments because they minimise the costs of emission abatement by leaving it up to the plant operator whether to apply expensive technologies or to opt for paying fees or buying tradable permits. Technical standards tend to increase costs because they may impose high expenditures on single firms for complying with the regulation. In contrast to taxes and subsidies, an emissions trading scheme provides certainty of the environmental outcome if a cap is imposed on total emissions.

For a long time, environmental policy has relied on command-and-control instruments and on subsidies for environmentally friendly behaviour. In recent years, some countries have introduced eco(logy)-taxes. Denmark, Norway, Sweden and the Netherlands introduced $CO_2$ taxes; Belgium, Finland and Germany imposed additional energy taxes to encourage emission abatement (Osterkamp 2001). Voluntary agreements were part of pre-Kyoto climate change policy in Finland (energy conservation agreements), the Netherlands (long-term energy efficiency agreements), Sweden (eco-energy programme), France (agreements on $CO_2$ reduction and energy efficiency), Denmark ($CO_2$ emission abatement), the United Kingdom (agreement on energy efficiency improvement) and Germany (declaration by German Industry on global warming). Since the Kyoto Protocol (December 1997), new commitments have been agreed in Switzerland (Action Programme Energy 2000) and in Italy (climate pact between government, industry and NGOs). The UK introduced a Climate Change Levy (CCL) that defines a group of voluntary commitments as complementary measures and Germany amended the declaration on global warming. On the level of the European Union, the Commission and the automotive industry agreed on the reduction of specific $CO_2$ emissions of new cars (Jones et al. 2001). With respect to these manifold instruments adopted for the reduction of greenhouse gas emissions, including standards, taxes, voluntary agreements and emissions trading, attention must be paid to the extent to which they overlap and whether conflicts between them are to be expected.

*Emissions trading and command-and-control instruments*

**In order to combine command-and-control instruments and emissions trading the IPPC rules will have to be modified**

The IPPC Directive on integrated pollution and prevention control[1] is the backbone of the regulations regarding stationary pollutants in the European Union. It requires that installations be operated in such a way that all appropriate preventive measures are taken against pollution, in particular the application of the best available techniques (BAT) and the efficient use of energy (Rehbinder and Schmalholz 2002). In principle, both technology standards and energy efficiency standards are incompatible with emissions trading because they do not allow the operator of an installation to choose between applying the BAT or buying tradable permits. However, the IPPC Directive does not yet cover any of the six greenhouse gases. Methane ($CH_4$), dinitrogen monoxide ($N_2O$), hydrofluorocarbons (HFC), perfluorinated hydrocarbons (PFC) and sulphur hexafluoride ($SF_6$) are listed as harmful substances in Annex 3 of the IPPC guideline, but there are no emission standards imposed on them. Carbon dioxide is only implicitly regulated by the energy efficiency requirements of the IPPC guideline (Freshfields Bruckhaus Deringer 2002).

Following the proposal on emissions trading, the granting of permits for greenhouse gas emissions will have to be based on the procedures under the IPPC Directive. But in contrast to other IPPC regulations, such a permit would only require the operator to hold a sufficient number of allowances to cover the installation's emission in a given period and not limit its direct emissions of carbon dioxide or other greenhouse gases except as they may have significant local effects. If the IPPC guideline and corresponding national regulations were modified such that the principle of preventive action and requirements on energy efficiency are not applied to emissions subject to a trading scheme, existing command-and-control instruments could be combined with emissions trading.

*Emissions trading and voluntary agreements*

Industry associations within the European Union have expressed their strong preference for long-term voluntary agreements as the prime instrument for the pursuit of climate policy goals. In most cases, voluntary agreements are based on specific targets expressed in emissions per output or energy use per output. At first glance, such relative industry targets are incompatible with national absolute targets because growth of industrial production can result in an increase of absolute industry emissions even if specific emissions are declining. This is in contrast with absolute targets imposed on the European Union and its member states by the Kyoto Protocol and the EU burden-sharing agreement (OECD 1998).

However, after the Kyoto Protocol had been passed, industry interest in using the flexible Kyoto mechanisms has increased although there are still only few concrete proposals on how to combine the Kyoto mechanisms and voluntary commitments. A corresponding approach has been developed and realised in the United Kingdom (ETG 2000).

The approach of the UK Emissions Trading Group (ETG) offers a practical answer on how to combine an emissions trading scheme with voluntary agreements. It distinguishes between a so-called “absolute“ sector with absolute emission targets and a “unit“ sector with agreements on specific targets. In the absolute sector, firms can participate in the emissions trading scheme voluntarily via the “direct route“ by accepting an absolute cap on their carbon dioxide emissions, getting financial support in return. In the unit sector, firms that have joined the CCL Agreement take part via the “agreement route“. In the absolute sector a cap-and-trade scheme is established while in the unit

[1] Council Directive 96/389/EC concerning integrated pollution prevention and control.

sector emissions trading is of the baseline-and-credit type. The former imposes an absolute cap on a single firm's emissions that can be freely traded among the participants of the scheme while the latter defines a baseline for the specific emissions of firms that have to buy allowances only if their specific emissions overshoot the baseline and can sell allowances only if they over-fulfil their obligations. The baseline is defined with respect to the industry's obligation in the voluntary commitment. The main difference between cap-and-trade and baseline-and-credit trading is that in the former participants hold property rights over all allowances whereas in the latter property rights are extended only to the "earned" credits which polluters obtain by over-achieving the emission reduction targets. Furthermore, the unit sector may only participate in national trading whereas companies in the "absolute" sector may participate in international emissions trading as well. To prevent allowances from the unit sector swamping the absolute sector, the scheme attempts to limit sales from the former to the latter via a "gateway". This means that trade between the "absolute" and the "unit" sector is unrestricted as long as there is no net flow from the "unit" to the "absolute" sector. In the reverse case, the gateway will be closed.

*Emissions trading and eco/energy taxes*

Both, energy and carbon-dioxide taxes and emissions trading schemes provide incentives to reduce $CO_2$ emissions. They differ in that price controls fix the marginal costs of compliance and lead to an uncertain level of total emissions whereas quantity controls fix the level of compliance but result in uncertain marginal costs. With respect to European and national emission reduction targets, emissions trading seems to be superior to eco-taxes. In many European countries climate-change related energy taxes or $CO_2$ taxes already exist, however, and will not be abolished in favour of trading schemes. In consequence, an additional burden would be placed on companies that are already subject to environmental taxation if they had to join an emissions trading scheme. To avoid this, a tax reduction could be given to firms that join the emissions trading scheme. This is the case in the United Kingdom where companies signing the climate change agreement and participating in emissions trading obtain an 80 percent reduction of the climate change levy. In Germany, energy-intensive industries already enjoyed an 80 percent reduction of the eco-tax until the end of 2002. From 2003 on this eco-tax reduction is only 40 percent. Therefore, an incentive for voluntary participation in the emissions trading scheme could be given by levying an eco-tax of only 20 percent on trading firms and of 60 percent on all others.

## Concluding remarks

The superiority of the British emissions trading scheme lies in the clear interaction of already existing regulatory instruments, voluntary agreements and carbon taxes, on the one hand, and the new emissions trading scheme with the coexistence of absolute and relative reduction targets on the other. No wonder that the Dutch $CO_2$ Trading Group proposed the introduction of a similar scheme in the Netherlands. In this proposal a distinction is made between an "exposed" sector of energy-intensive industries faced with international competition and a "sheltered" sector embracing all other industries and private households. The exposed sector, which is subject to a voluntary commitment to the government, underlies relative reduction targets deduced from energy-efficiency standards that are part of the voluntary commitment, whereas an absolute emission target is imposed on the sheltered sector. In the exposed sector the initial allocation of emission allowances is free of charge; in the sheltered sector they are auctioned annually with the revenues of the auction being channelled back to the participating firms and households by means of a reduction of labour and income taxes and social security premiums. As in the British scheme, trading between the exposed and sheltered sectors is possible. To prevent an unanticipated increase in emissions from both the exposed and the sheltered sectors, the government should be able to adjust the amount of allowances quickly or to buy the excess supply of allowances from the market (Kink et al. 2002).

**The British emissions trading scheme provides for interaction with existing regulations and voluntary agreements**

If the Commission's proposal for a guideline on emissions trading were modified to a hybrid system akin to the British and the Dutch models, it would be easier to integrate already existing national schemes into the European trading system. For the first trading period 2005–2007 preceding the Kyoto commitment period, participation in emissions trading should be on a voluntary basis and the initial allocation of allowances should be free. In the absolute sector, these allowances should be allocated according to the requirements of the IPPC guideline. In the unit sector, allowances for single firms should be allocated on the basis of output-related performance standards defined by voluntary commitments between the corresponding industry and the government. Trade in the absolute sector should be of the cap-and-trade type, and in the unit sector of the baseline-and-credit type. Allowance trading between both sectors should be possible but controlled by a gateway. As an incentive to join the emissions trading scheme, participating firms should obtain an energy/carbon dioxide tax reduction. In addition to allowances resulting from emission reductions within the European Union, Certified Emission Reductions (CERs) obtained from CDM projects and Emission Reduction Units (ERUs) resulting from Joint Implementation should be introduced in emissions trading within the EU from the beginning in order to benefit from cost efficient energy saving projects abroad.

**The EU will allow opt-outs and opt-ins as well as pooling**

The political agreement reached by the Council in December 2002 modifies some issues of the initial proposal. Although trading will start in 2005, individual installations or economic activities can be exempted from emissions trading in the initial period 2005–2007 ("opt-out"). Opt-outs are subject to approval by the Commission, on strict conditions. These notably include fulfilling the same emission reduction requirements as companies and installations participating in the scheme, which can be realised by instruments like voluntary commitments. In addition, member states can unilaterally include additional sectors and gases from 2008 on ("opt-in"). The agreement also provides for the possibility of companies pooling their emission allocations until 2012 ("pooling"), with a pool manager acting as representative on the market for emission allowances. Allocations of emission allowances will be free of charge, but Member States can auction off up to 10 percent of allowances from 2008 (EU Commission 2002).

While these modifications deal with special situations in certain member countries, the question of how to combine emissions trading and traditional environmental policy instruments in an optimal way still remains unanswered. We therefore urge the Commission to modify the proposal further by creating a hybrid emissions trading scheme similar to the British or Dutch type.

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