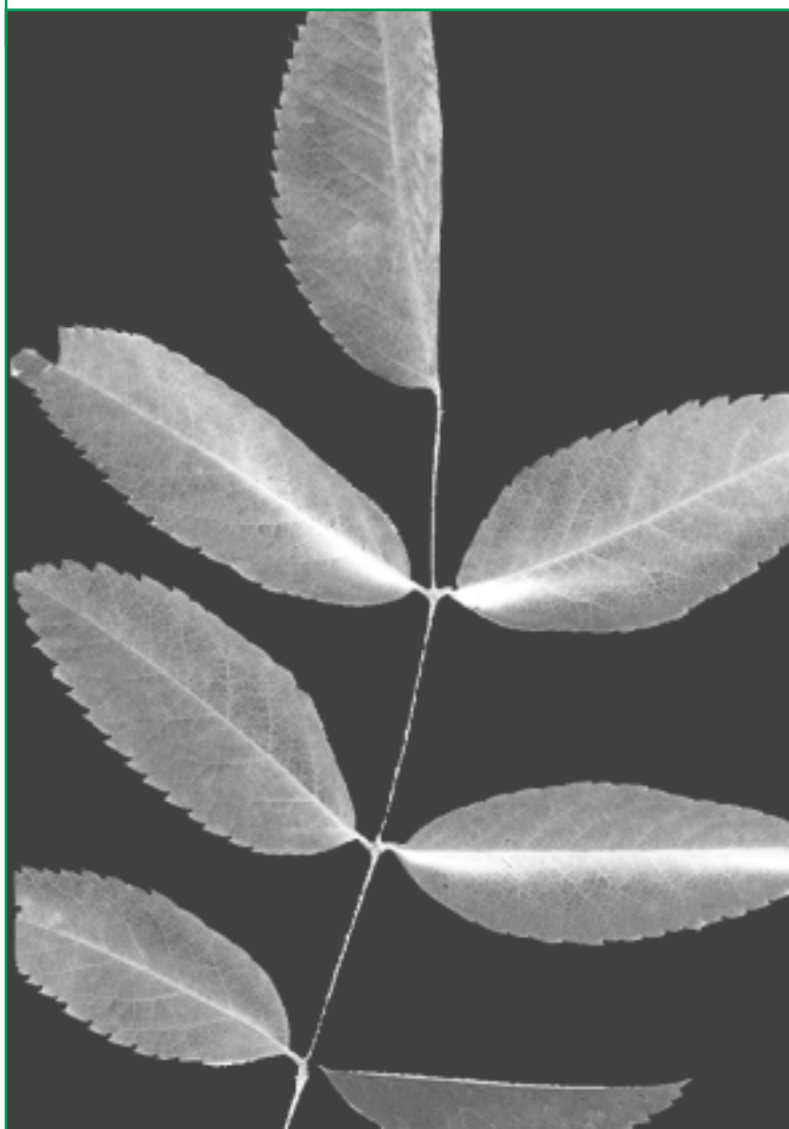


Implementing Domestic Tradable Permits for Environmental Protection



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FOREWORD

In the context of OECD work on the use of tradable permits, an international workshop was organised at OECD on the theme: “Domestic Tradable Permit Systems for Environmental Protection: Issues and Challenges” (24-25 September 1998). The objectives were to get an overview of experiences in OECD countries and to assess the outcome and implementation issues related to domestic tradable permits systems. This book contains the contributions to this workshop.

The Secretariat wishes to acknowledge financial assistance provided by the European Commission (DG XI) and the United States Environmental Protection Agency, the active participation of workshop participants and the contributions of the co-authors. The views expressed are those of the workshop contributors. The book is published under the responsibility of the Secretary-General of the OECD.

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SUMMARY AND CONCLUSIONS*

Background and objectives

There is a growing body of evidence confirming both the environmental effectiveness and the economic efficiency of tradable permits (TPs) in environmental policy. This positive experience (largely from the United States) is generating increased interest in the TP option in several OECD countries, as well as internationally. The most recent example of international interest is the explicit inclusion of the tradable permit approach in the 1997 Kyoto Protocol to the UN. Framework Convention on Climate Change. At the national level, some interesting opportunities to expand the use of TPs are emerging, especially in Europe, where this option has so far been largely ignored.

To contribute to the exploration of these opportunities, the OECD organised a Workshop on the “Domestic Use of Tradable Permits: Issues and Challenges” (Paris, September 1998). The Workshop agenda was structured in such a way as to provide background information on: *i*) experience with the actual use of TPs in various areas of environmental policy; *ii*) the main obstacles currently being encountered in expanding this use to new applications; and *iii*) special issues related to the compatibility of TPs with existing environmental policies, as well as to competitiveness problems.

The Kyoto Protocol was a natural reference for much of the Workshop discussions, but the main focus of the meeting was on domestic applications rather than international ones. The papers presented in Part I of this book therefore provide an overview of the use of TPs in these domestic settings, with particular focus on applications in the fields of air pollution (D. Harrison), land management (V. Renard), water management (A. Kraemer and K. Banholzer), and fisheries management (P. Wallis). Part II examines some of the specific national settings in which tradable permit solutions have been proposed, and offers suggestions as to why these proposals have not yet been successfully implemented. The specific cases of Norway (P. Schreiner), Poland (T. Zylicz), Switzerland (C. Jeanrenaud), and the United Kingdom (J. Steedman) are discussed. Part III focuses on the two challenging issues of compatibility (S. Smith) and competitiveness (N. Johnstone).

Inventory of applications

The US was the first country to apply TPs widely in the context of its environmental protection programmes, and even today, most applications of the TP tool can be found in that country. These applications have mainly related to the emission of air pollutants, but both water-based and land-based applications have also been involved at times. There have also been applications in other OECD countries (*e.g.* Germany, Australia), but usage of the tool has been less in these countries than in the US. The annex to this chapter provides an overview of the various systems currently in place which were discussed during the Workshop.

Many of the TP applications discussed during the Workshop are “traditional”, in the sense that they have involved commercial trades of assets, in return for the right to emit into, to develop, or to use parts

* This chapter was prepared by the Workshop Rapporteur, Tomasz Zylicz (Warsaw Ecological Economics Center, Warsaw, Poland) and the OECD Secretariat, with additional contribution from Olivier Godard (CNRS, Paris, France).

of the natural environment during production or consumption processes. Other situations have merely “resembled” permit trading markets, in the sense that they involve trades which are not necessarily commercial, but which are “equivalent to” TP systems. The latter have been included because they reflect the widening application of the TP concept as a whole, and because they could eventually evolve into more commercial TP arrangements. For example, TP-type arrangements have been used in both the US and France to regulate the development of land, in accordance with an environmental objective.

It is clear from this inventory that TPs can increase economic efficiency (*i.e.* minimise compliance costs). Useful lessons can also be learned from studying how TPs can be designed and implemented in order to maximise their environmental benefits, while still satisfying political viability, administrative efficacy, and compatibility criteria.

Lessons learned from actual experience with TP regimes

One important lesson is that TPs remain somewhat controversial, even in the US, with its relatively long history with the tool. It also turns out that, in the case of TP systems aimed at *air pollution* abatement objectives, political support for these programmes is highest when they are introduced as completely new initiatives, rather than as “add-ons” to an already-existing policy. Therefore, it may *not* be appropriate to opt for an “incremental” approach to the implementation of TP schemes for air pollution (*i.e.* starting with simple “bubbles”, then gradually moving to more complex schemes, and eventually generating a full-fledged TP system). It may be preferable to introduce the TP system *en bloc*, following negotiations with all key parties involved.

On the other hand, experience (albeit limited) with trading in the *water* sector suggests that TP systems may be more politically-acceptable when they are based on pre-existing and long-standing regulatory or institutional arrangements. In these situations, a gradualist approach to implementation may be appropriate.

The Workshop discussions also revealed few fundamental ethical objections to TPs. There is apparently a strong recognition (at least among these particular experts) of the fact that it is no more immoral “to sell the rights to pollute” via a TP system than it is to apply alternative policy instruments. In the case of environmental taxes, the end result would be virtually the same – the polluter pays for the “right to pollute”. Even in the case of environmental regulations, the polluter would actually receive the right to pollute for free hardly a better solution than the tradable permits option would generate. The main point is that economic actors make decisions every day which have the effect of either “selling” or “buying” access to the environment – tradable permits only help to make the choices underlying these decisions more explicit.

Two broad types of TP system are in operation: those based on emission reduction credits (ERCs), and those based on *ex ante* allocations (“cap-and-trade”). The former approach takes a “business as usual” scenario as the starting point, and compares this baseline with actual performance. If an emitter/user performs better than the anticipated baseline, a “credit” is earned. This credit can then either be used by the emitter/user himself, either at the current location or elsewhere, or sold to some other emitter whose emissions are higher than the accepted baseline (and presumably at a lower price than what it would cost the latter to abate on his own). The “cap and trade” approach sets an overall emission/use limit (*i.e.* the “cap”) and requires all emitters to acquire a share in this total before they can emit. Emitters may be allocated their shares free-of-charge by a relevant environmental authority, or the shares may be auctioned. No matter how the initial allocation of shares is arrived at, their owners can then either utilise them as emission permits in current production, save them for future use (if “banking” is allowed), or trade them to other emitters.

US experience suggests that “cap and trade” systems are easier to negotiate with stakeholders than credit-based systems, and are therefore more likely to be politically-acceptable. Credit-based trading requires the computation of arbitrary and often controversial baselines in order to establish a reference point for future reductions. The certification of credits has also proven to be administratively-cumbersome in some cases. Another problem with crediting occurs when emitters seek to expand the

scale of their activities. Even if these activities become less pollution-intensive on a per unit basis, total “baseline” emissions may grow to unacceptable levels from an environmental perspective, due to the increased scale of economic activity that is implied.

Independently of the choice between either of these two approaches, determining the initial allocation of permits to individual emitters/users is a critical task. Despite a theoretical preference for an auction approach, “grandfathering” of the initial permits has been applied in virtually all applications that have been observed in practice. Only in the US SO₂ allowance programme is a portion of each “vintage” of the permits offered at auction. This is done in an attempt to make up for market imperfections and/or to accommodate newcomers to the market. On the other hand, the “grandfathering” principle has occasionally been substantially modified, by taking such variables as past environmental performance, technical standards, and equity considerations into account.

However, Workshop participants also pointed out that auctions can still play an important role, even if only a small part of the permit allocation is channelled through this particular mechanism. This is because auctions help to reveal important information about the marginal costs of abatement – information that is not necessarily even known to the individual emitters themselves. To the extent that the focus of company engineers is placed on technical solutions (rather than economic ones), these officers may not know what the actual marginal abatement costs facing the firm are. More importantly, they may not be aware of when (and by how much) these costs are lower than the costs of continuing to emit. By being forced to participate in permit auctions, this information gap may be at least partially filled, perhaps leading to “win-win” situations (*i.e.* where both the environment and the company’s profits benefit).

There are natural reasons for *water* use (or water quality) to be less-often controlled using TPs than air quality is. First, water is relatively heavy, which raises the transaction costs of its use in other locations. Second, water can often only be economically-traded in a downstream direction, which tends to reduce the potential size of the permit market. This particular constraint does not apply to waterborne pollution, but here as well, trading must be confined to those emitters/users who influence a given body of water – which again limits the size of the potential market. Water pollution/use markets will be even further constrained if it is not only the pollution *load* which matters, but if the distribution of pollutants over *space* or over *time* also matter.

It is worth noting here that the Urban Wastewater Treatment Directive of the European Union allows for regional averaging of pollution loads. This leaves room for inter-city co-operation, which could in turn pave the way for TPs to be used. No regions, however, have taken advantage of this opportunity yet.

There have not been many applications of TPs in *land use management*. Most existing experience (again, mainly in the US) is in the area of “Tradable Development Rights” (TDRs). A key feature of TDR systems is that trades can occur only once: after a development right has been transferred and used, it cannot be used another time (it cannot even be returned to the original seller). Several legal issues arise from difficulties involved in trying to separate land rights into their various components. In some jurisdictions, it is not possible to separate ownership of the land from ownership of the right to develop that land. In others, however, this separation is legally-recognised, so various titles attached to the same lot *can* be independently-traded.

A few issues are sometimes considered to be specific to the implementation of TP-based management regimes. For example, TP systems are often perceived to require very costly monitoring programmes, in order to be certain that emitters/users do not exceed their permitted amounts. Indeed, the US SO₂ allowance programme actually requires emitters to install continuous emission monitoring systems (CEMS). On the other hand, it can be argued that, so long as the objective of the policy is really to reduce emissions/use, similar monitoring requirements (and costs) apply to *any* potential policy instrument, such as taxes or regulations.

It is also sometimes argued that TPs present a risk of strategic manipulation by market participants who are seeking to exert monopolistic power. This problem has been extensively-researched in the theoretical literature, but in none of the actual applications has it ever emerged as a serious practical problem. Apparently, precautions adopted in existing systems related to constraints on banking opportunities, auctions for newcomers, etc., have proven sufficient to avoid this problem.

There is also a concern about how to accommodate “newcomers” into a TP system in which the initial permits were originally “grandfathered”. In the TP regimes reviewed during this study, this problem was resolved either by requiring newcomers to buy permits from other emitters, or by leaving a small initial allocation of permits in public hands, to be eventually channelled through an auction to the newcomers when the need arose. In none of these cases was there ever a rule adopted that would have the effect of guaranteeing newcomers access to sufficient supply of permits with which to launch their business. “Environmental carrying capacity” is apparently at least as scarce as land itself, and there is no tradition of guaranteeing newcomers that the public sector will provide land for the commencement of a new business – in all cases, the land must be purchased from existing owners. All practical experiences with TP systems therefore points to the conclusion that there is no need to give special access privileges to newcomers.

A prerequisite for TPs to be successfully implemented is a credible system for their enforcement. But enforcement implies some form of direct control over firm behaviour – a result that most businessmen would generally wish to avoid, in the interest of economic efficiency. As indicated by business representatives, industries therefore tend to prefer long-term voluntary agreements over TPs. Business will usually support a TP approach only when the “voluntary” alternative is not likely to be acceptable. On the other hand, TPs provide lower costs for business relative to a taxation approach (if the permits are grandfathered). Given this particular choice, business has therefore tended to prefer the TP option. Business will also usually prefer a TP approach involving grandfathering of permits over one that involves auctions, because of the increased asset values that the former implies.

TDRs provide a good example of some of the equity issues involved with implementing TP systems. The very reason for using TDRs in the first place is to allow land owners who cannot fully exercise their property rights *in situ* to exercise these elsewhere (*i.e.* where development constraints are less stringent). Other things being equal, the imposition of stricter development constraints on a particular site should lead to capital loss for the owner of that site. The existence of a TDR system effectively compensates the land owner for that capital loss. Furthermore, this compensation is provided in such a way that the public budget does not have to bear the financial cost.

On the one hand, the beneficiaries of TDR-based compensation may be wealthy absentee landlords, some of which may lobby hard for their land to be included in “non-development” areas in the first place, so that they may receive the right to trade on the (present) conservation status of this land. This has apparently been the experience with some of the TDR systems which have been used in the French Alps. Obviously, decisions about how and where environmental constraints are imposed should be based as far as possible on environmental considerations, rather than on economic ones. On the other hand, social equity objectives may be promoted by TDRs, if the main beneficiaries are the rural poor. This is the situation which may exist in under-developed municipalities in biodiversity-rich regions of Central and Eastern Europe (at least before the property is transferred – at low prices – to wealthy newcomers).

As noted earlier, various types of grandfathering have been used in virtually all existing TP applications, despite the theoretical superiority of an auctions-based approach. Usually, grandfathering is done in order to satisfy some perceived equity consideration. As long as the number of firms which are granted TPs free-of-charge is large, and as long as TP markets eliminate rents from the newly-created assets, grandfathering seems to be socially-acceptable. However, there was at least one case where the boundaries of social acceptability were apparently surpassed. This involved ozone depleting substance (ODS) quotas allocated in the US under the Montreal Protocol. In this case, CFC producers and importers were expected to receive large windfall gains as a result of the introduction of the TP quota creating a scarcity rent. These windfall profits were then taxed away by the US Congress.

Apart from this particular example, however, there seem to be some reluctance to, and at least lack of experimentation in, using TPs and taxes in combination with each other (see also the later discussion of this point).

The US SO₂ allowance market illustrates the volatility that can be associated with permit prices under certain conditions. In this case, prices had been gradually declining for many months before they hit a value corresponding to only 1/20 of the level that had been anticipated during policy discussions in

the early 1990s. Even though this decline tempted some to consider the trading programme a failure (and to intervene), no “fine tuning” was eventually undertaken. Prices then started to increase, and no corrections seem necessary now.

TPs are often required to make up for whatever deficiencies are generated by more conventional policy instruments. For example, early US experiments with emissions trading were burdened with constraints aimed at correcting defects that had emerged in applying “command-and-control” approaches. These constraints meant that the TP regimes did not function as effectively and efficiently as they might have. Perversely, this also had the effect of increasing overall opposition to the TP idea.

The volume of trading in “cap-and-trade” (allowance) systems tends to be larger than it is for emission reduction credit systems. This is at least partially because the commodity being traded is usually better-defined under allowance-based regimes. Under the ERC approach, especially when the baselines which determine the ERCs are unclear, trades are often controversial, often ending up with permanent administrative oversight and/or cumbersome certification procedures. This uncertainty also complicates the initial allocation of rights, which can also involve controversial procedures and assessments.

Most TP programmes in the US related to air pollution have used *banking*, in order to provide additional flexibility in the intertemporal allocation of abatement efforts. Banking works especially well when a significant phase-down in emissions is envisaged as part of the TP programme. By allowing early abatement efforts to be banked for later use, the environment benefits can be increased, in the sense that largest emission reductions are achieved when emissions are the highest (*i.e.* when the marginal damages caused by these emissions are the highest). There are also economic benefits associated with banking – because it reduces the exposure of emitters to large shifts in permit prices.

CEMS offer some degree of certainty in the emission monitoring process, and are therefore often mandated by TP programmes. However, this (costly) requirement often precludes the participation of small emitters, whose combined emissions can sometimes be more significant than those of the larger emitters. Experience with the US RECLAIM Programme proves it is possible to combine both large and small sources, with different monitoring regimes being applied to both groups. In that case, small sources were given an opportunity to participate in the programme, with emissions being *estimated* (*e.g.* by fuel meters and/or using emission technology factors). This demonstrates that, under certain circumstances, it is possible to delink the TP regime itself from the question of monitoring. However, it should be noted here that this approach has only been tried in the US, where emitters tend to be covered by a complex and well-established system of supplementary environmental controls, in addition to the TP system itself. The same TP programme replicated in other jurisdictions could therefore face compliance and enforcement problems.

There is a trade-off between the increased efficiencies generated by expanding the size of the TP market, and the risk of creating local environmental “hot spots”, as a result of the trading in TPs. In some programmes, this problem has been solved by constraining the size of the market, so that only nearby emitters can trade with each other. Another approach is to use two “tiers” of environmental regulation. For example, in the US SO₂ programme, there are no geographical constraints on the market (*i.e.* trading can occur between *any* two participating sources who wish to do so). However, these sources are also subjected to additional local environmental constraints, with the result that holding an appropriate number of SO₂ allowances is a *necessary*, but not a *sufficient*, condition for emitting. Any source (in fact, any economic agent) can legally purchase any number of allowances, but emitters are also subject to specific regulations aimed at protecting local ambient air quality.

The crucial role of local regulations is also evident in TP-based land-use management programmes. The success of these programmes ultimately depends on the stability of local zoning regulations. If the spatial planning status of a particular site can be easily changed, the credibility of the TP system will be undermined, TDR prices will become more volatile, and equity considerations may then make it more difficult to maintain political support for the TP regime. This again underscores the fact that successful implementation of a TP scheme usually depends on the adequacy (although not necessarily the stability) of the wider institutional framework in which that system must operate.

Lessons learned from *proposals* for new TP systems

Other important lessons were also obtained during the Workshop by studying the reasons why TP systems were not implemented (or did not work very well where they were implemented) in selected OECD countries [Norway (P. Schreiner); Poland (T. Zylicz); Switzerland (C. Jeanrenaud); and the UK (J. Steedman)]. Because these countries are all located in Europe, much of what follows is discussed in terms of a US-Europe comparison.

First, there is the fact that TP systems apply market forces in order to stimulate a cost-effective allocation of abatement or conservation efforts. While many US stakeholders appreciate the benefits of a market-based approach, it is less clear that European stakeholders are as enthusiastic about market-based solutions to environmental problems. On the other hand, where market instruments *are* used, taxes will often be chosen over TPs in Europe, but the reverse will be true in the US. The Workshop discussions confirmed these general preferences, even though several European participants acknowledged that the exemptions often associated with “eco-taxes” can sometimes seriously undermine the environmental performance of these taxes.

Moreover, any new TP programme must take into account the existing regulatory framework. The European Union has already developed an extensive body of regulations, based on the principle of Best Available Technology (BAT). This approach was seen by some participants as being incompatible with TPs, because it focuses on equalising the *technologies* being used at a common (high) level. TPs, on the other hand, focus on equalising the *costs* of the abatement effort across emitters, leaving the choice of technologies to the emitters themselves. Another situation in which TPs are perceived to work against existing environmental policies is where TPs are expected to replace an existing tax. In cases involving grandfathered TP regimes, policy-makers may worry that public sector revenues will be adversely affected by the adoption of the TP system.

One of the main advantages of TPs is that they achieve cost-effectiveness in the allocation of abatement effort, and let emitters either save on abatement costs (TP buyers), or profit from over-compliance (TP sellers). However, what is perceived to be a virtue in one political culture may have little or no appeal in another. Quite often, therefore, “there is no constituency for efficiency”. As long as policy-makers can demonstrate their concern for the environment by doing what is considered to be satisfactory by key interest groups, rather than by improving the environment in a more cost-efficient way, there will be little demand for TPs. Likewise, there will be no pressure from major industrial polluters to move toward TP regimes, as long as these firms can build their “green image” without having to seriously consider abatement cost questions.

The lessons learned from difficulties in introducing TPs in several European countries can also be phrased in normative terms (*i.e.* as conditions necessary for the adoption of such a policy innovation). The following steps were identified as general prerequisites for accepting TPs in countries which have little or no experience in applying them (summary provided after the Workshop by O. Godard).

- *Key stakeholders should perceive that improved economic efficiency is likely to be an important part of the solution to the environmental problem at hand.* Not all stakeholders are as convinced of the need for increased economic efficiency as economists are.
- *Those who run the existing regulatory regime should optimally have to bear some of the costs of not resolving the environmental problem at hand.* If those who run the current system are not the same people who suffer the economic costs of its inefficient design and operation, there will not be as strong a constituency for change as might otherwise exist.
- *Key stakeholders should be convinced that it is hard to solve these problems by simply amending or enhancing the old regime.* If most stakeholders think that the environmental problem can be resolved by a slight amendment to the existing “command-and-control” system, the institutional climate for TPs will not be very accommodating. For example, in the several of the European cases reviewed by the Workshop, it was suggested that voluntary approaches might be a more appropriate way of

providing the type of regulatory flexibility that business needs. Such an approach may satisfy stakeholders' immediate concerns, but it does not provide permanent (*i.e.* system) solutions to the problem. It may also nurture coalitions which make future policy reforms even more difficult.

- *Key stakeholders should be convinced that TPs can actually provide a solution.* Stakeholders need to agree that TPs are a practical way of addressing the environmental problem at hand, and that they can be effectively implemented.
- *Key stakeholders should be able to form a coalition in order to advocate a policy change.* Such a coalition might involve utility managers, NGOs, political parties, business leaders, etc. Without such a coalition, the chances of successful adoption of a TP regime are reduced.
- *The emergence of a strong opposition coalition should be prevented.* Vocal minorities can sometimes block policies that would otherwise benefit the “silent majority”.
- *TPs should be perceived as being consistent with the social philosophy prevailing in a given region.* TPs have traditionally been opposed on the basis that “buying the right to pollute” is not socially-appropriate. This view is now being nuanced in many jurisdictions, but the adoption of TPs can proceed only as quickly as this nuancing proceeds.

Special issues

Compatibility of TPs with other policy instruments

In today's complex regulatory environments, questions about the compatibility of different policy instruments are among the most fundamental ones (S. Smith). In the early US experiments with TP systems (*e.g.* Fox River emissions trading), TPs were introduced in a way that was fully compatible with the then-existing “command-and-control” regulations. The TP arrangements were largely based on existing legal concepts, and they were overseen by the same administrative bodies. Hence, a potential incompatibility problem was totally avoided, although at the expense of retaining some of the negative features of the old system in the process.

One of the success stories in TP applications – the US lead trading programme – was successful in large part because this compatibility problem did not exist. The lead trading scheme and the pre-existing regulations were simply directed at different goals, and did not conflict with each other. Trading went extremely well, and no special authority was needed to approve the transactions (other than someone to reconcile the accounts of sellers and buyers).

In most other TP programmes, compatibility questions *do* arise, since a mix of instruments is virtually always applied to address the same problem. On occasion, however, the mixing of instruments can be:

- wasteful (if using more than one instruments leads to additional administrative costs to be borne by either the government or the regulated emitters);
- inefficient (if keeping an inflexible instrument in place prevents the efficient reallocation of abatement effort); and
- unnecessary (if only one instrument actually “bites”).

There are sometimes good arguments for combining TPs with taxes. At least two types of theoretical situation exist in which one might expect combinations of taxes and TPs to be both effective and efficient, both of which were discussed during the Workshop. These are:

- Situations involving the management of multiple (and synergistic) pollutants, where taxes may be necessary to supplement permits (or *vice versa*) if the firm's pollution behaviour (for one reason or another) is more sensitive to one instrument than it is to the other.
- Situations where the level of uncertainty about total abatement costs is high, and where there is considerable intolerance associated with not meeting at least a minimum environmental objective. In these cases, a TP approach might operate over the range of emissions deemed to be especially important from an environmental perspective (thereby assuring the achievement of that

objective), but a taxation approach might operate over emission ranges lower than this level (thereby increasing the predictability of total abatement costs). This could be termed the “safety-valve” approach.

There are also sometimes good reasons for combining direct regulations with TPs, in order to let the trading occur smoothly without the risk of creating local “hot spots” in the environment. As a rule, however, “packages” of instruments should be applied only when there is a convincing reason to do so, and these packages should be designed with considerable care. Otherwise, the effectiveness or efficiency of the policy response may be compromised. Moreover, favouring “traditional” policy instruments in the policy mix may, in the long run, slow down the policy innovation process, and constrain the use of more effective options.

Competitiveness issues

In aggregate, there is little doubt that TP systems can increase the efficiency of environmental policies, relative to the regulatory schemes which they typically displace. However, individual firms will gain more than others, and some firms will actually be made worse-off by the TP system. These effects will differ, depending on how the TP system is designed and implemented. The different economic effects of the TP regime on individual forms will, in turn, open up the possibility of either gains or losses in economic competitiveness (N. Johnstone).

In the short-term, the main determinants of the competitiveness effects of TP regimes will probably be related to:

- the grandfathering/auctioning decision for the initial permit allocation;
- the way in which any auction or tax revenues are “recycled” into the public finances; and
- the extent to which firms are able to use their particular position in the TP market to concentrate their economic power.

Over the long-term, the “dynamic efficiency” effects of TPs on innovation (and thus, on economic competitiveness) may be the most important benefits associated with those systems. Care should therefore be taken to ensure that the design and implementation of TP regimes take account of their impacts on factor productivities, on investment rates into the firm, on innovations in abatement technologies, and on any relocation options that may be available to the firm.

Outlook for the future

Given the positive results achieved so far in various TP programmes in the US, TPs seem likely to continue to be applied there (and perhaps at an increasing rate). However, the political climate for TPs in Europe is not as certain. On the one hand, it seems likely that European environmental policies will increasingly embody a philosophy that takes an ambient quality target as the starting point, and one which uses an emission cap as part of the strategy for meeting that target. This would pave the way for a wider application of TPs. On the other hand, several European governments are currently contemplating long-term programmes involving pollution taxes (and with the tax rates gradually being increased over time). Such systems seem unlikely to be very supportive of TP regimes.

For its part, European industry seems to prefer abatement commitments that are expressed on a per output basis (*e.g.* per kWh or per tonne of commercial output) – in effect, technology-based rules. Where technology-based rules cannot be applied, the preference seems to be for voluntary arrangements, rather than for economic instruments, such as TPs. US industry, on the other hand, seems to have generally accepted that the increased flexibility associated with economic instruments is preferable to the administrative constraints imposed by a regulatory approach, and that a TP regime offers lower total costs than a tax-based one does.

Environmental NGOs in the US have also come to understand that TPs sometimes offer the possibility of higher levels of pollution abatement than would otherwise have been politically possible, without imposing excessive costs to emitters. They have therefore been increasingly willing to consider the TP option, under certain conditions. In Europe, however, NGO suspicion of the TP tool remains relatively high, mainly because of concerns about their environmental effectiveness.

Nevertheless, both actual experience with the TP tool, and the willingness to experiment with it, *do* seem to be expanding. Perhaps most importantly, these practical experiences at the national level may also help to ensure wider implementation at the international level – an important consideration in view of the fact that trading forms a key part of the recently-signed Kyoto Protocol to the Framework Convention on Climate Change. The implementation of an international TP system will probably be made easier if domestic systems have already familiarised local stakeholders with the TP approach. Domestic trading can also help by facilitating the early identification of marginal abatement costs, and thereby facilitating trades between private transactors at the international scale. Similarly, domestic TP systems would facilitate the eventual establishment of monitoring and control systems at the international level.

In short, considerable potential seems to exist for wider use of the TP approach, both nationally and internationally. The key challenge now is to realise that potential.

Annex 1. Summary of TP applications discussed during the OECD workshop

Country/area/agents	Programme	Period	Commodity	Initial allocation	Banking	Effects	Savings
Air protection							
US/AQCR ¹	EPA emission trading	1975-	ERC ²	Actual emissions	Yes	Mixed	1-12 B\$ ³
US/refineries	EPA lead-in-gasoline	1983-1987	Lead additives	Grandfathered; (shrinking)	Yes ⁴	Elimination of lead	226 M\$
USA/firms	EPA ODS ⁵	1986-1998	ODS allowances	Grandfathered/1986 market share; (shrinking)	No	Compliance with Montreal Protocol	49%?
US	SO ₂ allowance	1990-	SO ₂ allowances	Grandfathered/1985-7 production; (shrinking and auction)	Yes	Early reductions	225-375 M\$ ⁶ (25%-34%)
US/Los Angeles area	RECLAIM ⁷	1994-	NO _x and SO ₂ allowances	Grandfathered/modified; (shrinking)	No	8.3% and 6.8% annual reductions planned for NO _x and SO ₂	Over 40 M\$ ⁸
US/engine producers	ABT ⁹		FEL ¹⁰ for HC+NO _x	Production volume	Yes	75% reduction planned for HC	?
US/Northeast/stationary	OTC ¹¹ NO _x budget	1994-2003	NO _x allowances	Grandfathered; (shrinking)	Yes	75% reduction planned	80 M\$/y (30%)
Switzerland/Basle Canton	VOC control	1993-	VOCs	Emission limit (minus 20%)	Yes ¹⁸	Very few trades	?
Poland/Chorzów	Demonstration project	1991-1992	ERC	Grandfathered	No	Speeding up abatement plans	?
Water resource management							
US and Australia	Tradable abstraction rights	19th century-	M ³ /y	Grandfathered	No	Stability of use	?
Germany/Hamburg	Groundwater abstraction fee	1989-	Retiring water rights			Transfer of water rights	?
Water protection							
US/Wisconsin	Fox River	1980-	BOD	Allocated by emitters	No	Ambient quality	?
US/Colorado (point-non-point)	Dillon Reservoir	1984-	Phosphorus	Grandfathered	No	Reduced eutrophication	1 370 \$/kg P (hypothetical)
US/North Carolina (point-non-point)	Tar-Pamlico River	1989-	Phosphorus/nitrogen	Grandfathered	No	Nutrient discharges reduced by 28%	?
Australia/all States	Inter-state salinity trading	1992-	"Salt-credits" ¹²	Grandfathered	Yes	Reduced salinity	?
Australia/New South Wales	Hunter River salinity trading	1992-	Salt allocations for coal mines	Grandfathered//modified	Yes	Reduced salinity	?
Land-use management							
US/Pinelands	Cohansey aquifer and forest conservation	1980-	TDR ¹³	Zone assessment process	Yes	Slowing down land conversion	?
US/Montgomery County	Agricultural land preservation	1980-	TDR	Setting housing density limits	Yes	Slowing down land conversion	?

Annex 1. Summary of TP applications discussed during the OECD workshop (cont.)

Country/area/agents	Programme	Period	Commodity	Initial allocation	Banking	Effects	Savings
US/Lake Tahoe	Watershed preservation and recreation	1987-	TDR	CC ¹⁴	Yes	Reduced development-related stress	?
France/Alps	Landscape preservation	After 1977	TDR	Urban housing density limits	Yes	Reduced urbanisation	?
New Zealand/Cities	Revised district scheme	1987-	TDR	Density allowances	Yes	Protection of historic sites	?
Fisheries							
Australia	4 species	..	ITC ¹⁵	Grandfathered	No	Conservation, efficiency, rents	?
Canada	14 species	..	ITC	Grandfathered	No	Conservation, efficiency, rents	?
Iceland	16 species	..	ITC	Grandfathered	Yes ¹⁶	Conservation, efficiency, rents	?
Netherlands	4 species	..	ITC	Grandfathered	No	Conservation, efficiency, rents	?
New Zealand	33 species	..	ITC	Grandfathered (with buyback to achieve sustainable harvest)	No ¹⁷	Conservation, efficiency, rents	?
US	4 species	..	ITC	Grandfathered	No	Conservation, efficiency, rents	?

1. AQCR: Air Quality Control Region.
2. ERC: Emission Reduction Credit.
3. Until 1986.
4. After 1985.
5. ODS: Ozone-depleting Substances.
6. In 1995.
7. RECLAIM: Regional Clean Air Incentives Market.
8. Based on the dollar value of transactions.
9. ABT: Averaging, Banking and Trading.
10. FEL: Family Emission Limit (emission cap for each engine model produced by a manufacturer).
11. OTC: Ozone Transport Commission.
12. Salt credits can be obtained either by abating salt loads, or by enhancing river flows in the Murray-Darling basin.
13. TDR: Transferable Development Right.
14. CC: Coverage Coefficient – the portion of the surface area of a lot that can be built-up.
15. ITC: Individual Transferable Quota.
16. 20% of under-harvested quota can be used in the following year.
17. After 1996.
18. Permits lose 20% of their value *per annum*.

Part I

EXPERIENCES WITH EXISTING SYSTEMS

Chapter II

TRADABLE PERMITS FOR AIR POLLUTION CONTROL: THE US EXPERIENCE

by

David Harrison, Jr.

NERA Associates, Cambridge, Massachusetts, USA

Introduction

Interest in the use of tradable permits to deal with air quality concerns has increased dramatically in recent years.¹ Once the province of economic theorists, tradable permits are increasingly included in air quality policy. For example, regulations recently promulgated in the US to reduce emissions of nitrogen oxides (NO_x) in 22 Eastern states and the District of Columbia included a model tradable permits programme to implement the reduction. The most prominent example of increased attention is the extraordinary interest in the development of tradable permits mechanisms to implement the proposed Kyoto agreement on carbon dioxide (CO₂) and other greenhouse gases (GHGs). The Kyoto agreement provides for several tradable permit mechanisms, including an emissions trading programme as well as a joint implementation programme and a so-called Clean Development Mechanism. Many countries and groups of countries – including the US and the European Union – are considering the use of tradable permits for domestic implementation of the Kyoto agreement.

This increased interest in tradable permits is due to several factors. These include the increasing cost of pollution control, the support of some segments of the environmental community, the importance of “new” environmental problems that do not suffer from entrenched regulatory mechanisms, and a political shift to “market-oriented” policies in general.² With regard to the current interest in applying the concept to greenhouse gases, perhaps the most important factor is the impression that previous applications of the approach have been successful in reducing the overall cost of meeting environmental objectives. Indeed, the US government has pointed to the success of the sulfur dioxide (SO₂) allowance programme as evidence of the likely success of marketable permits programmes for GHGs.

Purposes of this paper

This paper provides an overview of the tradable permits programmes that have been developed in the US to deal with air quality and related issues.³ There are three specific objectives:

- *Provide a framework for describing programmes.* Many decisions must be made to develop a specific marketable permits programme. It is useful to organise these decisions in a framework that can be used to compare the choices that are made in the different programmes.
- *Evaluate successes of the programmes.* Existing programmes have had varying success. We provide summaries of the success of these programmes, as judged by trading volume and cost savings achieved.
- *Determine lessons learned.* The results of existing programmes provide a basis for some general lessons gained from the US experience. The paper speculates about these lessons, using the framework developed to describe the programmes.

It should be noted, however, that a full exploration of all these issues would require a more substantial undertaking than was possible in this scoping report.

Framework

The theoretical advantages and characteristics of marketable permits programmes have been well-established for many years. Economists have long pointed out that creating a market for emissions can lower the cost of meeting emission reduction targets.⁴ There are many decisions that must be made, however, to turn theory into practice. This paper organises these decisions into the three broad categories, corresponding roughly to the chronology in which they would be addressed.

- **Threshold Issues.** These include decisions regarding the basic purpose and nature of the programme.
 - *Emission goals to be achieved.* This includes the distinction of whether the trading programme is set up when emission reduction goals are set, or whether the trading programme is set up as an adjunct to existing goals.
 - *Geographic area covered.* This includes the decision of whether trades can be made within local air quality control regions, states, groups or states, or the entire US.
 - *Nature of the commodity to be traded.* This issue involves the important distinction between two types of programmes: i) “cap-and-trade” programmes in which overall emissions are capped and parties trade an allowance (*i.e.* the right to emit a unit of the given pollutant); and ii) “credit-based” systems, in which the commodity is an emission reduction credit (*i.e.* a credit based upon a showing that the seller has reduced emissions below a baseline level).
- **Design Issues.** These include the decisions that arise as the programme is designed and turned into a specific regulatory programme.
 - *Allocation of initial allowances.* This issue is only relevant in cap-and-trade programmes. Some method is required to distribute the initial allowances. Methods include various formulas to distribute initial allowances to participants as well as auctioning of the allowances.
 - *Geographic or temporal flexibility or restrictions.* This includes the possibility of restricting trades among different parts of the overall geographic range of the programme. It also includes the possibility of banking (*i.e.* reducing emissions more than required and “banking” the surplus for future internal use or sale).
 - *Emission sources that are required or allowed to participate.* This includes specification of the universe of sources that must participate in the trading programme. It also includes the possibility of allowing other sources to “opt in” to the programme.
 - *Institutions established to facilitate trading.* This includes the possibility of encouraging third parties (*e.g.* brokers) to participate as well as the possibility of setting up an auction or other institutions to increase liquidity and establish market prices.
- **Implementation Issues.** These include the decisions that come to the fore as the programme is implemented.
 - *Certification of permits.* This decision applies to credit-based programmes, which require that emission reductions be certified before they can be traded.
 - *Monitoring and reporting of emissions.* Methods must be designed to monitor and report emissions from each participating source.
 - *Determining compliance and enforcing the trading programme.* These decisions relate to the means of determining whether sources are in compliance with, and of enforcing the programme if sources are out of compliance.
 - *Maintaining and encouraging participation.* This relates to decisions made to keep sources in the programme and encourage participation of sources whose participation is optional (*e.g.* those given the opportunity to opt-in).

Although these categories are clearly interrelated, they provide a useful way of organising the many specific elements and decisions that must be made to develop a concrete programme.

Organisation of the paper

The paper is organised according to the rough chronology of when the US tradable permits programmes were developed. Section II provides information on the early experience, which includes various programmes developed in the late 1970s and 1980s. Section III provides information on the two major existing programmes, the SO₂ allowance programme for emissions from electric utilities and the Southern California RECLAIM programmes for NO_x and SO₂. Section IV describes two recent programmes. Section V discusses the lessons learned from this experience with tradable permits in the US.

Early experience

Although current comments tend to focus on recent programmes – particularly the SO₂ trading programme initiated in the 1990 Clean Air Act Amendments – several tradable permits programmes were developed in the late 1970s and 1980s. These include the following.

- Emissions Trading (ET) programmes;
- lead-in-gasoline; and
- ozone-depleting chemicals.

The performance of these early programmes has varied widely. Indeed, these early experiences show how decisions on the elements of a tradable permits programme can affect the volume of trades and the cost savings actually experienced.

EPA emissions trading (ET) programmes

Starting in the mid-1970s, the US Environmental Protection Agency (EPA) and the States developed several programmes designed to increase flexibility and reduce the costs of achieving the national ambient air quality standards. The programmes are the following:

- *Netting*. This allows large new sources to be exempt from certain burdensome review procedures if existing emissions elsewhere in the same facility are reduced so that the net addition from the new source is below the level requiring new source review.
- *Offsets*. Before a major new source can locate in an area that does not attain a given national ambient air quality standard, it must offset whatever emissions remain after installing stringent controls by getting an existing source to reduce its emissions by at least the same amount.
- *Bubbles*. The “bubble” policy allows firms to combine standards for several different sources into one overall limit. The name evokes the image of placing a “bubble” over several individual stacks and then measuring the sum of their emissions, rather than each one individually.
- *Banking*. The intention of the banking programme is to allow firms to take actions to reduce emissions below the application standard, and then “bank” the credits for future internal use or sale.

The programmes are related by the common objective of providing sources with additional flexibility to comply with traditional command-and-control standards.

Description of elements in ET programmes

Table 1 summarises the main elements of the ET programmes, with differences among the programmes noted.

Table 1. Summary of Elements of Emission Trading (ET) Programmes

Threshold Issues	
<i>Emission Goals:</i>	• Implement existing “command-and-control” standards. • Offsets were intended to reduce net emissions; others are neutral.
<i>Geographic Area:</i>	• All Air Quality Control Regions (AQCRs) are eligible to develop ET programmes. • Offsets are only relevant in nonattainment areas.
<i>Commodity Traded:</i>	• Emission reduction credit.
Design Issues	
<i>Initial Allocation:</i>	• Not relevant.
<i>Geographic/Temporal:</i>	• Trades only within a given AQCR. • Banking is one of the ET programmes.
<i>Parties Trading:</i>	• Firms owning relevant sources.
<i>Institutions:</i>	• Brokers evolved to handle transactions, although no formal brokerage was established. • Some states developed offset credit registries that allowed firms to obtain offsets with fewer search costs.
Implementation Issues	
<i>Certification:</i>	• Detailed procedures are required to certify allowable credits.
<i>Monitoring/Reporting:</i>	• No additional requirements.
<i>Compliance/Enforcement:</i>	• Periodic facility inspections.
<i>Maintaining/Encouraging Participation:</i>	• State efforts are made to increase available offsets to encourage economic development.

- Threshold elements in the ET programmes

- *Emission goals.* The ET programmes were designed to provide additional flexibility in implementing the existing set of emission standards set by local areas and states in the State Implementation Programmes (SIPs). Thus, no specific emission reduction goals were set when the ET programmes were established. The offset programme for nonattainment areas is an exception. New sources obtaining offsets are required to obtain offsets than exceed emissions by a ratio of between 1.15 to 1.3, depending on the severity of nonattainment. This provision was designed to provide “something for the environment” in exchange for the right to add a new source in a region whose air quality exceeds national ambient air quality standards. For the other three ET programmes, however, the trades are designed to result in the same overall level of emissions.
- *Geographic area.* The offset programme is designed for nonattainment areas. The other three provisions can be utilised in attainment areas as well. Thus, in theory, the trading provisions can be used in any air quality control region (AQCR). In practice, states have differed a great deal in the extent to which they have implemented the programmes, with some states developing programmes targeted at facilitating trades. These programmes have typically been in nonattainment areas, where local authorities have assisted new firms to obtain offsets as a means of promoting economic development.
- *Commodity traded.* Each of the programmes generates Emission Reduction Credits (ERCs) for a given AQCR. Each credit gives the right to emit one ton of a specific pollutant.

- Design issues in the ET programmes

- *Initial allocation.* Not relevant.
- *Geographic/temporal.* ERCs can only be traded within a given AQCR. The banking programme allows ERCs to be banked for future use or sale by the firm. Banked permits and purchased permits, however, are subject to the risk that rules might change and permits might be taken or devalued.
- *Parties trading.* ERCs are generated and transferred within firms and between firms. Brokers have facilitated trading in the more active markets, such as Los Angeles (*e.g.* Foster and Hahn, 1996).

- *Institutions for trading.* Some states have developed programmes to assist firms in finding trading partners and to minimise administrative costs of applying for ERCs. States have developed registries for selling firms to deposit of ERCs until buyers can be found. States have also developed ERC funds to provide ERCs to small sources and essential public services, and to promote economic development.
- **Implementation elements in the ET Programmes**
 - *Certification.* Each proposed trade under the offset, netting, bubble or banking programmes must go through a rigorous evaluation by EPA or the state environmental enforcement agency before the trade can be approved. The most important and difficult part of this certification process is the approval of baseline emission levels from which reductions are measured. As discussed below, many sources routinely emit less than allowed by their permits or relevant standards. To deal with the problem of these firms applying for “paper credits” (*i.e.* credits for emission reductions that are already achieved), the baseline is typically set at allowable or actual emissions, whichever is smaller. A similar problem arises with “shutdown” credits for facilities that close operations because of concerns that these shutdowns would occur even without the credit incentive (see below). The process for making these baseline determinations is complex, and has resulted in most applications being disapproved.
 - *Monitoring and reporting.* Aside from the certification process, there are no additional monitoring or reporting requirements beyond those incorporated into existing air quality regulations.
 - *Compliance and enforcement.* Enforcement is implemented through periodic facility inspections to insure that pollution control equipment is installed, operating, and meeting the regulatory requirements.
 - *Maintaining and encouraging participation.* States have developed programmes aimed at facilitating use of the ET provisions. The most significant programmes have been aimed at providing offsets to promote economic development.

Evaluation of the ET programmes

The performances of the bubble programme and the other ET programmes have generally fallen far short of their cost-reducing potential. Table 2 presents estimates of the number of transactions that had occurred through 1986, as well as the cost savings achieved. On a proportional basis, relative to the hundreds of billions of dollars spent on air pollution control over the decade, the cost savings are modest. Most of the savings have come from the netting programme, which has led to about \$525-12 000 million in savings (in \$1986). No cost savings estimates can be developed for the offset programme, since offsets allow firms to enter the region rather than save on control costs. Savings from the other two programmes have been minimal. Bubbles have led to savings estimated at \$435 million, while banking savings have been very small.

The primary problem appears to be that ET has had many restrictions on allowable trades and many procedural requirements that raise the costs of trades and create uncertainty and delay.⁵ As a result of these problems, relatively few firms have bothered to try to trade, and virtually all trading has been

Table 2. **Summary of Emission Trading Through 1986**

Activity	Cost Savings (millions of \$1986)	Number of Trades
netting	\$525-\$12 000	5 000-12 000
offsets	Not applicable	2 000
bubbles	\$435	132
banking	Very small	100

note: 1986 dollars.

source: Hahn and Hester (1989, Table 2).

internal (*e.g.* a bubble within a plant or across two nearby plants owned by the same firm), rather than between companies.

The restrictions on trading have, in turn, reflected strong concerns on the part of many environmentalists and some government officials that trades may not be “legitimate”. To some extent, these concerns represent a philosophical opposition to trading: if a firm *can* control emissions, it should do so, and should not be rewarded for it. There is also a more practical element to the objections, however, which is based on uncertainty about the starting point from which ERCs should be given. Many sources routinely emit less than allowed by their permits or the relevant standards. Because of variability in the level of control actually achieved, firms must aim to do better than the standard if they are to be in compliance all (or virtually all) of the time. As a result, average performance typically is better than that required by the standard. Moreover, permits often set overall limits based on peak production levels, which rarely are achieved. Opponents of trading argue – with some justification – that giving firms “paper credits” for reductions below the levels allowed by standards or permits will result in increased overall emissions.

Similarly difficult issues arise in trying to determine “shutdown credits” for closing older, high-emitting plants. Closures of older plants – or reductions in plant output – can be a cost-effective “control” method, and thus, arguably, should be included in the ET programme. On the other hand, environmentalists point out that most closures are not designed to reduce emissions and would occur even if firms were not given ERCs. Moreover, under a credit-based system, closure or reduced output at one source may be compensated for by increased production – and thus emissions – at another source.

These and other concerns led to the development of detailed regulations governing ET transactions that were issued by the EPA in 1986. The 1986 trading regulations – which consumed 47 pages of multi-column fine print in the *Federal Register* – largely eliminated the risk of granting paper credits, but the price was a cumbersome system that makes it too expensive and uncertain to pursue many potential trades. The impact of the 1986 tightening of the rules can be seen in the fact that of the 40 bubble proposals reviewed under the new rules from 1986 to 1989, 31 – more than three-fourths – had been disapproved or were going to be disapproved (Creekmore 1989). Almost half of the disapprovals were based on the use of an “improper baseline”; the second most common reason was “enforceability”.

In sum, the results of the ET programmes generally have been disappointing. Although offsets continue to be needed – and used – in nonattainment areas, bubbles and the other elements have proved to be no more than small adjuncts to systems of emission standards.

Lead-in-gasoline

The programme used to regulate lead-in-gasoline during the mid-1980s offers a more successful example of the marketable permits approach. EPA regulated the lead content of gasoline starting in the early 1970s. Through 1982, lead limits were enforced on a refiner-by-refiner basis, with each refiner being allowed to average lead concentration across the gallons it produced. (In the language of the ET programmes, intra- but not inter-, plant bubbles were allowed.) In 1982, the rules were changed to allow trading across refiners. Under the new rules, a refinery could produce lead above the limit if it purchased a sufficient number of rights from refiners that produced lead below the lead limit.

In 1985, EPA promulgated a new rule to further reduce the lead limit more than ten-fold, in two steps, from 1.1 grams per leaded gallon (gplg) to 0.5 gplg in mid 1985, and then to 0.1 gplg on January 1, 1986 (50 *Fed. Reg.* 9400). As part of that rule, EPA allowed refiners to “bank” lead; if they reduced ahead of schedule during 1985, they could save the excess rights for use or sale in 1986 and 1987.⁶

Elements of the lead-in-gasoline programme

Table 3 summarises the major elements of the lead trading programme as it existed in its final version in the mid-1980s.

Table 3. Elements of the Lead-in-Gasoline Programme

Threshold Issues	
<i>Emission Goals:</i>	• Reduce average lead content in gasoline to near-zero levels (0.1 grams per leaded gallon).
<i>Geographic Area:</i>	• Entire US.
<i>Commodity Traded:</i>	• Quantity permits for lead used in gasoline production.
Design Issues	
<i>Initial Allocation:</i>	• Permits allocated quarterly based on leaded gasoline production in that quarter.
<i>Geographic/Temporal:</i>	• Banking was allowed for the latter portion of the programme. • No geographic differentiation.
<i>Parties Trading:</i>	• Firms (oil refineries). Alcohol “Blenders.” • Brokers.
<i>Institutions:</i>	• Mostly bilateral trading between oil refineries. • Some brokers trading between smaller “blenders” and refiners.
Implementation Issues	
<i>Certification:</i>	• Trades did not require pre-approval by EPA.
<i>Monitoring/Reporting:</i>	• Quarterly self-reporting of lead use and gasoline production.
<i>Compliance/Enforcement:</i>	• EPA determines compliance after trades made based on permit and lead use totals. • Permit “fungibility” (i.e. no permit IDs) makes tracing “bogus” permits difficult. • Most disputes administratively settled, with few court actions.
<i>Maintaining/Encouraging Participation:</i>	• No special effort made.

- Threshold elements in the lead-in-gasoline programme

- *Emission goals.* The lead phase-down programme involved a series of increasing restrictions on the lead content of gasoline. Standards eventually were reduced to 0.01 grams per leaded gallon (gplg) in 1986.
- *Geographic area.* Trading covered all refineries in the US.
- *Commodity traded.* Firms trade permits for lead used in gasoline production.

- Design issues in the lead-in-gasoline programme

- *Initial allocation.* Permits were allocated each quarter based upon the amount of leaded gasoline production that quarter.
- *Geographic/temporal.* There were no geographic restrictions on trading. There was some initial concern that, due to the greater geographic concentration of more modern refineries in the West, permits might migrate toward the East, where the marginal cost of lead reduction was higher. Although lead permits did migrate toward the East, an EPA analysis showed virtually no change in average lead concentrations. While not initially able to bank lead permits, starting in 1985, facilities could bank permits for future use or sale through 1987, when the programme ended. Banked permits could be held for any length of time and were not depreciated.
- *Parties trading.* The principal trading parties were oil refineries, though an unforeseen outcome was the large number of alcohol “blenders” who participated in the programme. These blenders – who blended alcohol with already-refined gasoline – could earn credits equivalent to those earned by refiners. Blending facilities were typically very small, however, and while the number of these blenders was large relative to the number of refineries, their share of the market’s volume was quite small.
- *Institutions for trading.* Most trading was directly between refineries, which had a long history of business relations and trading of inputs to production with one another. Brokers did, however, become involved in the credit market, in order to handle the trading between the smaller blenders and refiners.

- Implementation issues in the lead-in-gasoline programme
 - *Certification.* Trades did not require pre-approval by EPA.
 - *Monitoring and reporting.* Firms self-reported lead use and gasoline production each quarter.
 - *Compliance and enforcement.* Compliance was determined by EPA, which analysed whether the quantity of lead used by each firm was consistent with their level of gas production and the number of permits it owned. EPA might also make further compliance checks to determine whether reported records were true. These compliance measures were taken after the trades were made. Enforcement faced a number of difficulties. Because permits were “fungible” (*i.e.* not individually identified), it was difficult to trace “bogus” permits. Since the permit trades between parties could be complicated, determining who owned “bogus” permits was often difficult (Nussbaum 1996). Most disputes were worked out administratively, with very few going to court (Nussbaum 1996). EPA had to increase enforcement stringency after increases in cheating were observed, especially after the banking programme was implemented. One GAO report found 35 per cent of reports contained errors, including both deliberate errors and mistakes (Kerr and Maré, 1997).⁷
 - *Maintaining and encouraging participation.* No special efforts were made to encourage participation.

Evaluation of the lead-in-gasoline programme

The lead programme appears to be a success, both when trading was initially permitted in the 1983-85 period and when banking was added in the 1985-87 period. From mid-1983 (when the new rules took full effect) until early 1985 (when a further phase-down began), an increasingly vigorous market in lead rights developed. In a typical quarter, over half of all refineries participated in the market, and up to one-fifth of the lead rights were traded (Hahn and Hester 1989). In 1985, when banking was added, an even larger fraction of lead was bought and sold in the market. No estimates were made of the cost savings from trading prior to 1985, but refiners were enthusiastic about the programme.

Banking also appears to have been very successful. EPA predicted that refiners would bank 7 to 9 billion grams of lead, resulting in savings of up to \$226 million (in 1985 dollars, discounted at a 10 per cent real rate) over two and one-half years, or about 20 per cent of the estimated cost of the rule over that period (US Environmental Protection Agency 1985b). Although no new estimates of the cost savings were made after banking occurred, the level of banking was even higher than predicted – 10.6 billion grams, almost 17 per cent higher than the upper end of the predicted range (Hahn and Hester 1989). Thus, it seems likely that the actual savings were higher than the estimate. The large increase in trading after the banking programme was put in place shows the importance of banking to a trading programme.

While the programme generated a large number of trades and significant cost savings, the majority of trades were made within firms. In the sample of firms for which they obtain data (65 per cent of the industry capacity), Kerr and Maré find that 67 per cent of the quantity purchased was purchased within the company, while 70 per cent of the quantity sold was sold within the company.⁸ They suggest that the large proportion of internal trades can be attributed to the lower transaction costs (particularly search and negotiation costs) for internal trades than for trades with other firms.

The EPA's difficulty with pursuing enforcement actions and the large number of improperly-filed reports found by the GAO suggest that there may have been some cheating that could potentially have compromised the environmental objectives of the programme. There is no evidence, however, that these problems led to significant excess lead emissions. On balance, the benefits of the accelerated phase-down allowed by the trading programme outweigh the leakage that occurred through enforcement problems.

Trading and banking were very successful and widely accepted in the case of lead-in-gasoline for several reasons:

- after 1982, there were no constraints on trading among refiners; any refiner could trade with any other;
- administrative requirements were minimal, with no preapproval required – refiners reported trades in their normal quarterly reports to EPA;

- the implicit allocation of allowable lead each quarter was clear and easy to determine (based on that quarter's production of leaded gasoline); and
- in both 1982 and 1985, the trading/banking provisions were enacted as parts of other changes that tightened the law.

As a result, trading and banking were widely viewed as being important elements of a cost-effective strategy to reduce lead-in-gasoline.

Ozone-depleting chemicals

As part of its implementation of the Montreal Protocol for control of ozone-depleting chemicals – instituted by the US on April 21, 1988 – EPA set up a system of fully tradable production quotas (which we shall refer to as “allowances”) to phase-down the use of chlorofluorocarbons (CFCs), halons, and several other ozone-depleting substances (ODS) (58 *Fed. Reg.* 65018).⁹ Trade in allowances was one approach taken to minimising the costs of this phase-out, which was implemented through a series of declining caps over the period 1989 to 2000. Trading programmes exist at both the international and domestic levels, though the rules, procedures, and institutions differ. International trade is enabled through provisions allowing “industrial rationalisation” of production, where multiple countries can jointly meet their production target. Domestic or regional trading between firms, however, must be enabled by legislation in individual countries (or in regional groups such as the European Union). In the US, the EPA administers a programme that allows firms to trade the allowances they are allocated. This paper focuses on the US programme, and the international programme to the extent that it affects trading of US firms.

Elements of the ozone-depleting chemicals programme

Table 4 summarises the main elements of the trading programme for ozone-depleting substances developed in the US.

Table 4. **Elements of the Ozone-Depleting Substances Programme**

Threshold Issues

- | | |
|--------------------------|--|
| <i>Emission Goals:</i> | <ul style="list-style-type: none"> • Zero production of certain CFCs by 1998. • Eventual elimination or near elimination of production of all ozone-depleting substances (ODSs). |
| <i>Geographic Area:</i> | <ul style="list-style-type: none"> • Global (Montreal Protocol). • Entire US (EPA trading programme). • European Union, Singapore and New Zealand also have trading programmes. |
| <i>Commodity Traded:</i> | <ul style="list-style-type: none"> • Allowances to make or import ODS in the US. • Trading across ODS allowed, based on ODPs. • “Industrial rationalisation” (Montreal Protocol) allows trading with international firms. • Excise tax on ODS sales in US. |

Design Issues

- | | |
|-----------------------------|--|
| <i>Initial Allocation:</i> | <ul style="list-style-type: none"> • Based on 1986 ODS market share. |
| <i>Geographic/Temporal:</i> | <ul style="list-style-type: none"> • Banking allowed. • No geographic differentiation. |
| <i>Parties Trading:</i> | <ul style="list-style-type: none"> • Trading between international and domestic producers of ODS. |
| <i>Institutions:</i> | <ul style="list-style-type: none"> • Bilateral trading. |

Implementation Issues

- | | |
|---|---|
| <i>Certification:</i> | <ul style="list-style-type: none"> • EPA certification with a three-day response. • Embassy confirmation for international trades. |
| <i>Monitoring/Reporting:</i> | <ul style="list-style-type: none"> • Self-reporting of production levels, imports, exports and “recycled” ODS. |
| <i>Compliance/Enforcement:</i> | <ul style="list-style-type: none"> • Penalty of \$25 000 per kilogram. • Possible illegal labelling of imported and exported ODS as “recycled.” |
| <i>Maintaining/Encouraging Participation:</i> | <ul style="list-style-type: none"> • None. |

- Threshold issues of the ozone-depleting substances programme
 - *Emission goals.* The Protocol has the goal of eliminating production of ODS to near-zero levels. Actual reduction levels depends upon the specific ODS and are less stringent for developing countries. The reduction schedule has accelerated as subsequent scientific information has suggested greater damages from ozone depletion and lower costs of a phase-out. For CFCs the limits were 100 per cent of 1986 levels by 1989, 25 per cent of 1986 levels by 1994, and 0 per cent (i.e. elimination) by 1998.
 - *Geographic area.* International trading is possible with any of the signatory countries to the Protocol through the “industrial rationalisation” provision. These trades are implemented by confirming that the international firm’s home country deducts the appropriate number of allowances from their overall obligation. Signatories include most of the ODS-producing countries. Signatories are prohibited from any trade either allowances or stocks of ODS with non-signatories. Domestic trade is allowed between any US producer, importer or exporter. The European Union, Singapore and New Zealand also have trading programmes.
 - *Commodity traded.* The US programme created production and consumption (production plus imports minus exports) allowances that entitle firms to make or consume ODS. Initially, a “basket” approach was used where companies could make any combination of ODS, using the ozone-depleting potentials (ODPs) as weighting factors. In 1990, allowances were distributed for each ODS separately, though inter-pollutant (ODS) trades could be made using the ODP. A premium of 1 per cent of allowances traded was taken for all trades.
- Design issues in the ozone-depleting chemicals programme
 - *Initial allocation.* Based upon the phase-out schedule in the Protocol, the US has a fixed number of allowances to allocate to firms within the US. The US programme distributed allowances domestically based upon 1986 ODS market share. This initial distribution was limited to only 28 firms (five CFC producers, three halon producers, fourteen CFC importers and six halon importers) (Cook, 1996). Distributions in subsequent years decline proportionally for all firms. The distribution of allowances resulted in windfall profits to the producers who received the allowances. An excise tax on ODS sales was imposed to reduce this windfall.¹⁰ The tax started at \$1.37 per pound in 1990 and rose to \$5.35 per pound in 1995.
 - *Geographic/temporal.* There are no geographic restrictions on trades. Firms are not allowed to bank unused production allowances for use in future years. They may, however, stock-pile ODS produced with production allowances in one year for consumption in a future year.
 - *Parties trading.* Trading occurs between producers, importers and exporters of ODS.
 - *Institutions for trading.* Most trades occur through bilateral agreements between producers, exporters, and importers. The limited role of brokers in the ODS allowance market may be due to the small number of firms and the pre-existing relations between them.
- Implementation issues in the ozone-depleting chemicals programme
 - *Certification.* Before a trade occurs, the trading firms must notify the EPA and provide the following information: firms’ names, types of allowances being traded, type of ODS, control period and remaining allowances for each firm. EPA checks the allowance balances for each firm and responds within three days. When a US firm trades with a firm in another country, the EPA will only approve the trade after the country’s embassy has confirmed that it has adjusted the level of its production rights under the Protocol.
 - *Monitoring and reporting.* Producers, exporters and importers self-report information on the level of production, imports, exports and “recycled” ODS. Since recycled ODS are not included in the measurement of total ODS consumption under the Protocol, recycling and reusing ODS represents one strategy for maintaining consumption even as production is eliminated.

- *Compliance and enforcement.* The EPA maintains a computerised ODS allowance tracking system which is used to both insure compliance and verify that firms have adequate allowance balances to make trades. Each firm is responsible for insuring that their actual production is within their account balances. Thus, sellers are liable if their production is in excess because of having sold too many allowances. The penalty for non-compliance in the US is a fine of \$25 000 per kilogram. One enforcement problem that has emerged is the labelling of imported and exported ODS as “recycled” to avoid consumption restrictions. There have been anecdotal reports of this abuse.
- *Maintaining and encouraging participation.* The Protocol has prohibitions on trade in ODS within countries that have not signed the Protocol. One rationale for this provision is to provide an incentive for participation, since most of the large markets for ODS are in countries that have signed the Protocol.

Evaluation of the ozone-depleting chemicals programme

The programme appears to be a success. Trading in ODS has been substantial. A total of 321 million kilograms (in 561 trades) were traded by firms within the US over the period of 1989 to 1995. The 171 Mkg traded in 1992, the most active trading year, represented about 13 per cent of the total allowances in that year. About 60 Mt of ODS involving 160 trades was transferred *within* firms. A large portion of these trades were likely “inter-pollutant” trades. As a result of inter-ODS trades, the level of production of some ODS exceeded the 25 per cent cap on all ODS, while the production levels of other ODS was below the cap. International trading has been relatively limited, with only 20 trades over the period 1992 to 1995 totalling 36 Mkg of ODS.

No estimates have been developed of the cost-savings from trading. An earlier study estimated that a marketable quota scheme to reduce CFC emissions would cost about 49 per cent less than an equally effective system of mandatory controls (Palmer *et al.*, 1980).

For a variety of reasons, the US reduced ODS production more quickly than required under the Montreal Protocol. There have been claims that market mechanisms have helped promote these early reductions (*e.g.* Lee, 1996), although the rationale for this is less clear. Early reductions appear to have been the result of cost-savings due to technological change and the development of substitutes, rather than cost-savings due to trading.

A number of factors explain the success of this programme. First, the number of potential traders is small and they have experience in business dealings with each other. These factors lower transaction costs. Second, the certification process for trades is relatively simple, also reducing transaction costs. Third, inter-pollutant trading is permitted and is relatively simple. A large portion of the savings from trading result not from differences in marginal costs of production across plants, but differences in the market value of different ODS.

While there has been significant domestic US trading, as noted, international trading has been less active. Potential explanations for the differences between domestic and international trading include differences in the level of previous business experience, differences in the transaction costs of implementing trades with international partners, and perceived differences in the likelihood of getting trades approved.

Major existing programmes

Two major tradable permits programmes were developed in the early 1990s, the SO₂ allowance trading programme mandated by the 1990 Clean Air Act Amendments and the Regional Clean Air Incentives Market (RECLAIM) developed for the Los Angeles air basin.

The SO₂ allowance trading programme

Title IV of the 1990 Clean Air Act calls for the creation of an allowance trading system for SO₂ emissions from electric utilities. The programme caps the total number of allowed SO₂ emissions and allows utilities to trade the SO₂ allowances.

Elements of the SO₂ allowance trading programme

Table 5 summarises the main elements of the SO₂ allowance programme.¹¹

Table 5. Elements of SO₂ Allowance Trading Programme

Threshold Issues	
<i>Emission Goals:</i>	• Reduce annual emissions from electricity generation sector by 50 per cent.
<i>Geographic Area:</i>	• Entire US.
<i>Commodity Traded:</i>	• SO₂ emission allowances.
Design Issues	
<i>Initial Allocation:</i>	• Basic formula is based upon average heat input for 1985-1987. • Many details.
<i>Geographic/Temporal:</i>	• None.
<i>Parties Trading:</i>	• Electric utility generating facilities. • Some non-utility boilers that voluntarily opt-in. • Brokers.
<i>Institutions:</i>	• Bilateral trading, brokers, and EPA auctions of additional permits. • Information sources for permit prices included reports of several early trades, broker reports, and EPA auctions.
Implementation Issues	
<i>Certification:</i>	• No certification of trades is required, beyond reporting account transfers to the EPA.
<i>Monitoring/Reporting:</i>	• Continuous emissions monitoring (CEM). • EPA maintains allowance accounts.
<i>Compliance/Enforcement:</i>	• Trade quantities and buyer and seller information are reported to EPA. • Compliance determined annually. • EPA computer system compares actual emissions data to allowance holdings. • Fine of \$2 500 and a one allowance reduction in the following years allocation for each excess ton of SO₂.
<i>Maintaining/Encouraging Participation:</i>	• A number of opt-in provisions exist to increase participation in the programme. • Some non-electric sectors may participate in trading. • Phase II plants may participate in Phase I trading.

- Threshold issues in the SO₂ allowance trading programme

- *Emission goals.* The programme provides for a 50 per cent reduction in SO₂ emissions from the electric utility generating sector from 1980 levels. The eventual cap on annual emissions is 8.95 million tons (Mt) SO₂ per year in the year 2010, though the reductions are achieved gradually over time. Over the implementation period, the size of the cap is gradually constrained as the number of utilities affected is expanded. The cap on Phase I plants (263 units) starts at 7.1 Mt in 1995 and drops to 6.0 Mt in 1999. In 2000, Phase II plants are added, making virtually every fossil-fuel fired electric generating units subject to the cap. At this point, the cap rises to 9.48 Mt per year (including bonus allowances for installing flue gas desulfurization systems). Over the next ten years, the cap falls to the eventual target of 8.95 Mt cap as the bonus allowances are eliminated.
- *Geographic area.* Trading covers the entire US with no geographic distinction between permits.
- *Commodity traded.* The commodity traded is an SO₂ emission allowance, which entitles the owner to emit one ton of SO₂.

- Design issues in the SO₂ allowance trading programme

- *Initial allocation.* Allocation is based upon each unit's average heat input during the period 1985-87, although many exceptions were made for particular units, and additional allowances were provided to units designated as "extension", "substitution", and "compensation" units (see Joskow, Schmalensee and Bailey, 1997). Allocations can be changed in future years (*i.e.* the allowances are

not “property rights”), although political considerations suggest this is unlikely. Allowances also are available through an annual EPA auction for a small portion (2.8 per cent) of total allowances. New sources must purchase allowances through the EPA auctions or from allowance holders.

- *Geographic/temporal.* There are no geographic restrictions on trade. The initial proposals included restricting trades to subregions, but no restrictions were included in the eventual programme. Allowances may be banked for use or trade in future years. The feature is particularly important because the Phase II cap is significantly more restrictive than the Phase I cap.
 - *Parties trading.* A large number of groups have participated in trading including electric utilities, brokers, fuel providers, and other groups. Some suppliers of high-sulfur coal have begun to bundle coal with allowances to enhance the marketability of their product.
 - *Institutions for trading.* A number of options are available for buying and selling allowances, including bilateral trading, brokers, and EPA auctions of permits. The growth in brokers has been significant since the programme was first implemented. Currently, about 80 per cent of external trades are made through brokers. Bilateral trading has diminished over time. As noted, the EPA holds annual auctions for about 2.8 per cent of the allowances. Publicly-available sources of allowance price information contributed to the functioning of the permit market, particularly in its initial stages. These sources include reports of early trades, regular broker reports, and EPA auctions. One justification for EPA auctions was to provide price information to the market. Some analysts indicate that bilateral trading now informs the auctions, rather than *vice versa* (Joskow, Schmalensee and Bailey, forthcoming).
- Implementation issues in the SO₂ allowance trading programme
 - *Certification.* Firms are not required to pre-certify trades, although they must report account transfers to the EPA.
 - *Monitoring and reporting.* For all coal-burning units, utilities must install continuous emissions monitoring (CEM) which hourly measure the emissions rates for SO₂, NO_x, and CO₂. Such units are expensive, costing about \$120 000 per stack. Oil and gas fired units can use less expensive monitoring equipment. Each unit self-reports its hourly data to the EPA on a quarterly basis. EPA maintains an Allowance Tracking System (ATS) which is a system of allowance accounts for individual units, general utilities, brokers, fuel suppliers and other parties. Any change in account activity, including allowances trades or reallocations within utility accounts, must be reported. The reported information includes trade quantities (but not price), and buyer and seller account information. This transaction usually takes less than 30 minutes and is processed by EPA within 24 hours.
 - *Compliance and enforcement.* Compliance is determined automatically each year by a computer system that compares account balances for each unit in the ATS against an estimate of total emissions based upon the hourly CEM data (or comparable for other units). Sellers are implicitly liable for permits, since enforcement is based on having enough allowances to cover actual emissions. If a unit sells an allowance, but does not make the necessary reductions, they will be out of compliance – not the firm that purchased the allowance. The penalties for non-compliance are severe. A fine of \$2 500 and a one-allowance reduction in the following year's allocation is assessed for each ton of emissions in excess of account balances. The level of the fine is large relative both to expected permit prices before the programme began (which ranged from \$300 to \$800 per ton) and to current prices (which range from \$70 to \$200 per ton).
 - *Maintaining and encouraging participation.* A number of opt-in provisions exist to increase participation in the programme, particularly during Phase I. Phase II plants were allowed to voluntarily participate in Phase I trading, so long as they agreed to the same or similar provisions as targeted facilities. As a result, 182 Phase II units, representing 47 per cent of Phase I capacity, entered the programme. Most of these units (175) entered as “substitution” units which meant that they replaced the reduction obligations of an original Phase I unit. These units entered because their emissions were lower than allowance rates or because they were under-utilised or about to be

retired. In addition, many faced falling SO₂ reduction costs due to the availability of low-sulfur coal. Some non-electric sources were allowed to participate in trading, although few have entered the programme because of the high transaction costs.

Evaluation of the SO₂ allowance trading programme

The SO₂ trading programme has developed an effective market for SO₂ allowances. Trading volumes have grown with time and are a significant fraction of the total allowed under the cap.

Table 6 lists the total number of allowances sold through EPA auctions, total allowance transfers, and transfers between economically distinct organisations. Initially, the number of trades directly between utilities and between brokers and utilities were about the same. Over time, the proportion of trades performed by brokers has grown to about four to five times the quantity of direct utility trades. Not all of the trades can be considered cost-saving actions by utilities, since large fractions are broker-to-broker (over 20 per cent) and utility-to-broker (over 40 per cent) trades. In addition, many of these transfers are for allowances to emit in future years (up to eight years into the future). In 1995, its first year, only about 6 per cent of allowances led to cost reductions in 1995, though another 39 per cent were banked for future use (Ellerman *et al.*, 1997).

Table 6. **Allowance Transfer Through Trading and EPA Auctions**

	Total transfers		Transfers between economically distinct organizations ¹		Transfers within organizations ²		EPA Auctions
	Transactions	Allowances (millions)	Transactions	Allowances (millions)	Transactions	Allowances (millions)	Allowances (millions)
994	215	9.2	66	0.9	149	8.3	0.18
995	613	16.7	329	1.9	284	14.8	0.18
996	1 074	8.2	578	4.4	496	3.8	0.28
997	1 429	15.2	810	7.9	619	7.3	0.30

. Transfers between economically distinct organisations include transfers between utilities, between brokers and utilities, between utilities and fuel providers, and between brokers and fuel providers.

. Transfers within organisations include transfers reallocations from unit specific accounts to various general accounts and intra-utility trades between unit specific accounts.

source: US EPA (1998).

After an initial period of volatility, reported prices have tended to operate in a small range. Using data from five reported sources, Ellerman *et al.* (1997) show that after some initial volatility, reported prices converged by mid-1994 to a market equilibrium of about \$150 per ton. Consequently, prices had converged and information on prices was publicly available by the time the Phase I cap came into place in 1995. While the price has fluctuated over the last few years, all sources, including EPA auctions, report prices at roughly the same levels. This convergence corresponds with an increase in the sales volume of allowances. Prices started at about \$140/ton in 1995, then dropped to about \$70/ton in early 1996, before recently rising back to \$140/ton. Recent price increases and increases in trade volumes may reflect increased demand as the more stringent Phase II cap in 2000 approaches.

Actual permit prices have been significantly lower than pre-implementation estimates of Phase I permit prices (\$250-350). Several factors explain the lower-than-expected allowance prices. These factors include the greater economic viability of low-sulfur coal from the west due to railroad deregulation, cost-saving innovations in scrubbing, political pressure to invest in scrubbing (particularly in high-sulfur coal regions), and expectations of higher permit prices which drove early utility investments in scrubbing before long-run market prices were revealed. Analysis of cost savings is complicated by many factors, including ambiguity over the proper counter-factual regulatory requirement, the presence of banking,

voluntary opt-in of Phase II units, limited information on control costs for units using allowances, and uncertainty on the effects of trading on inducing cost-reducing innovations. Given these caveats, Ellerman *et al.* (1997) provide tentative estimates of the savings in 1995 of \$225-375 million dollars, or about 25-34 per cent of the cost of abatement without trading and the same initial distribution of allowances.

In sum, the SO₂ allowance trading programme appears successful. Many trades have taken place, leading to substantial overall cost savings relative to a command-and-control programme.

Southern California RECLAIM programme

Regulators in the Los Angeles air basin developed a tradable permits programme in the early 1990s as the SO₂ allowance programme was being formulated. The programme is called the Regional Clean Air Incentives Market (RECLAIM), an acronym designed to emphasise the objectives of reclaiming both good air quality and a sound economy. The programme was approved by the South Coast Air Quality Management District (SCAQMD) in October 1993, after a regulatory development period that lasted almost three years. RECLAIM began operation in January 1994.

Elements of the RECLAIM programme

Table 7 summarises the major elements of RECLAIM.¹²

- **Threshold issues of the RECLAIM programme**
 - *Emission goals.* RECLAIM was designed to implement provisions of the air quality management plans developed by the SCAQMD to bring the Los Angeles Basin into compliance with federal air quality standards by 2010. The specific emissions goals for RECLAIM were based upon the requirements established for RECLAIM facilities under the 1991 Air Quality Management Plan. These requirements translated into a weighted average annual reduction for all RECLAIM facilities from 1994 to 2003 of 8.3 per cent for NO_x and 6.8 per cent for SO₂.
 - *Geographic area.* The South Coast Air Quality Management District in California consists of a 6 600 square mile area containing the non-desert portions of four Los Angeles area counties with a combined population of about 13 million inhabitants in 1990. A combination of emission sources, meteorology, and terrain make this area the most heavily polluted region of the US as measured by urban ozone.
 - *Commodity traded.* The basic commodity consists of a Reclaim Trading Credit (RTC), which is the right to emit one pound of a specific pollutant (SO_x or NO_x) in a specific year. The SCAQMD initially considered defining the commodity as an emission reduction credit, but changed to the allowance formulation. The unit of trade in RECLAIM is thus equivalent to the unit established in the SO₂ trading programme. RECLAIM also allows for the creation of RTCs through credit programmes for mobile sources and area sources.
- **Design issues in the RECLAIM programme**
 - *Initial allocation.* Determining the formula to allocate initial RTCs to sources proved to be contentious and complex but ultimately manageable (see Harrison, forthcoming). The final allocations were based upon complicated formulas in which each facility received three sets of allocations of pounds of NO_x and/or SO_x: A starting allocation for 1994; a mid-point allocations for 2000; and an ending allocation for 2003. The basic ton allocations are based upon multiplying an appropriate emission factor (*i.e.* pounds per million Btu of energy input) for each of the three years by a single value for historic throughput or usage that is determined by each facility's peak activity over the period from 1989 to 1992. The emissions rates for each of the allocation years are based on adopted rules, as of December 1993, for each facility. The 1994 allocations are supplemented by the offsets facilities had obtained to comply with new source review requirements. In addition, facilities were given non-tradable credits for the first three years of the programme, if their

Table 7. Elements of the RECLAIM Trading Programme

Threshold Issues	
<i>Emission Goals:</i>	- Declining cap on emissions from RECLAIM facilities from 1994 to 2003. Constant cap after 2003. - Annual weighted reduction equal to 8.3 per cent per year for NO_x emissions and 6.8 per cent per year for SO₂ emissions.
<i>Geographic Area:</i>	Los Angeles air basin.
<i>Commodity Traded:</i>	RECLAIM Trading Credit (RTC), the right to emit one pound of a specific pollutant in a specific year.
Design Issues	
<i>Initial Allocation:</i>	- Complicated formula for RECLAIM facilities based upon three sets of allocations: a starting allocation for 1994; a mid-point allocation for 2000; and an ending allocation for 2003. - Allocations based upon emission factors and historic activity over the period from 1989 to 1992. - Additional allocations provided to deal with equity concerns.
<i>Geographic/Temporal:</i>	- Two geographic zones: coastal zone facilities can apply only coastal RTCs; inland facilities can apply either coastal or inland RTCs. - No formal banking; overlapping trading cycles (January-December and July-June) allow temporal flexibility.
<i>Parties Trading:</i>	- Facilities emitting more than four tons of NO_x or SO₂ are required to participate. Some categories of facilities excluded. - Other facilities can opt in to the programme. - Mobile sources and area sources can generate RTCs.
<i>Institutions:</i>	- Auctions. - Brokers. - Electronic bulletin board.
Implementation Issues	
<i>Certification:</i>	- No pre-approval of trades is required for allocated RTCs. - Mobile source and area source RTCs require certification.
<i>Monitoring/Reporting:</i>	- Large sources use continuous emissions monitoring (CEMs) to measure mass emissions. - Smaller sources have flexibility to use fuel meters and conversion factors to compute mass emissions.
<i>Compliance/Enforcement:</i>	- Quarterly reporting of transactions; Annual Permit Emissions Programme (APEP) report must be submitted within 60 days of end of each compliance year. - APEP reports are audited for accuracy and compliance.
<i>Maintaining/Encouraging Participation:</i>	- Additional sources are encouraged to opt-in to RECLAIM. - Regulations were developed to include credits for reductions from mobile sources and area sources.

reported 1987, 1988 or 1993 emissions were greater than their starting allocation. Allocations for each intermediate year for each facility were calculated based upon straight-line interpolation between the 1994, 2000, and 2003 allocations. The allocations were constant for 2003 and thereafter. The final step was to adjust the total allocations to equal the overall targets in 2002 and 2003 set under the AQMP for the source categories covered by RECLAIM.¹³

- *Geographic/temporal.* Because ozone formation takes several hours, upwind sources located near the coast contribute more to elevated ozone levels in the Los Angeles basin than downwind sources located inland. The SCAQMD originally considered dividing the region into the 38 zones used for offset transactions. The final RECLAIM programme involved a two-zone system in which the air basin was divided into coastal (upwind) and inland (downwind) zones. Inland zones can obtain and use RTCs from either zone, while coastal sources can only obtain and use RTCs that originate in the coastal zone. There is no formal programme for banking. However, facilities are split into two groups with a staggered compliance schedule, in order to minimise shocks to market prices during reconciliation periods.
- *Parties trading.* RECLAIM includes most facilities in the Los Angeles basin that emit four tons or more of NO_x or SO₂. Some exceptions include essential public facilities and some entire industry groups that emit less than four tons per year (*e.g.* restaurants). These criteria resulted in a total of 390 NO_x

and 41 SO₂ facilities at the start of the programme. The universe of facilities has changed over the course of the programme due to facility exclusions and inclusions (see South Coast Air Quality Management District 1998).

- *Institutions for trading.* A number of institutions have emerged. First, brokers facilitate trades and purchase RTCs outright. Second, the SCAQMD has held auctions in which RTC owners can place their RTCs up for bid. Third, a bulletin board system is operated by the SCAQMD that contains information on potential buyers and sellers and their proposed terms of trade.
- **Implementation issues in the RECLAIM programme**
 - *Certification.* Pre-approval of trades is not required, although trades must be registered with the SCAQMD. Mobile source and area source RTCs do require certification.
 - *Monitoring and reporting.* Major generators are required to use Continuous Emissions Monitoring (CEMs). Smaller facilities are allowed to use fuel meters and conversion factors to compute mass emissions. Certifying the accuracy of the CEMs data is a major difficulty in implementing the programme.
 - *Compliance and enforcement.* Compliance is determined annually, with a two-month reconciliation period. The compliance permit under the RECLAIM programme fulfils the permit requirements under Title V of the Clean Air Act.
 - *Maintaining and encouraging participation.* The SCAQMD allows additional firms to opt in to RECLAIM. Facilities that opt in are given allocations based upon the same criteria as the original facilities. In addition, the SCAQMD has developed regulations that allow RECLAIM participants to obtain RTCs for reductions in mobile source and area source emissions. For example, sources can obtain RTCs for scrapping older motor vehicles.

Evaluation of the RECLAIM programme

RECLAIM appears to be successful in keeping overall emissions from RECLAIM facilities below the targets, and in facilitating an active market in RTCs. Table 8 summarises emissions from RECLAIM facilities for each of the first three compliance years. The table shows that RECLAIM facilities did not exceed their allocations on an aggregate basis during the first three compliance years. Indeed, aggregate emissions were comparable to the levels that were projected to occur under continuation of command-and-control regulations.

Table 9 summarises the overall trading activity by pollutant. More than 1 200 trades totalling 244 000 tons of NO_x and SO_x RTCs were transferred in the market as of December 31, 1997. These trades included both RTCs traded with prices and transfers with a zero price, such as transactions between two facilities under common ownership. The total value of all reported RTCs traded was about \$42 million over the four years since the programme's inception.

Table 8. **Annual RECLAIM Emissions for the 1994 Through 1996 Compliance Years¹**

	1994		1995		1996 ²	
	NO _x	SO _x	NO _x	SO _x	NO _x	SO _x
Annual Emissions (ton)	25 314	7 232	25 764	8 064	24 200	6 484
Total RTCs (ton) ³	40 017	10 363	35 935	9 610	31 935	8 892
Unused RTCs (ton)	14 813	3 133	10 267	1 548	7 735	2 410
Percent Unused RTCs	37%	30%	28%	16%	24%	27%
Percent Change in Emissions from 1993 ⁴	1.3%	0.9%	3.1%	12.5%	–3.1%	–9.5%

1. The RECLAIM universe is divided into two cycles with compliance schedules staggered by six months. Compliance years for Cycle 1 facilities run from January 1 through December 31 and Cycle 2 compliance years are from July 1 through June 30.

2. 1996 data has not been fully audited for accuracy.

3. Total RTCs = Allocations + Converted ERCs.

4. 1993 Emissions: 24 982 tons for NO_x and 7 467 tons for SO_x.

Source: South Coast Air Quality Management District (1998).

Table 9. RECLAIM Trading Activity by Pollutant (1994-1997)

	RTCs Traded With Price	RTCs Traded Without Price
	NO _x	
trades	352	680
ons	28 662	152 932
ollar Value (millions)	\$23.6	NA
	SO _x	
trades	51	112
ons	13 505	49 123
ollar Value (millions)	\$18.0	NA

source: South Coast Air Quality Management District (1998).

In summary, the RECLAIM programme appears to be a success. Overall emissions have been kept below the cap set for the first years of the programme. Trading activity has been substantial. When the programme was being developed, the cost savings from RECLAIM were estimated at about 30 to 40 per cent relative to the command-and-control alternative (see Harrison, forthcoming). The programme appears to be on track to achieve these substantial gains.

Recent programmes

Two tradable permits programmes have been developed recently that build on the previous programmes.¹⁴ The first is the set of averaging, banking and trading (ABT) programmes developed for various mobile sources that resemble the earlier lead permits programme. The second is a trading programme for NO_x emissions in Northeast States, similar to the SO₂ trading programme. Although not yet implemented, these programmes provide additional insights into the decisions needed as the theory of tradable permits is put into practice.

Mobile source averaging, banking and trading (ABT) programmes

The US Environmental Protection Agency has promulgated regulations for exhaust emissions from various categories of mobile sources. These regulations set emission limits for new engines in various categories, including heavy-duty trucks and buses, gasoline-powered marine engines (including outboard engines and personal watercraft engines), and small utility engines (used in lawn and garden and other applications). Each of these regulations has included an averaging, banking and trading (ABT) programme.

Elements of the mobile source averaging, banking and trading (ABT) programmes

Table 10 summarises the main elements of the ABT programme. To be concrete, the table and related discussion summarise the features of the ABT programme promulgated specifically for marine engines.¹⁵

- Threshold issues in averaging, banking and trading (ABT) programmes
 - *Emissions goals.* ABT programmes have been developed as part of the initial regulatory requirements for these emissions sources. The emission goals vary. In the case of emissions from marine engines, the emission standards require increasingly stringent controls on hydrocarbon (HC) emissions over the course of a nine-year phase-in period beginning in model year 1998. By the end of the phase-in, each manufacturer must meet an HC + NO_x emission standard on a corporate basis that represents a 75 per cent reduction in HC compared to unregulated levels.

Table 10. Elements of Averaging, Banking and Trading (ABT) Programmes

Threshold Issues	
<i>Emission Goals:</i>	• Various reduction targets (e.g. marine: 75 per cent).
<i>Geographic Area:</i>	• US.
<i>Commodity Traded:</i>	• HC + NO _x emissions.
Design Issues	
<i>Initial Allocation:</i>	• Production volume (averaging programme).
<i>Geographic/Temporal:</i>	• Banking (advance and future).
	• Limited borrowing.
<i>Parties Trading:</i>	• Firms (internal and external).
<i>Institutions:</i>	• None.
Implementation Issues	
<i>Certification:</i>	• Family emission limit (FEL) for each engine model.
<i>Monitoring/Reporting:</i>	• Production line and in-use testing required.
<i>Compliance/Enforcement:</i>	• Annual report for production-level compliance.
	• In-use credit programme.
	• Recall programme as backstop.
<i>Maintaining/Encouraging Participation:</i>	• None.

- *Geographic area.* The ABT programme relates to emissions for marine engines produced for sale in the US.
- *Commodity traded.* The ABT programme involves the trading of credits and debits for emissions from different engine families. The calculation of credit or debit for a given engine family in a given model year is based upon the following factors: the difference between the applicable emission standard and the engine family emission limit (FEL), sales in the applicable model year, average annual use in hours, the power output of the engine family, the future survival probability of each engine, and the net present value of the credits or debits over time.
- Design issues in averaging, banking and trading (ABT) programmes
 - *Initial allocation.* ABT is an averaging programme that does not impose a cap on total emissions. In effect, manufacturers are given allocations based upon production volume.
 - *Geographic/temporal.* The programme does not make any distinctions for engines sold in different geographic areas. The regulations include provisions for banking as well as for early banking, which would allow manufacturers to earn partial credit for reductions in model year 1997. In addition, the regulations allow manufacturers multi-year averaging. Manufacturers who do not cover emission debits in model years 1998 or 1999 can make up the required reductions by model year 2000. (There are some limits on the percentage of reductions that can be made up for the two years.) This allows some limited “borrowing” of emissions.
 - *Parties trading.* The ABT provisions allow manufacturers to trade internally as well as with other manufacturers. There are, however, restrictions on allowable trades across the two major engine categories in the first years of the programme. For the first three years, the averaging sets for outboard engines and personal watercraft are separated. In other words, manufacturers are not allowed to use credits for personal watercraft to offset outboard debits (and *vice versa*) before the model year 2001.
 - *Institutions for trading.* No institutions are established under the regulations to facilitate trading.
- Implementation issues in averaging, banking and trading (ABT) programmes
 - *Certification.* All gasoline marine engines must be certified by EPA as meeting applicable emission standards before they can be introduced into commerce. In order to meet this requirement, manufacturers must submit an Application for Certification that identifies the engine family emission limit (*i.e.* FEL). The calculation of the FEL is based on various procedures depending upon the engine technology and model year.

- *Monitoring and reporting.* All manufacturers must implement certification procedures, production line testing, and in-use testing.
- *Compliance and enforcement.* The regulations provide flexibility in compliance and enforcement along various dimensions. One of the important sources of flexibility is the ability of manufacturers to engage in averaging, banking and trading of in-use emission credits to reconcile the in-use test results. This latter flexibility provides an alternative to mandatory recall of nonconforming engines. Recall is available, however, as a backstop.
- *Maintaining and encouraging participation.* No specific measures are included to maintain or encourage the participation of manufacturers in the ABT programme.

Evaluation of averaging, banking and trading (ABT) programmes

It is premature to evaluate the ABT programmes because they are just now being implemented. The programmes appear to be promising means of providing manufacturers with the flexibility to reduce the overall costs of meeting significant emission reduction targets.

Ozone transport commission (OTC) NO_x budget programme

In 1994, a group of north-eastern States – participating as members of the Ozone Transport Commission (OTC) – committed themselves to achieving region-wide NO_x emission reduction targets by 1999 and 2003 through an emissions trading programme. The NO_x Budget Programme is a “cap and trade” programme that allows large generators of NO_x emissions to trade allowances to meet the emission targets in a cost-effective manner. Since the multi-state committees that negotiated the agreement have no statutory authority, a Model Rule was developed to serve as a template for state laws that must be enacted for each state to participate. A number of states have since proposed and passed rules based upon the Model Rule.

Elements of the NO_x budget programme

Table 11 summarises the main elements of the NO_x Budget Programme.

• Threshold issues in the NO_x budget programme

- *Emission goals.* The participating states have committed to achieving a 75 per cent reduction from 1990 levels in NO_x emissions (55 per cent in northern areas) by the year 2003. The target will be achieved in two stages. In 1999, NO_x emissions are to be reduced 65 per cent from baseline (1990 levels) for inner zones, and 55 per cent for outer zones. In 2003, emissions are then to be reduced 75 per cent from baseline for inner and outer zones, and 55 per cent for the northern zones (Maine, New Hampshire, Vermont, and northern New York). Inner zones are those in non-attainment with Clean Air Act ambient standards and classified as “moderate”, “serious”, or “severe.” The more stringent reductions for these regions are an attempt to bring them into compliance by an earlier date.
- *Geographic area.* The participating states are the most north-easterly States, including Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Vermont, the northern counties of Virginia and the District of Columbia.
- *Commodity traded.* Allowances for NO_x emissions.

• Design issues in the NO_x budget programme

- *Initial allocation.* State allocations are based on 1990 emission rates, with allowances for quality assurance and exceptional circumstances, and a 10 000 ton reserve. Allocation to individual boiler units will be determined by the individual States.

Table 11. Elements of the NO_x Budget Programme

Threshold Issues	
<i>Emission Goals:</i>	• Reduce NO_x emissions by 75 per cent below 1990 levels (in most regions).
<i>Geographic Area:</i>	• North-eastern US (all or part of 11 states and District of Columbia).
<i>Commodity Traded:</i>	• NO_x emissions from sources.
Design Issues	
<i>Initial Allocation:</i>	• State allocations based on 1990 emission rates. • Boiler unit allocations to be determined by individual states.
<i>Geographic/Temporal:</i>	• Allowances may be banked, although the use of banked emissions is modified when the total quantity of banked allowances is high. • No geographic differentiation.
<i>Parties Trading:</i>	• Firms, principally electric utilities and manufacturers with large boilers.
<i>Institutions:</i>	• Under discussion.
Implementation Issues	
<i>Certification:</i>	• Trades approved by programme administrator after confirmation of adequate permit supply.
<i>Monitoring/Reporting:</i>	• Continuous Emissions Monitoring (CEM) for units already facing such requirements for the SO₂ trading programme. • Less costly alternatives allowed for other units. • NO_x Allowance Tracking System (NATS) maintains accounts of allowance activity. • Annual self-reporting of emissions.
<i>Compliance/Enforcement:</i>	• Compliance period from May through September. • True-up period running until the end of the year. • Non-compliance penalised through fines and a loss of three allowances for every excess emission in the following year.
<i>Maintaining/Encouraging Participation:</i>	• Provisions exist for voluntary opt-in of facilities not required to enter the programme.

– *Geographic/temporal.* There are no geographic restrictions on trades, although some regions are subject to more stringent requirements than other regions, and hence permit allocations will differ across the regions. Allowances may be banked, although there may be restrictions in their use depending upon the total quantity of banked allowances. A formula was developed to determine the fraction of banked allowances that can be used at various usage rates (*i.e.* allowances per emitted ton). At low levels of banked emissions, firms can use a 1:1 ratio, *i.e.* one banked allowance to cover one ton of emissions. After a certain level of banked emissions, banked allowances can only be used at a 2:1 ratio (*i.e.* two banked allowances to cover one ton of emissions).

– *Parties trading.* Trading will be between large, stationary generators such as electric utilities and manufacturers with large boilers. Provisions exist for other facilities to voluntarily opt-in. The decision of whether to allow trades with facilities outside the region is still being examined.

– *Institutions for trading.* The development of trading institutions, such as auctions or bulletin boards, and the rules covering trade by brokers are still under development. At present, there are no explicit plans for OTC or any of the individual states to develop such institutions.

• Implementation issues in the NO_x budget programme

– *Certification.* Trades must be approved by the programme administrator after confirmation that the selling party has adequate permits. The programme administrator maintains a NO_x Allowance Tracking System (NATS), which records the activity of all accounts.

– *Monitoring and reporting.* Continuous Emissions Monitoring (CEM) is required for units which already face similar monitoring requirements for the SO₂ trading programme. Less costly alternatives are allowed for other units, such as peaking units, non-utility generators, and small generators. The compliance period runs from May through September, with a true-up period running until the end of the year allowing facilities to purchase additional allowances if necessary. Units self-report their annual emissions to the administration based upon approved monitoring systems.

- *Compliance and enforcement.* Compliance is determined by comparing estimates of annual emissions based upon self-reported data against NATS accounts for each emitter. Non-compliance will be penalised through legal action (fines) and a loss of three allowances for every excess allowance in the following year.
- *Maintaining and encouraging participation.* Facilities that are not required to enter the programme may voluntarily opt-in if they accept the lower of 1990 or current emissions; have no built-in surplus; agree not to exit the programme (unless shutting down); and do not receive allowances for easing or curtailing operations.

Evaluation of the NO_x budget programme

The NO_x Budget programme is modelled closely along the lines of the EPA's SO₂ Trading programme. There are, however, some substantial differences. The banking provisions, for example, are more restrictive under the NO_x programme than the SO₂ programme.

The programme has yet to be implemented, so it is not possible to assess its actual success. One estimate, however, indicates that a regional NO_x cap and trade programme could lead to a 30 per cent savings (or about \$80 million in annualised cost savings), relative to a traditional emission standards programme (NESCAUM/MARAMA, 1996).

Lessons learned from US experience with marketable permits

Experience with marketable permits programmes for air emissions in the US shows varying results, ranging from modest gains to important successes. EPA's long-standing, credit-based ET programme has provided important flexibility in some cases, primarily for new sources in non-attainment areas, but for the most part has served as a relatively minor adjunct to traditional emission standards. In contrast, the lead-in-gasoline trading and banking programmes played important roles both in reducing costs and in achieving environmental goals more rapidly. The CFC tradable production/import allowance system appears vastly superior to a command-and-control alternative that would have established use-specific regulations or bans. It appears, however, that the tax on CFCs imposed by Congress has made the tradable cap largely irrelevant.

The SO₂ allowance programme and RECLAIM also appear to be working reasonably well. Both programmes have met annual environmental targets and appear to have robust permit markets that generate substantial cost savings. Although it is not possible to evaluate the performance of the recently-promulgated ABT programmes for mobile sources and NO_x budget programme for the Northeast, both programmes appear to be promising applications of previously-successful programmes.

This section suggests some lessons that might be drawn from the US experience with tradable permits programmes. We organise these lessons based upon the three types of decisions outlined above.

Threshold decisions

The following are lessons suggested by the threshold decisions made in existing programmes.

Tradable permit programmes appear to be more successful when they are developed to implement new initiatives

Tradable permits are widely regarded as a means of achieving given emission targets in a more cost-effective manner. In theory, therefore, it should be relatively unimportant whether the emission targets are already established or if they are set at the same time that the tradable permit programme is established. In practice, however, it appears to be easier to establish a successful tradable permits programme at the time that targets are set rather than as an adjunct to existing requirements.

There are several potential explanations for success being linked to new initiatives.¹⁶

- *Environmental support.* Environmental groups may be more disposed to support regulatory flexibility when it is tied to major emission reductions. For example, the SO₂ allowance programme obtained important support from the Environmental Defense Fund (EDF).

- *Agency support.* Regulatory agencies may provide flexibility in order to obtain emission reductions that may not be technically feasible for all regulated parties. The EPA supported the averaging programme for lead in part because of evidence that smaller refiners would not be able to meet the low levels called for in the phase-down.
- *Lack of constituencies for the status quo.* New programmes do not have the difficulties of modifying expectations that are set up for existing programmes among various constituencies, including the private sector, environmental groups, and government.

Tradable permit programmes appear to be feasible for different geographic scales, although the experience with inter-state systems is not yet available

Successful tradable permit programmes have been established for sources throughout the US (e.g. SO₂ trading programme, lead-in-gasoline), as well as for sources in individual geographic areas (e.g. RECLAIM). In both cases, however, trades take place within a single administrative region. No existing programme thus far has involved trades across different states or different air quality control regions within a state. The NO_x Budget Programme represents the first attempt at such a programme and should provide important information on the additional issues involved in inter-state programmes.

Cap-and-trade programmes tend to provide easier means of defining the commodity to be traded than credit-based programmes

One of the key factors in determining the success of trading programmes appears to be the clarity with which the commodity is defined. Clear definitions minimise the obstacles to trading and increase the likelihood that potential cost savings will be achieved. If the commodity is not clear, time and resources are wasted trying to determine what is available for sale or purchase.

The ET programmes illustrate the problems that can arise when the commodity is not clear. Because the baseline for determining emission reduction credits was unclear, proposed trades were often controversial and ultimately discouraged. In contrast, the emission allowances established under the SO₂ trading programme and RECLAIM were clear, facilitating the smooth operation of trading.

Averaging programmes can provide clear definitions of the commodity

In some cases, clear allocations can be developed without setting an overall cap. Both the lead-in-gasoline and the ABT programmes provide examples. These averaging programmes in effect allocate emissions in proportion to production, either of leaded gasoline or engines. The traded commodity is clearly defined, making it easy to engage in trades.

Design decisions

The following are lessons from the decisions involved in the design of tradable permits programmes.

Determining initial allocations can be contentious, but are feasible if most parties gain from cost savings

Cap-and-trade programmes require the regulatory agency to determine the distribution of initial allowances. This process is invariably highly contentious because valuable property rights are being distributed. The process of determining initial allowances was complicated by many “equity” considerations in both the SO₂ trading programme and RECLAIM. Nevertheless, the process of determining initial allocations appears to be feasible as long as most parties gain, relative to command-and-control standards. Most parties will gain, either as buyers and sellers, if the initial allocations are “grandfathered” to existing sources, rather than auctioned.

Note that allocation is much less of an issue under an averaging programme. There are, however, distributional issues that can arise in terms of how products are subdivided for averaging purposes. In the case of lead-in-gasoline, basing the programme only on leaded gasoline (rather than all gasoline) provided an advantage to refineries that produced higher than average shares of leaded gasoline. The restrictions on averaging across outboard motors and personal watercraft represents another example of

how distributional issues can arise. Restricting averaging may work to the disadvantage of firms that do not produce products in both markets, although trading across firms can reduce or eliminate the disadvantage.

Large gains to particular parties under a cap-and-trade programme may create pressures to tax “windfall gains” away

The fixed nature of the cap means that the value of the allowances is included in the costs of production. This effect occurs regardless of whether the allowances are allocated or auctioned – as long as the allocation formula does not vary with *current* production – because there is an opportunity cost to additional production. (Producing another unit of the good requires allowances that must either be purchased or obtained internally; in the latter case, the firm foregoes the money from allowances that otherwise could be sold.)

These effects can lead to substantial price effects – and “windfall profits” – if the programme covers a large portion of the relevant product market. For RECLAIM, the price effects are likely to be modest because many of the participants operate in national or international markets, so that the opportunity cost of the allowances does not lead to price increases or windfalls to RTC recipients. The CFC programme represents an example of a programme with substantial windfall profits that accrued to a small number of CFC producers. The fact that Congress taxed the windfall profits away suggests that large perceived windfalls are not likely to be allowed to continue.

Allowing inter-temporal trading (banking) can provide important flexibility

Banking provides important flexibility for sources to undertake early reductions to accumulate allowances that can be used to ease compliance in the future. This flexibility is particularly important where reductions are phased in over time. For example, both the SO₂ allowance programme and the lead-in-gasoline programmes involved substantial phase-downs in allowable emissions. In the lead phase-down, participating sources undertook considerably more control than would have occurred in the early stages of the programme, and the high level of banking in the SO₂ programme partially reflects a similar strategy. Banking during phase-downs is likely to generate environmental benefits, because it leads to greater emission reductions when overall emissions are relatively high – and thus, potentially more damaging at the margin, based upon decreasing marginal damages – and to smaller emission reductions when overall emissions are lower.

Banking can be important even when there is not a large phase-down. The need for allowances is likely to differ significantly from year to year, due to economic cycles and swings within given sectors. Banking allows sources to accommodate these swings without leading to large changes in allowance prices.

Private institutions typically develop to facilitate trades and provide market information, although government institutions (e.g. auctions) can be useful initially

Brokers are important elements of the trading programme for both the SO₂ allowance programme and RECLAIM. These brokers lower the overall transaction costs for trades, increasing the volume of trades and overall cost savings. In the SO₂ programme, for example, brokers and other private market transactions account for the bulk of allowance sales and purchases. In contrast, relatively few transactions are accounted for by the auction mechanism set up initially to provide market information and liquidity. Such auctions can, however, be useful in the early stages of a programme to provide information on price.

Implementation decisions

The following are lessons related to the implementation of tradable permit programmes.

Certification of emission reduction credits can be burdensome and thus limit the potential for trading

The certification required for emission reduction credits can substantially limit the programme. As emphasised above, the 1986 policy statement for the ET programmes imposed stringent requirements that helped ensure that “paper credits” would not be granted. But these regulations imposed so many requirements that trading generally was too expensive or uncertain to undertake.

Flexible monitoring requirements can increase participants beyond large sources

Emissions under a cap-and-trade programme must be monitored to ensure that the quantity of allowances purchased and sold is accurate. The SO₂ programme is limited to large stationary sources that have continuous emissions monitors to monitor mass emissions. In contrast, RECLAIM includes some relatively small emitters. These emitters are given the flexibility to use other means of determining mass emissions, including fuel meters and emission factors. These less expensive monitoring devices allow a programme to include more sources, and thus increase the overall cost savings from trading.

Compliance and enforcement do not appear to be major obstacles given the extensive mechanisms in place in the US

Emissions sources in the US are covered by a complex and well-established system of controls. These controls include compliance and enforcement mechanisms such as substantial fines for non-compliance. The existing trading programmes do not appear to have been burdened by difficulties in ensuring compliance among participants, most of which are large entities. Indeed, the trading programmes provide substantial flexibility that tends to avoid the major confrontations that can occur under command-and-control regulatory programmes.

NOTES

1. This report uses the term “tradable permit” to refer to programmes to allow regulated parties the right to trade emission limitations among themselves. This concept is referred to by other names, including emissions trading, marketable permits, emission allowances, and transferable pollution rights.
2. See Stavins (1998).
3. This paper draws on previous reports and papers including Harrison and Nichols (1990), Harrison and Nichols (1992), Harrison (1997), and Harrison (forthcoming). The author wishes to thank Dr. Albert Nichols for his contributions to our earlier joint work, Dr. Todd Schatzki for his contributions to this paper, the Electric Power Research Institute – and Dr. Thomas Wilson – for its support of these efforts, as well as the organisers of the OECD Workshop on Domestic Tradable Permit Systems for Environmental Management – notably M. Jean-Philippe Barde – and the participants in the OECD Workshop for providing an opportunity to discuss the issues with a perceptive group. Despite these contributions and support, the author alone is responsible for the views expressed in this paper and any errors or omissions it may contain.
4. Dales (1968) is generally considered the first complete treatment of the theoretical advantages of the tradable permits approach.
5. Other reasons also contribute to the small numbers of ET transactions. Few firms have found banking attractive, for example, in part because of fears that if they identified banked reductions, regulators would tighten standards to eliminate them. For a discussion of these and other issues related to EPA’s ET programme, see Dudek and Palmisano (1988).
6. EPA ended all trading and banking on January 1, 1988, because the new standard of 0.1 gplg was set to allow a minimum amount of lead to protect certain older engines against premature valve wear; EPA was concerned that with continued trading, refineries would be encouraged to reduce lead levels too far to provide that protection.
7. Many were clearly by mistake, since 20 per cent of the reported errors did not lead to commercial advantage (Kerr and Maré, 1997).
8. The figures for their sample may overstate the whole industry. While their survey covers 65 per cent of the industry capacity, the sample covers only 48 per cent of refineries and 27 per cent of companies. Thus, the remaining 35 per cent of capacity not sampled probably has smaller firms sizes which provide fewer opportunities for intra-firm trades.
9. To match provisions of the Montreal Protocol, the rule included consumption as well as production allowances, where consumption was defined as production plus imports, minus exports.
10. EPA recognised when it proposed the transferable allowance system that it was creating potentially large gains to manufacturers, because limiting the supply of allowances limited CFC production, which in turn would drive up the price of CFCs. The value of the allowances was equal to the difference between the price that buyers (typically manufacturers of products that used CFCs, such as auto air conditioners) were willing to pay for that quantity and the price at which producers would have been willing to supply that quantity. That value was captured by the firms to which the allowances were allocated.
11. For additional information on the acid rain trading programme, see Bohi and Burtraw (1997) and Ellerman *et al.* (1997).
12. For additional information on RECLAIM, see Harrison (forthcoming), Johnson and Pikelney (1996), South Coast Air Quality Management District (1994), and Regulatory Flexibility Group (1991).
13. This process – which was referred to as the mid-point or end-point “shave” – involved reducing each facility’s allocation by the percentage needed to achieve the overall AQMP target emissions.

14. In addition, several programmes have been announced but not yet developed into formal programmes. These include the open-market trading proposal put forth by EPA as an extension of the ET programmes and the NO_x trading programme to implement the recent SIP Call to reduce NO_x emissions in 22 states in the East and the District of Columbia.
15. The regulations are promulgated in 61 Federal Register 52088. The ABT programmes promulgated for other sources are similar in overall design, although specific elements differ.
16. See Stavins (1998).

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Chapter III

APPLICATION OF TRADABLE PERMITS TO LAND-USE MANAGEMENT

by

Vincent Renard

Centre National de la Recherche Scientifique, Paris, France

Introduction

In comparison with other fields where tradable permits apply, such as air or water, land has some quite distinct properties, not least because the many entangled legal instruments that govern it play a large part in determining its price. Property law varies widely from one country to another, particularly so in the case of development, law, the prime cost determinant when land is in demand.

For this reason, before going into the detail of actual examples of tradable permits, the relationship between property law, urban development regulations and tradable permits is first examined, with particular attention to the differences between legal systems of the type found in North America and those common in Western Europe.

Following this, a description is given of how the tradable permit system – which may still be deemed experimental since the practice is not yet widespread, even in the US – operates in three countries: the US, France, and New Zealand.

From consideration of these experimental schemes, a number of conclusions are then drawn concerning the value of the system, its effectiveness in reaching its goals, and the conditions required for it to operate satisfactorily.

Property law, urban development regulations, and tradable permits

Why tradable permits?

It is perhaps unnecessary to point out that the underlying reason for using tradable permits is the economic efficiency generally expected from market mechanisms, in contrast to the allotment mechanisms employed by administrative and planning authorities. Following the ground-breaking article by Ronald Coase (Coase, 1960) and subsequent theories of property law, it is generally accepted that the definition and transfer of property rights, because they allow a market in rights to operate, give results that are “better” (in the sense of Pareto optimality) than the alternative of being unable to transfer such rights.

It should be noted that Coase’s “theorem” rests on the basic premise that there are no (or negligible) transaction costs. However, the way property markets operate – and hence, the way markets in development rights operate – generally entails in high transaction costs and generates market imperfections (opaqueness, barriers to entry, oligopolies, monopsony situations, etc.) The underlying reasoning must therefore be applied with caution.

With regard to pollutant emissions, the introduction of tradable permits has the advantage both of allowing a measure of choice in ways to reduce pollution and of reducing the economic cost of complying with environmental constraints.

Their application to the implementation of urban development plans, to management of density in urban or peri-urban areas and possibly to areas of biological diversity not directly tradable (wetlands) still remains under-explored, particularly in Europe.

The only significant area of use has been development rights (or building rights) that can be traded independently of the supporting land.

The introduction of tradable rights systems creates both conceptual and theoretical problems, as well as practical difficulties in relation to implementation. In the case of the schemes considered in this paper, the technique would appear capable of reaching its goals only if it operates within a clearly-defined legal framework, and is closely tied to a rigorously applied land-use planning system.

In addition, there are similarities with contractual, fiscal and other techniques, akin to techniques to provide compensation for constraints. The Conservation Covenant Programme in the Australian State of Victoria is a good example. But such programmes have a unilateral component by the public authority, which is different from a market.

Recent developments: rights to pollute and land rights

Tradable rights have come to the fore in recent years, particularly in the general context of climate change, the greenhouse effect and air pollution – a contributing factor to the latter.

An important threshold was crossed in late 1997 with the adoption of the Kyoto Protocol, which envisaged trading in quotas or emission credits.

A primary conceptual difference should first be noted between a tradable permit attached to land and a tradable emission quota.

In the case of pollution, the object of the trade is an entitlement to emit an ongoing level of pollution, measured for example in tonnes of nitrogen dioxide discharged into the air per year. What is involved is a continuing process, so that the relevant quotas themselves may go on being bought or sold *ad infinitum*.

The idea behind tradable land rights is quite different, since the right concerned is sold outright or for a very long period. Admittedly, it is only saleable in part, or may be bought back at a later date, but the purpose of the transaction is in no way linked to an ongoing process. This will obviously have a major impact on the way the instrument is employed with respect to allotting rights, and the conditions for buying them back.

“Tradable permits” or “tradable rights”

To take the term “tradable permits” first, this is a concept widely applied in regulating pollutant emissions, but is not readily transposable to land rights (or only partially so).

Any transposition there might be between pollution and development, or more generally between pollution and change-of-use (drainage of wetlands to plant maize, building a major road through woodland, urban development). Such a change-of-use, unlike pollutant emissions, is a permanent right accorded by the regulations. It need not necessarily be used, but if it is used, it can be used once only, and in a manner that is often irreversible.

Instead of “tradable permits”, it would therefore be better to use the term “tradable rights” or “transferable rights”. In the area of development or construction rights, the latter is the term used in the US, where the “transfer of development rights” is usually referred to.

Property law and trade in its components

The concept therefore has to do with property law as applied to geographical space, and it reveals a major difference between the legal systems originating in Roman law, based on the indivisibility and absolute nature of land ownership, and the diametrically opposite view of the main variants of Anglo-Saxon law (in particular, North American law), which considers land ownership as a “bundle” of rights, some components of which can be treated separately, such as development rights, air rights, or mineral rights.

Naturally, in systems based on Roman law, legal concepts have also been introduced which allow one or more of these rights to be dealt with separately, and even to be sold individually. Examples that come to mind are hunting rights, rights of way and mineral rights. However, these are distortions of the system, rather than ones which conform to its underlying principles.

This difference in legal structures and in case law go a long way to explaining the difficulty countries of Western Europe or Japan have in coming to grips with concepts or techniques that are widespread in North America. It is particularly evident in the long-standing US practice of tradable environmental rights (easements) – a concept which appears nowhere in French law for example, with its need to determine dominant tenement and servient tenement in order to establish such a relation.

Compensation of constraints

Central to the creation of a market in tradable rights is the issue of the financial and fiscal implications of urban development regulations.

To recap: in urban and peri-urban areas, the value of a piece of land lies in the rights attached to it, which are conditioned by urban development regulations and by environmental regulations. Whenever regulations are introduced by public authorities, the price of land is strongly affected. Since any change to the regulations moves the price of land (up or down), it may be asked what corrective measures ought to be taken by the public authorities.

As far as urban and peri-urban areas are concerned (the problem being most acute in the latter), the response has differed from country to country.

Roughly speaking, most countries in Western Europe have adopted the principle that constraints on urban development are not liable to compensation. As expressed in the French Urban Development Code for example, this principle applies “to any constraint affecting the road system or prompted by health, aesthetic or any other considerations and concerned with such matters as land use, heights of buildings... or prohibition of development in given zones (Article 160-5 of the Urban Development Code). A constraint on the right to make use of a given piece of land is not considered grounds for compensation unless it infringes a vested right (for example withdrawal of a building permit already granted) or a change in the previous status of the site resulting in direct, material and indisputable damage to property. This comes close to the concept of “taking”.

Strict application of this principle, which makes landowners subject to unequal treatment, has naturally met with considerable opposition and has led to the generation of *de facto* and *de jure* loopholes.

In France for example, the introduction in 1976 of procedures for the transfer of development rights comes under this heading and was strongly attacked as a breach of the principle of the ineligibility of constraints for compensation. In essence, it was the pliability of urban development regulations, for example with respect to land-use planning, that generated the inequalities stemming from this principle. Although the law is pliable to some extent, this process is far removed from the basic assumptions governing a perfect market.

The underlying principle is different in the US. Although the constitutional legality of zoning has been well established since the 1926 Euclid decision (village of Euclid v. Ambler Realty Co., Federal Supreme Court), the dividing line between admissible constraint under police power which is ineligible for compensation and excessive constraint eligible for compensation (taking) or expropriation (eminent domain) is constantly shifting. The term “taking” which may be read as “seizure” is a contemporary borrowing from the Fifth Amendment to the US Constitution (“nor shall private property be taken for public use without compensation”).

Although this provision, which has been in force since 1789, is closely related to the French Declaration of Human Rights, it is being applied in a quite different way in contemporary case law, and its broad interpretation by the courts has led it to be considered in practice as the basis for litigation to test the legality of urban development regulations.

As illustrated by a number of recent decisions and the rise of the Movement of Private Property Rights, attention is drawn to recent developments which have tended to restrict the implementation of regulations and constraints (in particular the Clean Air Act and the Endangered Species Act) by systematically playing the “compensation card”.

The way case law in the matter has developed over time matches the gradual change that has taken place in property rights law, in which a distinction is made between what is private property in the strict sense of the term (and may thus be put on the market) and what is common property.

The keenness of the debate in the US, both from the purely legal standpoint and in the political arena, shows how crucial the initial definition of the “bundle of rights” is.

Added and lost value

Postulating the existence of tradable rights assumes that there is something to trade – in other words, that one of the parties is ready to relinquish an attribute of his property (the right to build, for example) to another owner, and that the exchange is “profitable” for both parties.

Whatever the circumstances, no market will operate unless the exercise is worthwhile – in other words, unless there is a demand for the “rights”. This raises the issue of initial allotment of rights. Here too, there are two concepts to be distinguished, depending on the methods used to value land and real estate, which are themselves based on the way property is conceived.

Under one concept, which is fairly widespread in northern Europe, ownership of land does not include a right to the value usually added by urban development or, in practical terms, by the installation of infrastructure, the cost of which is generally shared between the developers and the community. This is, for example, what happens in Sweden or the Netherlands (Needham *et al.*, 1995). By means of different mechanisms (long-term land reserves for Swedish towns, quasi-monopoly of towns in the development process in the Netherlands), it results in most of the value added by urban development being collected by the community.

The other concept, which prevails for example in several countries in southern Europe (there is no “pure” case, so it is difficult to be more precise here) consists in allowing the original landowner to keep the capital gain, subject to tax corrections – for example, the value added tax. In such systems, the introduction of urban development regulations or their amendment will generally be perceived as a constraint on previously-held rights, the assumption being that ownership was at the outset unconditional and, in particular, included the development rights.

This is the context in which it is possible to conceive of trading in a “right” that is assumed to be in existence, but the actual use of which has not been authorised. This point is essential to understanding the crucial importance of the original allotment of rights, and the conventional nature of that allotment.

Unilateral regulations and tradable rights

One final concept needs to be clarified in relation to the central focus of this paper, the idea of a market in tradable rights. Many regulations applicable to geographical space are subject to various trade-offs between the public authorities responsible for permits and applicants. In an urban development project (such as a planned-unit development), this happens when the municipality and the developer negotiate what construction is permissible, and what public amenities the developer will undertake to install.

It may also happen in the case of contractual land taxation systems in the US, where the contract negotiated covers (to some degree) both a commitment by the owner (for example, to continue farming for ten years) and the basis on which his property tax is to be calculated.

These techniques – essentially fiscal in nature – whose aims are close to make zoning constraints more acceptable to owners by offering compensation, use unilateral procedures such as assignment to a lower tax bracket or various methods of granting exemption from charges or according financial benefits, in return for a commitment from the landowner (and/or user) to comply with given constraints, to maintain, to engage in nature conservancy, to continue farming, to engage in organic farming, etc. The literature dealing with such tax incentives is fairly extensive (see Cohen de Lara and Dron, 1997). This situation

differs, however, from the matters dealt with in this paper, in that no market mechanisms are introduced. A unilateral decision is being made to offer a contractual tax technique. From this point of view, such contracts are outside the scope of this paper.

Other land rights markets exist. Mention may be made for example of the “market” in hunting rights, which is still extant although of minor importance (see Charlez, in Falque and Massenet, 1997).

However, the heart of the problem and the major experiments in rights trading are concerned with the market in rights to “intensify” land use, which are essentially rights to engage in urban development and to build. These are the rights on which this paper will focus.

Transfer of development rights: origins and applications

This section examines systems employing the tradable rights technique in operation regarding development rights in the US, France, and New Zealand.

The legal, economic and institutional background, however, varies widely in each case, as do their aims and sphere of application. The programmes described below were not chosen in order to achieve a statistically-representative group – a difficult task – but from a desire to present a judicious mix of examples that are significant because of their scale, their duration, their management methods, or their results.

Transfer of development rights in the US

Origins

The connection between the case law on “taking”, which became increasingly acrimonious in the 1960s and 1970s, and the advent of the technique of transferring development rights, were both discussed earlier. In the face of stringent regulatory measures that involved the public authority in compensation procedures, the technique helped to make the regulations acceptable while conserving public funds, since the value it added compensated for any value that was lost. This was the thinking that led to the emergence of transferable development rights in the early years of the 20th century.

The City of New York was a pioneer in forging this technique, first by successive amendments to its 1961 Zoning Resolution, which made it possible to exchange densities between adjoining lots. The exchange of development rights among neighbours, or among a given block of houses, has been a widespread practice in developed countries, and is generally compatible with the provisions of private law. A contract with the owner of a neighbouring piece of land provides a guarantee, for an agreed price, that it will be kept in good condition.

This form of exchange is generally limited to transactions affecting a small area and is only remotely akin to the concept of establishing a market in rights. The main idea is that of a right which is detachable from a given piece of land, and is usable in another area. A ground-breaking example in the US is given by the New York Central Station, the case *Grand Central Terminal v. City of New York* having generated much written comment (Babcock and Siemon, 1985).

In this case, the City of New York was forced to provide compensation for the hardship caused by refusal of a building permit (the proposal was to erect a very tall 57-storey building above the station, which was in process of being designated a listed building) by granting the railroad company (Penn Central) development rights to a piece of land elsewhere. This was the first example of an out-of-zone transfer of rights, but it could not really be considered as the launch of a market in rights.

It was the 1970s that saw the real emergence of tradable development rights as a technique, one mainly applied to large areas of natural scenery. However, the number of noteworthy examples of this is actually quite small. Although much has been written on the subject, its actual impact has been limited; throughout the US, a little over 20 000 hectares have been protected by means of transfer of development rights. Table 1 shows the main features of four of the most noteworthy examples.

Table 1. Programme description

	Pinelands	Montgomery County	Lake Tahoe Basin	Santa Monica Mountains
Population	450 000	781 022	51 775	11 336 (Malibu)
Year Programme initiated	1980	1980	1987	1979
Goals	Cohansey Aquifer and Forest Conservation	Agricultural Land Preservation	Watershed Preservation and Recreation	Erosion and Water Quality Control, Avoid Public Service Strains
zoning or Permit-ased	Zoning	Zoning	Permit	Permit
andatory or Voluntary	Mandatory	Mandatory	Voluntary	Voluntary
ual or Single Zone	Dual	Dual	Single	Single
umber of TDR/TDC ransactions	1 424 Severed Rights	4 300 Transfers	372 Coverage Transfers	493.7 Transfers
and Area Preserved	12 538 Acres Preserved	Approx. 34 000 Acres Preserved	Coverage Transferred: CA: 66 686 sq.ft. NEV: 9 361 sq.ft.	864 Lot Restricted
urrent TDR/TDC alue	\$3 500-4 500	\$10 000-12 000	CA: \$5-8/sq.ft. NEV: \$8.5-15/sq.ft.	\$17 000-21 000
rocessing Period	30-45 Days	4 Days	2-11 Weeks	2-4 Weeks

Pinelands National Preserve (New Jersey)

One of the most important examples of application of the tradable development rights method (transfer of development rights) has been progressively implemented over the past twenty years in this area covering almost 4 000 km² in southern New Jersey, one of the most populous regions of the US, situated between Philadelphia, New York and Atlantic City. Lying close to a very large groundwater basin, it is also extensively wooded, and home to many animal species.

It was with the express intention of protecting the environment that the US Congress decided in 1978 to establish the Pinelands National Preserve. Congress requested the Governor of the State of New Jersey to establish a special planning commission and to impose a moratorium on development in the zone, while awaiting the adoption of a development law. The plan ultimately adopted in this context – the Comprehensive Management Plan – was approved in November 1980.

The exceptional nature of the conservancy policy applied in this remarkable area is worth noting. Preparation of the plan came first, the transfer of development rights was merely an instrument designed to apply this plan. It involved very stringent zoning, in which areas of land (and bodies of water) were divided into 10 categories, some zones being designated as closed to development but able to transmit development rights; others being classified as receiver zones, where development was permissible subject to the purchase of rights from landowners in transmitter zones.

The mechanism works as follows: the allotment accorded each piece of land is determined on the basis of several criteria (location, type of terrain, past and present use, existing buildings if any, and method of land acquisition). Agricultural land and areas designated for conservation because of their outstanding natural beauty, the interest of their flora and fauna and the need to protect the aquifer (an extremely large one, over 60 thousand million cubic metres) are classified as transmitter zones, which are allotted development credits on scale which is subject to change. Four credits are required for a permit to build a house (*i.e.* to receive one development credit or DC).

In the core conservancy area (18 000 ha), where development is very restricted, one development credit is allotted for every 16 ha, one for every 8 ha in an agricultural zone, and one for every 32 ha in a wetland zone. These basic parameters are adjusted to take account of various factors.

The receiver zones, which generally lie on the periphery, cover an area of 32 000 ha, and have a development potential of 126 000 constructions, or approximately twice that available from the transmitter zones.

The initial allotment process was clearly a key issue and was carried out by the New Jersey Pinelands Commission. A threshold price of US\$10 000 per credit was originally set, but the first exchanges rapidly led to a much higher market price. The selling price of a credit in 1996 was about US\$20 000; the credits received by landowners in the farming zone were thus worth approximately US\$1/m². Once purged of its development credits, the “transmitter land” was closed to development in perpetuity.

Since 1980, the system has been managed by two bodies: the Pinelands Development Credit Bank (or Pinelands Bank) for the State; and (for the main county involved) the Burlington Exchange. The latter buys in credits from Burlington County only, but may resell them anywhere in the zone.

During the period 1981-1996, only 6 000 ha of the 65 000 ha in line for closure for development were in fact so closed, and 4 000 of the planned 26 000 houses have been built.

Quantitatively-speaking, the original aims of the programme have by no means been met in practice. That said, the programme is a long-term one and the finite upper limit placed on the number of development credits justifies the length of the process.

This is the most extensive and most developed example to date. The tradable rights programme has played its part there, albeit a minor one, not least as an educational tool to promote acceptance of the remarkable planning effort made by the Pinelands Commission.

It has also been accompanied by a heavy emphasis on public ownership of biologically-rich zones. So far, these make up 170 000 ha, and will ultimately cover 200 000 ha.

Montgomery County (Maryland)

This programme, like the Pinelands Preserve, is a noteworthy and interesting example because of its size and the length of time it has been in operation.

Located in north-west Maryland (*i.e.* close to both Washington DC and Virginia), Montgomery County faces the usual problems of protecting areas of outstanding natural beauty from strong urban development pressures.

Despite relatively rigorous urban planning regulations in place since the early 1970s, in particular, the requirement for a lot to be at least 2 ha for building to be permitted, nearly 20 per cent of the County's agricultural land was swallowed up between 1970 and 1980. This led the County Council in 1980 to draw up a programme for transfer of development rights as part of the machinery for implementing the Functional Master Plan for the Preservation of Agricultural and Rural Open Space. The special Commission responsible for execution of the programme (the Maryland National Parks and Planning Commission) decided to establish a rights-transmitting agricultural zone occupying 40 000 ha of prime agricultural land, where the permissible housing density was reduced from one dwelling per 2 ha to one dwelling per 10 ha – the difference between the two constituting the tradable rights.

Only a few “receiver” zones had been designated when the project began, but they were not officially recognised as such until later; preference was given to zones with infrastructure already in place, in order to prevent allotment of development rights from being too influenced by existing expectations. According to the players involved (see Canavan, 1993), the designation of the “receiver” zones was the crucial step, together with the problem of allotting a development right that gave sufficient incentive to set the scheme in motion.

The procedure was simpler than in the Pinelands case, since only residential developments were authorised. The ratio for a “transmitter” zone is one development right per 2 ha (no splitting being allowed), regardless of the nature of the terrain. Final severance of development was entered on the title deeds by the County's administrative authorities. The County has established a bank (the Development

Rights Fund) that buys and sells development rights. This may provide security for loans backed by tradable development rights. From a starting price of US\$3 000, the cost of a development right subsequently rose to between US\$10 000 and US\$20 000. The value of the right in the transmitter zone is thus US\$0.5/m².

The system has worked well. By the end of 1997, 4 300 of the 4 700 rights had been transferred, giving permanent protection to almost 15 000 ha. The success of the scheme became apparent when the conversion of agricultural land slowed down significantly. Although the conversion rate had been 1 000 ha a year before 1980, that was approximately the total area converted between 1980 and 1991.

A high pressure information campaign and the simplicity of the procedures involved were deciding factors in Montgomery County's success.

In addition, the procedure clearly made the development restrictions introduced by the plan more easily accepted. The price of the transferable development right (equivalent to US\$0.5/m² in the farming zone) was not a decisive factor in this process. The price has in any event been sluggish for the past two years. However, owing in particular to the active and participatory attitude of the Park Commission, it has helped to make rigorous zoning more acceptable.

Lake Tahoe (California-Nevada)

This is undoubtedly the best known example, not only because of its land management method, but also because the shores of this natural lake of 80 000 ha, of crystalline water are outstandingly beautiful and naturally much sought after, on the line between the States of Nevada and California.

The zone as a whole is managed by a special agency, the Tahoe Regional Planning Agency (TRPA), which is responsible for planning and issuing development permits. Under conflicting pressures from owners anxious to increase the value of their land and residents hoping to limit development, and faced with water resource constraints, the TRPA began (in the mid-1980s) to make use of a system of tradable rights with very special features linked to the water resource.

The tradable right here is the "coverage coefficient" – that is, the portion of the surface area of a lot that may be built on. Calculation of the transferable rights allotted to each lot is determined on the basis of a calculation of ecological sensitivity, producing a set of 8 factors connected to type of soil, slope, hydrological features, exposure, etc. This sophisticated lot evaluation system (IPES, Individual Parcel Evaluation System) was established after lengthy public debate. The evaluation process is carried out by a team of three (a soil specialist, a hydrologist and a town planning expert) and ends by assigning a factor to each lot from which the coverage coefficient can be derived. Since the introduction of the evaluation system in 1989, over 12 000 lots have already been assessed in this way.

From coverage transfers, it can be seen that prices are higher in Nevada (US\$85 to 150/m²) than in California (US\$50 to 80/m²). Although the system is quite complex, it is now reasonably well-developed, with evaluation taking about six to eight weeks, and the transfer itself taking two to three weeks.

The market in transferable coverage is managed in California by a special body, the California Tahoe Conservancy. The Land Coverage Bank provides a cash reserve for purchase and restoration of degraded sites, as well as for purchase and sale of land coverage rights. By late 1995, almost 400 transfers had been made.

The Lake Tahoe project faces a special problem. Since the limiting factor is the water resource, the total rights now allotted represent more or less the ultimate limit to lakeside development. It is therefore conceivable that the market will come under pressure as that limit approaches. Should prices then be allowed to rise to levels unforeseen today? (Prices are already at US\$200 /m² on the Nevada side).

Transfer of development rights in France

The origins of the 1976 Act

As indicated earlier, French urban development law – as in most European legal systems – is based on the principle of no compensation for urban development constraints or, in the terminology used in North America, "police power" takes over from "eminent domain". This feature clearly creates inequities for landowners whose land is differently affected by urban development schemes.

In the French case, such inequities posed a particularly thorny problem in the early 1970s with the launch of urban planning by the Land Use Act (Loi d'Orientation Foncière) of 31 December 1967, which for the first time introduced zoning as a general principle (Plan d'Occupation des Sols), from which there was no exemption. After a number of abortive attempts and lengthy controversy, the Urban Development Reform Act (loi sur la réforme de l'urbanisme) of 31 December 1976 finally made it possible, in some zones, for development rights to be transferred from one subzone (transmitter) to another (receiver). This principle is today embodied in Article L 123-2 of the Urban Development Code:

“In zones to be conserved because of the quality of the landscape (...) land-use plans may determine the conditions under which the development potential determined by the land-use coefficient set for the zone as a whole may, subject to authorisation by the administrative authorities, be transferred in order to promote concentration of development on other lots in one or more sectors of the same zone.”

This wording is somewhat ambiguous in its reference to zones “to be conserved because of the quality of the landscape”. Is the intent to exclude productive agricultural areas as well as zones available for development? Can such a distinction be made? There is no case law as yet on this point.

The provision generated a great deal of impassioned debate before, during and after its adoption. Considerable criticism has been levelled on three points: the tie to zoning, incompatibility with the principle of no compensation for constraints, and distributive justice.

With regard to the first point, the difficulty resides in application of the term “quality” to landscape which seems to equate it with a conservancy area, in which all development is prohibited. But how can productive agricultural areas be excluded since they are often areas in which the landscape is of outstanding quality?

The pressures that could be brought to bear on owners of agricultural land to apply for transferable development rights can be imagined. This ambiguity in the relationship to zoning has had an adverse impact on the first trials of the technique in France, for example at Lourmarin (Vaucluse).

On the second point, what is being called into question is the very principle of transfer of a development right, in that it implies compensation for constraints, since it applies solely to land to be “conserved because of the quality of the landscape”.

From the point of view of distributive justice, it may be said that the equity of the procedure is basically conditioned by the original distribution of land holdings. Where land has been distributed in a fairly equitable way among the inhabitants, the procedure would be neutral in terms of distributive justice.

However, the actual situation is generally very far from this. The technique then represents a transfer from the community as a whole, the legitimate beneficiaries of any value added by urban development, to the sub-group represented by the landowners in the zone concerned.

Some concrete examples

Although the law allowing transfer of the land-use coefficient has now been in place for 23 years, examples of the practice remain few, disparate and difficult to summarise. Details are given below of five of the earliest examples, located in the French Alps and in the south of France.

Taninges – the Praz de Lys plateau

In this relatively clear-cut case, the task was to manage and organise the development of a gently undulating plateau at an altitude of between 1 600 and 2 000 m, where the expansion of winter sports, in particular cross-country skiing, was to lead to the building of a road to make the plateau accessible to motor traffic.

To prevent haphazard development, a development plan was drawn up before the road was built, which divided the plateau as a whole (zone to be conserved because of the quality of the landscape) into development rights “transmitter” zones (in general accorded a transfer right of 0.035) and “receiver” zones where development was permissible, with development rights varying between 0.10 and 0.30,

whose owners were permitted to build provided they had acquired additional rights from owners in the transmitter zones. It should be noted that the commune, which owned land in a transmitter zone, was granted a COS (floor-area ratio) of 0.10.

To give an example, the owner of a 7 000 m² piece of land in a receiver zone of density 0.20 ha, in terms of his entitlement of 0.035, a surface area available for development of $7\,000 \times 0.035$ or 245 “points”. In order to build at the authorised density of 0.20, the developer needs $7\,000 \times 0.2$ or 1 400 points, and must thus buy 1 155 points, which must come from $1\,155/0.035$ or 33 000 m² of land that will thereby become permanently closed to development.

The system has worked fairly well. It should be noted that the commune is heavily involved in its operation. Indeed, it initiated the movement itself by making the initial transfer between two lots belonging to the commune, one in the transmitter zone, and the other in the receiver zone. The private market then took over, leading to prices of between FF 300 and FF 500/m² as the floor price before work began. The compensation paid to owners in the transmitter zone, whose land then became closed to development in perpetuity thus came to FF 10-17/m².

Although not negligible, this sum represents only a fraction of the value of building land in the zone. Furthermore, the value has remained unchanged or has even dropped over the past four years.

Le Grand Bornand (Haute-Savoie)

This winter sports resort, close to the town of Annecy, has been using transferable development rights for the past 18 years. The technique is applied on a regular basis in one zone only, the Bouchet Valley, and has caused no particular difficulties.

In order to promote full use of development rights in the receiver zone, the land-use planning regulations recently made it mandatory for any project to transfer at least 50 per cent of all development rights usable in the zone. This does not merely provide an incentive for urban development to comply with the aims of the land-use plan, but is also a means of “storing” development rights on receiver land to avoid having to relaunch the procedure should the project be extended in the future.

Table 2 shows the number of transfers made since the scheme began.

Lourmarin (Vaucluse)

The situation in the commune of Lourmarin is quite different in its origins, practice and the conclusions to which it leads. It is an attractive area in the Vaucluse where land is already being “nibbled away” by both primary and second homes.

Table 2. **Number of New Building Projects Authorised in the Bouchet Valley**

	Number of building projects
1985	2
1986	3
1987	5
1988	1
1989	2
1990	0
1991	3
1992	1
1993	0
1994	3
1995	2
1996	3
1997	0
Total	25

Source: Johnston and Madison (1997).

The process, launched in the late 1970s by a resolute municipality, included agricultural areas in transmitter zones. This set off a long-standing dispute with the urban development authorities, which have remained unenthusiastic about the technique. It has however worked well under the strong leadership of the municipality which launched it, and which maintains the “market” in development rights.

A development rights “exchange” has been established, run by the Mayor's office and the Lourmarin notary. Currently about three-quarters of the housing density foreseen for the receiver zone has been used. However, the price for transferable rights is remaining relatively steady at FF 15 000 to 20 000 per 50 m² of land where building is permitted. Since the density allocated to the transmitter zone is 50 m² per hectare, development rights thus cost FF 1.5 to 2 per m² of building land. This by no means fills the gap between the price of agricultural land and that of building land, but, as the Mayor says, it has given some relief to landowners in the conservancy area closed to development. In view of the complexities of managing the system and its modest results, the Municipality does not intend to take the scheme further.

Les Gets

This is another example of what can be done by a small mountain Commune, under conditions the Commune considers to be satisfactory. The urban development plan has designated a fairly large number of transmitter zones, and allotted a development right equivalent to a density of 0.05, the development rights ratio being one-third from the receiver zone and two-third from the transmitter zone.

The process was slow to begin. Often, a transfer led to no outcome (building project abandoned) and the number of transactions remains small (10 to 20 a year), even though they represent nearly 50 per cent of the entire market. The “price” of development rights, after a fairly rapid rise (to FF 50 per m² of land in the transmitter zone), has tended to stagnate or even fall in the past few years. In any event, the level is low in relation to the price of building land (FF 250 to 400 per m²).

Two comments are called for by the Les Gets experiment. Firstly, the process initially had difficulties in matching supply to demand in development rights. A pragmatic solution was found in that the notaries, who in France are “ministerial officers” (in other words, public officials), agreed to take on the task of organising the development rights market. As mandatory go-betweens in all transactions, they circulated information between those offering development rights for sale and those seeking to buy them. They therefore played a key role in the operation of the scheme.

The second notable feature has been the way the mechanism is perceived by landowners. The practice is a long-standing one (20 years old) and has accustomed owners of land closed to development to think of their land as having a development right attached to it, whatever the final location of the development. It therefore introduced a stiffness – a sort of “ratchet effect” – into zoning procedures.

Lastly, the Commune introduced an incentive making it mandatory for development rights to be acquired in the largest, and most attractive, of the zones set aside for urban development. This procedure – which is generally found satisfactory – illustrates the importance of the connection between zoning and the transfer of rights technique.

Since only a small fraction of the available transferable rights have yet been used, the reallocation of new rights is not yet a problem.

La Clusaz

The last French example, La Clusaz, is a high grade winter sports resort. As such, it is exposed to strong land market forces and high land and house prices. Transfer of development rights was introduced in 1983, in a part of the commune under considerable pressure, and where provision had been made for some development.

The urban development plan set the development right at 0.08 throughout the zone. In order to be allowed to build, owners of land in the receiver zone had to hold an area in excess of 700 m² and buy development rights from landowners in the transmitter zone at a rate of from 0.14 to 0.22.

Example: plot of building land: 800 m²

existing development right: $800 \times 0.08 = 64 \text{ m}^2$ of floor space

minimum purchase: $800 \times 0.14 = 112 \text{ m}^2$ of floor space.

The developer thus has to “purge” of its development rights a plot of 112/0.08 or 1400 m² in the transmitter zone. Surface areas and coefficients were set in such a way as to make transmitter capacity equal to receiver capacity. The procedure has worked and has enabled landowners to accept very different urban development regulations.

However, it has also led to thorny legal problems. Successive appeals to the courts have hampered the progress of the scheme. Furthermore, the prices reached did not meet expectations, keeping price differentials high where actual development was possible.

The system at La Clusaz, like that at Les Gets, is very complicated to manage, with a large number of transmitter and receiver zones; keeping it going calls for sensitive handling.

Transfer of development rights in New Zealand

Property law in New Zealand is similar to that in the US, in that it countenances compensation for urban development constraints (“taking”) when they cause loss of value. It is in fact the likelihood of having to disburse large sums of money to compensate owners of old buildings listed as of historic interest that has led towns and cities to take this path.

It is now considered a useful instrument for controlling development or redevelopment in town and city centres. However, it is not applied to outlying areas of natural countryside.

The tradable rights technique was not introduced in New Zealand until the early 1970s, or more exactly in 1974 with the approval of the Auckland City Central Area Plan intended to protect buildings of historic interest or architectural merit:

“... Schemes whereby air rights might be traded from one property to another. This facility is seen to be particularly beneficial in the preservation of old buildings of historic or architectural merit: owners would be encouraged to retain some older buildings and upgrade the structures to meet current requirements if financially compensated by the sale of any unused site potential under the present provisions.”

This instrument thus stresses the financial compensation aspect that renders more acceptable those restrictive regulations which may lead to a loss of market value. The Auckland scheme did not define the method at the outset. This failure was probably the reason why the mechanism remained virtually unused in its early stages. It was relaunched with the Revised District Scheme in 1987, which: i) specified the basis on which development rights were to be calculated – and explicitly described the bonus zoning system in which the owners of a listed building received an incentive in the form of transferable development rights if he agreed to restore and maintain his building; and ii) builders received a bonus in the form of an extra density allowance, if they bought tradable rights.

Lastly, as the key to establishing the rights market, the city of Auckland played the role of a development rights bank, storing rights and releasing them in relation to demand.

In the case of Auckland, the development rights market is also closely linked to the planning process. Transferable rights can only be used in two development zones, consisting mainly of high density office buildings.

Matters are quite different in other towns and cities in New Zealand where this method is applied, such as Wellington or Christchurch. Here, as with the decision of the US Federal Supreme Court in the Penn Central case, its application is restricted to situations where the owner of a listed building also owns a nearby site that can take over the development rights. This dual proviso is very restrictive, and it is clearly conceived as only one of a number of possible ways of compensating an owner who is subject to a constraint.

Some thoughts on evaluation

Trials of the practice have not yet reached a critical mass that would allow statistically-reliable conclusions to be drawn. Even though there are a fairly large number of examples in the US, the examples occur in different geographical areas, have different aims, use different operating methods and, naturally, show different results.

Many examples of the practice – generally on an informal level – occur among very small groups of owners, and operate by consensus without any formal legal or institutional framework being involved. Such examples, generally occurring in built-up areas, have a long history in the form of TDRs in the US and private law constraints in France.

On the other hand, use of the method in a vast geographical area by means of a universally-applicable mechanism formally established in advance is still fairly limited and represented by the few widely-quoted landmark examples discussed above. However, most have features specific to themselves that make it difficult to reach general conclusions.

The examples presented and the information available on other schemes nevertheless enable some assessment of the practice of attaching tradable rights to land to be made with respect to the various aims it hopes to achieve. After an examination of the real nature of the rights being traded, the key links between tradable rights and zoning are reviewed. Consideration is also given to the way these rights markets operate and the prices (market prices?) they generate, illustrating the distributive effects of the system. Lastly, a number of conditions are listed that seem vital, if the system is to work well.

Sufficient experience with the technique is still lacking

As already mentioned, experience with the TDR technique is still limited, and is therefore difficult to interpret in general terms. Although it is a fairly widespread practice on a very small scale in urban areas, and for the purpose of sharing development rights among neighbouring owners, relatively large-scale examples where a true market can operate (and where transparency and atomicity prevail), are much less common. Such examples are of considerable interest and have produced substantial results, but each has its own story and has only come into being through a combination of factors (economic, legal, institutional and development factors) that are not easy to reproduce. This partly explains the anecdotal nature of the examples discussed below.

Plurality of aims

The aims attributed to most schemes are generally environmental or architectural. The most frequent is nature conservancy (Pinelands, Lake Tahoe, Taninges).

Many examples, often smaller in size, focus on architectural merit and the preservation of listed structures. This is generally the case in New Zealand, in particular the major scheme in Auckland. Nevertheless, the most frequent objectives remain nature conservancy, preservation of sites of outstanding natural beauty, and protection of agricultural land in the vicinity of built-up areas.

Indeed, it is often in terms of surface area preserved for conservation in perpetuity that the success of a programme is measured – an area “conserved in perpetuity” being taken to mean one that has transmitted all its development rights and is thus closed to future development. However, this particular aim is frequently merely the backdrop to the prime objective of the technique, which is distribution, namely to provide compensation for the constraints society places on the use of property, or in other words, to render acceptable the inequalities created by zoning laws (which by their nature cause development rights to be distributed inequitably).

The goals of nature conservancy or preservation of structures of architectural merit are the prime objectives of the regulatory procedure. The transferable development rights technique is therefore more to be regarded as an intermediate instrument to facilitate implementation of a plan. The concept of equity promoted by transferable development rights schemes is discussed below in more detail.

The technique may also serve as a legal safety net for the planner. Even if the scheme is not actually operational, the mere fact that it is in place will enable disputes over compensation for constraints to be avoided. This is an important aim in North America, in view of the widespread litigation prompted by “taking”.

The legal status of transferable rights

In all countries that have made use of transferable development rights of one sort or another, the legal status of those rights has been a point of contention. Are they an integral part of property (even when destined to be used at another site), or are they merely a financial instrument to provide compensation for value lost as a result of a constraint?

This is an important point, both because of its impact on the legal appreciation of the issue (in France or Germany, for example, it is unlawful for planning restrictions to be liable to compensation, except in very special circumstances), but also because of the way it is applied and in the way compensation is assessed.

As explained earlier, in the US and Canada, the case law on zoning (“taking”) has played a key part in determining the way urban development operates and development rights are managed by providing for an owner to be compensated when a regulation considered extremely restrictive is viewed as a “taking”.

With the sword of Damocles represented by the threat of litigation under the Fifth Amendment to the Constitution hanging constantly over the planner, he may find transferable development rights a useful means of mitigating this threat. It was, in fact, as a consequence of a proceedings of this sort (the decision in *Penn Central Transportation Company v. City of New York*, 1978) that the practice was first introduced.

Although case law may have assimilated the principle that the title to one plot of land might include, even if only implicitly, a right to make a use of another plot, has set limits to the practice, which however have not yet been finalised.

The concept of property rights itself has never been finally defined. Many commentators turn to the idea of a “bundle of rights”, whereby ownership of land is constituted by a series of autonomous, separable rights – rights to use, to develop, to overfly, to cross, etc. However, this idea does not settle the question of transferable rights in legal terms, namely which of the rights attached to land are by their nature part of ownership (such as the right to farm the land) and which are rights whose attribution may be determined by the social function of the property (such as the right to build).

The *Penn Central* decision, which had a major impact on the case law on “taking” in the US, expressly took the stand that the transferable development right was a useful instrument to secure compensation for some of the effects regulation had on land and housing values. In Judge Brennan's own words explaining the majority view, “while these rights may not have constituted just compensation if a ‘taking’ had occurred, these rights nevertheless undoubtedly mitigate whatever financial burdens the law has imposed on appellants and, for that reason, are to be taken into account in considering the impact of regulation”. Whether or not “taking” had occurred, the procedure was an additional instrument that could help landowners to accept stringent zoning regulations.

The recent *Suitum* decision (*Suitum v. Tahoe Regional Planning Agency*, 1175 ct. 1659, 1997) gives a good illustration of current trends in the relevant case law, which has taken a clear stand on the point. Under the Lake Tahoe transferable developments rights scheme described earlier, the appellant, who had been refused a building permit because his land was listed as a Stream Environment Zone was awarded tradable development rights to compensate for the loss of value of his property. He then sued the TDR scheme on the grounds that the right to build was an inalienable part of property rights, was as such usable *in situ*, and was not transferable to another site. This was the first time such reasoning had been taken all the way to the Federal Supreme Court.

The *Suitum* decision was thus of major importance because it questioned whether it was lawful for the procedure to set limits to the financial implications of land-use regulations.

Such a development could well put a real brake on the practice of awarding compensation in the form of TDRs and thus strengthen the powerful Movement of Private Property Rights. The courts could thus find themselves facing an alternative: either “taking” occurred and is liable to full compensation, or it did not occur and there is no liability for compensation.

The decision did not in point of fact categorically rule out the technique as a means of providing compensation, but drew attention to the defects and inconsistencies of the Lake Tahoe scheme.

In a quite different context, a similar debate has been going on in France, although from a different starting point since the basic premise is that constraints are not liable to compensation. When the Act was drafted, voices were heard denouncing the risks involved in introducing rights that could be considered as imaginary (Lenôtre-Villecoin, 1975).

Like the *Suitum* decision in the US, this amounts to an attack on the very principle behind the creation of the legal entity of “the transferable right”. According to J. Lenôtre-Villecoin, “the capacity to transfer an imaginary development right establishes a *jus abutendi*, or a right of disposal, in a case in which the public interest, in the form of regulation of urban development, is against existence of the right to build at all. Perhaps it is time to deal finally with the inseparable trio (*usus*, *fructus* and *abusus*) which “abuse” opinion in Latin countries.

Zoning and transferable rights

There is a clear link between these two ideas. The transferable rights procedure is in itself a zoning instrument, since it implies a division into “transmitter” and “receiver” zones. Greater precision may well be introduced by stipulating that the zones must be of precisely specified dimensions; otherwise the whole scheme will be invalidated. If the scheme is to operate properly, owners in both transmitter and receiver zones need to be given the right incentives, which should help to balance supply against demand with respect to development rights.

In the case of receiver zones, where conventional planning regulations operate as usual, the purpose of zoning is to ensure a high standard of urban development. The quality of urban development is thus the main criterion to be taken into account.

As the process develops, however, it becomes difficult to provide adequate incentives. Many schemes, such as the Pinelands one, use a system of “bonus zoning” – in other words, the density authorised increases if transferable rights are purchased. This makes it very tempting for the planner to reduce the “ordinary” density (where no rights have been purchased) and increase the “bonus” density. However, such a policy is likely to fall afoul of the principle of vested rights, and to lead to litigation. Incentive zoning is thus a difficult process to handle.

Another sensitive issue is the eligibility of a zone to be designated a transmitter zone, which opens the way to a grant of transferable rights. The subject is one of endless debate, with no evident way of settling it on a systematic basis.

First of all, it involves agricultural land. Should such land be allocated transferable development rights, and if so, on whose behalf? Generally speaking, the price of agricultural land could be considered to reflect its productivity, the current net value of its future yield. It is paradoxical to allot “development rights” to land on which farming is expected to continue even if the rights are not to be used on that land.

Arguments are often based – this is generally the case in France and in most of the American examples – on the natural beauty and biological diversity of the site. However, this takes no account of existing usage and places owners having quite different relationships with their land on the same footing.

Although no general conclusion can be reached on the linkage between zoning and transferable rights, note should be taken of the risk of distorting zone demarcations and urban planning regulations in order to make the rights market work. It is important to maintain a proper perspective; the transferable rights procedure is no more than an aid to good urban planning, and not an end in itself.

Technical and institutional feasibility

The examples described above make it clear that sensitive management is needed to make the TDR technique work. However, it is also important to do away with the misconception that the TDR procedure requires strong and stable institutions if it is to operate properly.

The landmark examples in the US all rely on an *ad hoc* institutional base – such as the Pinelands Commission – which assumes responsibility for the whole area concerned, without any right of oversight by common law institutions, counties or districts. This structure takes care of both zoning and management of development rights.

On the other hand, when an elected council is responsible for operation of the system, an election often succeeds in undoing a previous body's achievements. Another difficult matter, in addition to zone demarcation as mentioned above, is management over time. The initial allotment of rights in transmitter and receiver zones has to correspond to some point of equilibrium at some given time in order to enable the market to operate, and give sellers an incentive to sell and buyers to buy.

What about the long-term dynamics of the system? From the standpoint of the seller, a plot of land that has been stripped of its development rights is closed to development in perpetuity. As time passes, and if demand remains at a given level or increases, pressures will build and the price of rights will increase. Should more rights then be allotted to land that has been stripped of their rights, and under what circumstances?

Can the initial allocation of development rights be considered immutable (this is the view of the Pinelands and Lake Tahoe schemes) even if this is likely to lead to a price boom? What should be done with regard to owners in the transmitter zone who hold on to their rights, thereby speculating on a rising market? Should their development rights be expropriated?

None of the examples discussed has yet had to face this problem. Either the mass of rights has been very large, or demand has been overestimated, or incentives were not strong enough, or the scheme was too new, or the problem has not yet taken an acute turn. Its advent is inevitable and the reallocation of rights will require very careful handling. In any event, schemes need at the outset to determine what the medium-term prospects are, and to define the reallocation procedure to be used.

Another technical question to be resolved is the way rights are to be managed. In order to allow the market to operate, there has to be a body with the power to buy, stock, and sell rights at any given time. In view of the special nature of such a market and the likelihood of discontinuities, it is a mistake to think that it will be able to operate on a decentralised basis from the start. Here again, the examples that are working well are doing so with the help of a “development rights bank”, which operates in close association with the general aims of the system.

One last need is to educate owners, both transmitters and receivers. Trading in transferable development rights is an activity that is not self-evident and needs extensive explanation and demonstration in order to work properly.

Is the price of rights a “real” market price?

It would be pleasant to be able to answer this question in the affirmative, taking “market price” to mean the price that would balance supply against demand under conditions of atomicity, transparency, etc.

Even in the most successful cases, the number of transactions involved (low) and the time the scheme has been in operation (short) does not allow any statistically-significant findings to be made.

The only firm conclusion, reached in settings as different as Auckland, Montgomery County or the French Commune of Taninges is that prices rise sharply when the procedure is beginning to settle down, and then level off or even decline. Could the status of the international property market have had anything to do with this? It is also a fact, however, that acquisition of transferable rights, when not mandatory, places a burden and a brake on the procedure which every effort should be made to prevent.

To be more specific on this important point, there has to be a way to make detailed analyses of local markets in order to set the “market price” of a development right on a countdown basis (from the market price of the end product, the building, less the costs of the operation, a deduction is made for the highest likely level of land tax, and thus, the value of the development rights to be purchased). The actual price will probably then be seen to be nowhere near this “market price” unless the purchase of tradable rights is mandatory and there is no alternative nearby (the development zone is not subject to the transfer system). This comes back to the paradox mentioned earlier, that this system of tradable rights will only work properly in a context where land use is subject to strict planning regulations.

The concept of a “real market price for development rights” can be taken even further. Although it can have a meaning at a given point in time, this may prove an illusion when a look is taken at the way the situation may develop. The allotment of development rights is carried out by way of a social and political mechanism that is forced on the various players in the market. A public authority may well (as is frequently the case) decide to restrict or even freeze urban development, with foreseeable repercussions on land prices. Unless other corrective measures, which will be described later, are applied, the price of the development right will rise indefinitely. The price of the development right, even though it may reflect the point of equilibrium between supply and demand at any given time is overestimated as a rule by urban development, which is bound to change.

Equity and efficiency

On the subject under review, the concept of equity has to be considered from the point of view of landowners, and from the point of view of the inhabitants of a zone as a whole.

In the first case, tradable rights fulfil an essential function in the absence of a fiscal system capable of removing added value. The price of land is very dependent on the development rights allowed by the zoning regulations, tradable rights making it possible to correct inequities introduced by zoning.

If the concept of equity is extended to all inhabitants, assessment of the method becomes more difficult, and depends on the way property rights are conceived, and on the tax system allowing them to be put into operation. There are two contrasting situations:

- In some countries, such as in North America and south-western Europe, there is no universally-applicable mechanism for recovering capital gains by urban development and/or payments for development rights. The practice of transferring development rights is equivalent in such cases to distributing the overall capital gain by urban development among landowners only, whereas it might well be expected to return to the community as a whole, in particular when the public amenities that give rise to that added value are funded by the taxpayer. Introducing rights transfers is thus, roughly speaking, a means of formalising a transfer from those acquiring houses to the landowners. This is a limited view of equity, which may admittedly be of help in particular cases, but which has no general application in this context, unless (which is unusual) land is divided up in a very comprehensive and equal way among the inhabitants.
- The second type of situation, which is found mainly in northern Europe, is founded on the principle that the capital gain by urban development should return – at least in very large part – to the community. Using various methods (long-term public land holdings in Sweden, municipal monopoly on the supply of building land at controlled prices in the Netherlands), the initial procedure that increases the value of land essentially benefits the community rather than the landowner. The equalisation made possible by transferable rights therefore serves no purpose.

We are thus brought back once more to the definition and content of property law, the key to the problem.

As noted by Donal Krueckeberg (Krueckeberg, 1996): “Property is not just the object of possession or capital in isolation, but a set of relationships between the owner of a thing and everyone else's claim to the same thing. This understanding of property highlights considerations of distributive justice that are particularly important in light of the issues in the contemporary debate about property rights. Rights to personal use of property are fundamental to individual and social well-being: rights to profit from property, in contrast, have always been subject to reasonable constraints for the benefits of the entire

community and society. Attempts to establish a contrary case by appealing to natural right, market necessity, liberty, social utility, or just desert all fail to withstand scrutiny... These concepts of use rights and profit rights in property are at the heart of the planning question."

Some necessary conditions

The following is a summary of the main conclusions to be drawn – explicitly or implicitly – from what has been said above:

- The method will only work if there is a demand for development rights; in other words, if there is added value to be shared out, and also an urban development plan authorising development.
- Incentives are necessary (bonus zoning) where transfer is not mandatory.
- Receiver and transmitter zones must be sufficiently large to allow the development rights up for exchange to reach a critical mass.
- A special body is needed to ensure that the system is properly managed, to keep records of supply and demand, to follow developments over time, to monitor the use made of plots closed to development, and to ensure that price information is transparent. Such a body should help to keep transaction costs down.
- The public authorities (or a special body) should take on the task of operating a development rights exchange, operating a development rights bank, and perhaps regulating rates.
- The transferable rights system should be closely aligned with zoning regulations and any later amendments thereto. Zoning must be firmly established and independent of market forces in development rights.
- Introduction of the scheme calls for educational efforts to explain a mechanism whose workings are not immediately obvious. Motivating owners in the receiver zone is crucial.
- Demarcation of conservancy areas (transmitter zones) must be based on clear and explicitly stated criteria in order to gain the active support of the local inhabitants.

Conclusions

The preceding discussion reflects a degree of disenchantment with a method that was brought into being with great hopes. Although, under some conditions, the theoretical virtues of bringing the tradable rights system into widespread use seem attractive, more rigorous reasoning and the trials made of the method seem to restrict its scope.

The method has made it possible to solve particular problems. The most appropriate targets would seem to be highly attractive areas where owners of second homes wish to protect their environment without losing the right to sell plots (or rights) at a good price in a context of strong demand and hence of rising land prices. This is the case, for example, of Lake Tahoe or the Praz de Lys plateau at Taninges.

However, widespread use of the method, in addition to a number of technical and institutional difficulties that are hard to overcome, would have distributive effects that could be contested on ethical grounds.

Firstly, the only examples where the system has worked well are those where the tradable rights procedure has been closely tied to a strict land-use plan that has remained relatively unchanged over time. Where zoning is too flexible, the procedure would eventually lose credibility.

Another requirement to make the system work is careful and sensitive management. This has generally been achieved in the examples given, but the many unsuccessful cases where this condition has not been fulfilled should not be forgotten.

The success of the method (very relative) in the US is due to the specific legal content of its property law and above all to its case law on added and lost value. The radical differences in practice with west European countries in particular may partially explain the difficulties of transferring the system.

The last word may be left to Ann Louise Strong, an expert in the field, who showed great interest in and enthusiasm for these schemes at the outset. She now has a more reserved outlook: "Although there has been some success with TDR, we are sceptical of the technique in general. It has generated far more discussion than actual land preservation. There are serious practical and legal obstacles to implementing it. [...] The author believes that the limited success stories, such as the one in Montgomery County, Maryland, results from regulation, with TDRs as only one small part of the programme" (Strong, *et al.*, 1996).

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Chapter IV

TRADABLE PERMITS IN WATER RESOURCE MANAGEMENT AND WATER POLLUTION CONTROL

by

R. Andreas Kraemer
Kai M. Banholzer

Ecologic – Centre for International and European Environmental Research, Berlin, Germany

Introduction

As a contribution focusing on water as one environmental medium, this chapter is in large part descriptive, since much of the empirical evidence on the application of tradable water abstraction and pollution permits has hitherto been scattered and has simply been brought together here.¹ The result redresses the often-held belief that tradable permits are rare in water management, and are more often found in the field of air pollution control through “bubbles” (*e.g.* OTA, 1995).

A standard set of questions underlies each case study, even though not all could be answered in any of the cases studied. These questions include: Where does the tradable permit system apply? Who trades what, and under what conditions? How was the trading system designed and how is it administered? What approaches are taken to monitoring, reporting and enforcement of the trading rules?

The description based on these questions then forms the background to an investigation of the way in which the tradable permit system is applied in each case. In particular, implementation problems are addressed, including the compatibility of the tradable permit system with other environmental regulation and enforcement regimes. One specific concern is the effects of systems on non-participants (third parties) and the protection of the legality and efficiency of the tradable permit system itself. Concerning the functioning of the trading system, the interface between buyers and sellers of permits, entry to and exit from the market, and the publicity for trades and prices as well as the issue of transaction costs are also addressed, where such information could be obtained.

Methodology

The research leading to this report was carried out mainly as a desk study on the basis of published documents and unpublished material. The analysis of these sources was complemented by information from water management experts and economists; some of the information was obtained through interviews in person, some by telephone or correspondence. In general, information on tradable systems for water resource management and water pollution control is very limited. Only a few OECD Member countries appear to have water-related tradable permit systems so it was also deemed appropriate to include cases from non-Member countries in this report (Chile), in order to provide a more complete picture. Many of the cases are highly specific responses to particular environmental pressures and it can thus not be assumed that any effects observed would necessarily appear more generally. Finally, in many cases, information is sketchy and does not give answers to all the questions presented above.

Whilst much research remains to be done, it is possible to establish a simple typology of cases which underlie the structure of this report. These cases also allow for provisional assessments to be made of a number of issues, which are presented later.

A simple typology of cases

When discussing tradable permit systems relating to water, three fundamentally different fields of application can be identified:

- Tradable permits for water, or tradable water abstraction rights, for *quantitative* water resource management. Such water rights can be permanent and unlimited (property rights to the water source) or temporary and limited (transferable rights to use water without right of abuse).
- Tradable discharge permits, or tradable water pollution rights, for the protection and management of (surface) water quality. Such pollution rights can relate to point or to non-point sources, and trades can even be arranged among different kinds of sources.
- Tradable permits to use or consume water-borne resources, such as fish or the potential energy of water at height, or the kinetic energy of flowing water. This third category is not considered in depth in this chapter, but interesting case studies can be found. The case of the Scottish salmon fisheries (Box 1) shows that trading may work, as long as there are no significant externalities (*i.e.* impacts on, or from, other water uses or functions). However, the conflict between fishing as a recreational activity (rather than to secure the nutritional base of the anglers) and conservation requirements is also highlighted.

Box 1. Freshwater Fisheries: Fishing Rights in Scotland

In Scotland, responsibility for protecting and developing inland salmon fisheries rests with District Salmon Fishery Boards. Unlike in England and Wales, individual rod licences (fishing licences) are not issued. Instead, salmon fisheries are privately-owned and operated by the owner or tenant, within a legislative framework set by central Government. Although salmon does not “belong” to anyone, there is no public right to fish for salmon. The right to fish belongs to the person who owns the exclusive rights at any one site (fisheries). In most of Scotland, such rights are owned independently of the land itself (Scottish Office, 1997).²

The Crown Estate still owns many fisheries and leases them to fishermen on standard five-year leases. Elsewhere, rights may be held by individuals, public companies, businesses, or fishing clubs. Fishery owners in any District may set up a District Salmon Fishery Board. The owners can rent their fishing rights to others, and where they do so, it is usually on a daily or weekly basis. Time-sharing has also become increasingly popular in the last 10 years, so that individuals get a lease to fish for specific period of the year.

The majority of salmon anglers pay to rent a fishery for a specific period of time. The rental price depends on the prospects of catching fish, and is often based on the five-year average catch. On the major Scottish salmon rivers (*i.e.* where the great majority of fish are caught), prices for purchasing beats currently range from £6 000-8 000 per fish, based on the average catch per year for that beat. The fact that individuals own the exclusive right to fish at a site (*e.g.* river or loch) is now considered to be one of the main obstacles to the designation of freshwater habitat protection areas in Scotland.

Within the first two fields of application, a number of further distinctions can be made. Concerning tradable water rights, these relate to the “*intensity*” of trading, which can be permanent or temporary (seasonal), or even one-off. In addition, there are some instruments, such as taxes or specific institutional arrangements which can be regarded as *equivalents to tradable permit systems*. In relation to water pollution rights, further distinctions can be made according to the *polluting substance* (or class of substances) in question.

Structure

The next Section of this chapter describes the policy background to the use of, and interest in, tradable permits in water resource management and water pollution control. This is done primarily for the few countries where experience with such market-based instruments is concentrated: the Commonwealth of Australia, the US, and Chile. The European Union is also considered because it is an interesting case where legislation gives scope for emission trading, but where no Member State at present uses this scope. The report then focuses on the three fundamental fields of application indicated above, with further case distinctions being made. Within each of these fields, a number of case studies are documented, which form the main part of this report. In many cases, successful tradable permit regimes are built on the foundations of pre-existing water management institutions, such as water management associations, which manage water (and sometimes water pollution) as a common resource. A few conclusions are drawn at the end of the paper.

Policy background to water-based tradable permits

Disagreements over the allocation of water from shared resources are now frequently identified as potential causes of conflict among segments of the population sharing these resources or, in the case of international water bodies, even of conflict between states. In spite of the importance of water resource allocation in the international debate, little attention has so far been paid to potential economic mechanisms for efficient water allocation. One obvious such instrument is the creation of transferable rights to use (or to pollute) water, and the creation of efficient markets in which these rights can be traded. Even though the approach appears to be conceptually sound and should be attractive for efficiency reasons, only three countries have accumulated much experience with water-based tradable permits. This is also evident from a broader overview of recent development in OECD Member countries by Jones (1998). The three countries in question are the Commonwealth of Australia, including many of the Australian States, the US (US), including some, but by no means all, of the US States, and Chile (cf. Anderson and Snyder, 1997: 19 FF).

The two OECD Member countries, Australia and the US, are both federations where examples of tradable water permits are found in more than one state. There have been different patterns of diffusion in the use of tradable permits in both cases, and the experience is not at all the same. Nevertheless, the hypothesis can be established that federal structures facilitate innovation in the use of policy instruments because they provide freedom for regional (state) authorities to experiment, and to create a framework which facilitates “policy learning” on the basis of these experiments. The European Union (EU), which in some ways resembles a federation, provides another example (although for the time being a negative one). The EU does on occasion make use of “bubbles”, for instance in the implementation of the Montreal Protocol on substances that deplete the ozone layer. It has also adopted a provision allowing for water pollution trading, but this has not yet been applied anywhere; thus, the postulated process of “policy learning” cannot currently be observed in the EU with respect to tradable permits in water management.

There are so few examples of “pure” trading instruments in water management and pollution control that the question arises as to what the alternatives are. As seen in the case studies described below, tradable permits are a new solution to an old problem, and there are often “equivalent instruments” (*i.e.* instruments that serve the same purpose as tradable permits) that have traditionally been used in water management. These include environmental quality objectives and emission limits, economic instruments and specific institutional arrangements, including financing mechanisms for water allocation and other water uses. A few such examples are included in this report. Some relate to the Federal Republic of Germany, a country in which there is some (theoretical) interest in tradable permits in water management (SRU, 1998), and which shares many of the constitutional constraints, as well as the freedom of decentralised units to develop policy, that are found in Australia or the US.

Australia

In the Commonwealth of Australia, several States have introduced transferable rights for water use (James, 1997: Chapter 11) and there is some experience with tradable pollution permits (James, 1997: Chapter 10). The States of New South Wales, South Australia, Victoria and Queensland have all established legislative provisions for transferable water use rights. In Western Australia and Tasmania, trial schemes have been established and transfers are being considered for some regions. These developments, even though fairly recent, have been the subject of various studies.³

However, there is still no example in Australia of a free market for trades in water rights, and a number of restrictions are imposed by water management authorities. These include restrictions on the spatial transfer of water use, volume controls, and environmental considerations such as the preservation of river flow, control of salinity, and protection of wetlands and river ecosystems. The water authorities facilitate trades, rather than participate in the water market, although they usually are empowered to veto trades if the conditions are unacceptable. Most agencies monitor trades and keep registers of buyers and sellers (James, 1997: Chapter 11).

The New South Wales Environment Protection Authority (EPA) has introduced a pilot system for salinity control in the Hunter Valley, and a number of States are considering tradable discharge rights (James, 1997: Chapter 10). Tradable rights have also been introduced for the Murray-Darling Basin, providing for salinity trades between New South Wales, Victoria and South Australia (inter-State trading). New South Wales is adopting a supplementary scheme, but this scheme is limited in both scope and potential impact.

An important factor in the development and diffusion of the use of tradable permits for water resource allocation and water pollution control has been the co-operation over the river Murray and its tributaries through the River Murray Commission (later the Murray-Darling Basin Commission), established originally by the River Murray Waters Agreement. Since 1983, the Commission's brief has included environmental and recreational uses, and in 1985, the geographic limits were expanded to include the entire Murray-Darling Basin. Parallel to the expansion of the Commission's role, a number of other changes have taken place in Australian water resource protection and management, including the privatisation of several irrigation areas, the commercialisation of and, in Victoria, the break-up of water authorities (Sturgess, 1997). The Australian case studies in this report illustrate that a culture of water trading (and later also of pollution trading) has developed in this specific environment of inter-state co-operation.

United States

According to Howe (1997), water markets in the US have a "long but rather narrow history", with their roots going back to irrigation water auctions in Spain in the sixteenth century. The Western and Southwestern regions of the US apply the "prior appropriation" doctrine, according to which the first to use a water resource acquires a right to use water, for instance, from streams or aquifers. Such rights are recognised as personal property that can be transferred, sold, or leased to other parties and purposes, usually on condition that the water is put to "beneficial use", and that there is no damage to other water users.

Howe (1997) relates the results of a study by MacDonnell (1990) of permanent transfers of water rights in six Western states: Arizona, California, Colorado, New Mexico, Utah, and Wyoming. According to this study, Arizona is highly dependent on groundwater and restricts transfers from overlying lands. California is highly dependent on federal and state water supply projects, and the major projects carried out by federal organisations generally disallow transfers out of districts that have contracted for water. Within these constraints, Colorado and New Mexico have developed active water markets, as did Utah. Wyoming had few transfers because it experienced little pressure on its surface water resources.

In the two most active states (Colorado and New Mexico), the predominant form of transfer was from agriculture to non-agricultural uses, but transfers among agricultural users remained important, as a result of both increasing consolidation and commercialisation. Utah experienced significant transfers between non-agricultural users, especially those involving cooling water for large thermal electric installations.

At least three states, Wisconsin, Colorado and North Carolina, have established programmes for trading credits for reducing water effluents. The Wisconsin programme, which allows point sources to trade water effluent reduction credits along the Fox River, has been in effect since 1981. Colorado has two programmes to trade credits for rights to further reductions of phosphorus discharge into reservoirs in the Denver area: Dillon and Cherry Creek. More recently, North Carolina implemented a programme that will permit trading in further reductions of nutrients released into the Tar-Pamlico Basin.

Chile

Chile is a developing country that has encouraged the use of markets in water resource management (Gazmuri Schleyer, 1992; Brehm and Castro, 1995; Hearne and Easter, 1995). In 1981, a system of transferable water-use rights had already been established by the new *National Water Code*. Under this law, the state grants existing water users (farmers, industry, water and power utilities) property rights to water without charge. These rights are independent of land use and land ownership, therefore trades of water rights are not tied to land sales. The state also auctions new rights. These rights can be traded by, and transferred to, anyone at freely negotiated prices. Although the law specifies the rights by volume of flow per unit of time, in reality, rights are defined as a share of stream flow. Rights are also divided into:

- Either permanent or contingent rights. Contingent rights are granted for surplus water (*i.e.* water flows that have not been assigned by permanent rights holders).
- Consumptive and non-consumptive rights. Non-consumptive rights, used for example for hydro-power generation, grant the use of water, as long as it is returned to its source at a specified location and quality. The bulk of the 300 000 owners of water rights own consumptive rights, with agriculture being the main use. However, the overall volume (in m³) of non-consumptive rights is twice that of consumptive rights.

The monitoring, distribution and enforcement of rights is carried out by water user associations. The *Water Code* does not actually set up a market for water rights. Instead, it attempts to set up the preconditions for such a market to emerge spontaneously, where needed. The State rights agency, the *Direccion General de Aguas* (DGA) issues rights free of charge whenever water is physically and legally available. Once constituted, all water rights are governed by civil, rather than public, law.

European Union

The European Union can adopt Directives that are legally binding on its Member States. Among its legislation concerning water resource protection and management, the Urban Waste Water Treatment Directive (91/271/EEC) has a reputation for being the most expensive item of European legislation in the environmental field. Its purpose is to stimulate Member States to invest in the collection and treatment of urban waste water. Different requirements and deadlines apply to “sensitive”, “normal”, and “non-sensitive” areas, meaning water bodies and their catchment areas. The Directive leaves the Member States much freedom in its implementation, such as a choice between limit values for treatment plant effluent and percentage reduction goals or a choice between reducing phosphorus (P) or nitrogen (N).

In sensitive areas (*i.e.* areas tending towards eutrophication, because of excessive levels of P and N), adequate collection and “more stringent than secondary” (*i.e.* tertiary) treatment systems were to be installed by 31 December 1998 for all discharges from agglomerations of more than 10 000 population equivalents. Discharges from such systems must meet emission limit values for either P or N. The limit values for P are 2 mg/l in agglomerations of between 10 000 and 100 000 population equivalents and 1 mg/l in larger agglomerations (measured as P). The limit values for N are 15 mg/l for agglomerations of between 10 000 and 100 000 population equivalents and 10 mg/l in larger agglomerations (measured as N). Alternatively to the use of limit values, P may be reduced by 80 per cent or N by 70-80 per cent.

The Directive makes provisions for trades in P and N emissions. Article 5(4) of the Directive states that the above requirements need not apply in sensitive areas, where it can be shown that the minimum percentage of reduction of the overall load entering all urban waste water treatments plants in that area is at least 75 per cent for total P, and at least 75 per cent for total N. With the wording “overall load entering all urban waste water treatments plants”, this Article clearly opens the possibility for emissions trading

within an “emissions bubble” thus described. However, it also establishes restrictions. Notably: *i*) the “bubble area” must be a “sensitive area” within the definitions of the Directive; *ii*) the reduction would need be attained over all urban waste water treatment plants and not only the larger installations; and *iii*) the reduction probably must be attained for both P and N simultaneously. The weight of these restrictions is not clear, but is unlikely to present serious obstacles to any pragmatic implementation of an emissions trading regime.

None of the EC Member States so far appear to have taken advantage of the possibility of establishing emissions trading in “sensitive area” bubbles, and the possibility appears to not even have been discussed among the national experts in the Technical Committee established under the Directive. As an indication of some interest, the Netherlands have mentioned the possibility in their first implementation report to the European Commission, and have asked a national committee to develop scenarios. The evident lack of general interest so far may be regrettable, since a potentially important source of economies in pollution abatement costs remains untapped, in spite of the wide-spread concerns about the financial implications of the Directive.

Tradable water abstraction rights

Most of the trading in water abstraction rights appears to be related to agricultural use, predominantly for irrigation. The examples described in this Section relate to the longest-established institutions in this context. In fact, some of the trading schemes have developed from much older, traditional water allocation regimes, based on community associations.

This Section describes cases of water trading, including banking, in the US (in Idaho, California, and Texas). All case studies were selected because of the long-standing nature of the underlying institutional arrangements, as well as for interest in their recent evolution.

A cluster of case studies on irrigation water trading in Australia is introduced in a description of the origins and operation of the Murray-Darling Basin Commission, in whose territory all of the irrigation case studies lie.

One of the main objectives when introducing tradable permits to use water is often to provide an instrument for the re-allocation of water rights so they can be put to more (economically) beneficial use. The case study on a water abstraction tax shows how this objective can also be attained by another economic instrument.⁴

Trading of water abstraction rights in the United States

Idaho Water Supply Bank on the Upper Snake River

A system of water banking, using reservoirs on the Upper Snake River in Idaho, has been in operation since the 1930s (Wahl, 1989: 133-135). Idaho formally authorised its ongoing water allocation arrangements by establishing state water banking legislation in 1980. This legislation provided for state-wide banking operations through either short-term or long-term leases. Initially, however, the only active banking operation was the one established prior to the 1980 Act, on the Upper Snake River.

Prices, or “rates”, for water supplied through federal water projects are very low (about \$0.19 per acre-foot). Because of this, some storage space holders maintain large water holdings behind dams as an insurance for water-short years, and recover their costs by placing unneeded supplies in the bank. The bank leases storage space in Bureau of Reclamation reservoirs. It is managed by a local “Committee of Nine”, which sets prices and has established a priorities for water uses.⁵ At the end of the 1980s, the Bureau required the price set by the Committee of Nine to be based on space-owner costs (recovery of capital and operational costs), and also required leases to be restricted to one year.

To encourage early commitments of water to the bank, water made available before 1 July is sold first, and revenues from sales are divided proportionally.⁶ Water made available after 1 July is sold on a first-come, first-served basis, and the water owners are reimbursed in a similar manner. The established price rose from US\$0.30 per acre-foot in the 1950s, to US\$0.50 per acre-foot up by 1977. Since the late

1980s, it has increased to US\$2.50 per acre-foot, which includes a US\$0.50 per acre-foot charge accruing to the bank for updating its equipment for measuring water. These prices are higher than the average of US\$0.19 per acre-foot for federally-supplied water.

Water supplied to the bank has exceeded 800 000 acre-feet in some years, although the amount actually sold by the bank is usually considerably less. The principal buyer has been the Idaho Power Company, for hydroelectric power generation at times of high electricity prices. Irrigation constituted the remainder of the purchases during these and other years (Wahl, 1989: 133-135).

Northern Colorado Water Conservancy District

The Northern Colorado Water Conservancy District (NCWCD) in the Fort Collins area was founded in the 1930s to contract with the federal Bureau of Reclamation. Its objective is to serve the water demands on the east slope of the Continental Divide, by using the more abundant water supplies from Colorado's west slope (Wahl, 1989: 135-136). The Bureau's Colorado-Big Thompson Project (C-BT) diverts water at the headwaters of the Colorado River and transports it through the Continental Divide to supplement the water supplies of about 720 000 acres of irrigated land, as well as the supplies of several east-slope cities.⁷

There are three principal types of water rights in the NCWCD area: rights to C-BT water imported from the west slope, rights to water stored in irrigation company reservoirs, and direct flow rights from east-slope streams. Although water rights from each category have been transferred, C-BT water is the most easily marketed. Each year, the NCWCD divides the amount of available C-BT water proportionally among the owners of its 310 000 shares or units. Shares can be bought, sold, and leased within the district. For example, about 65 000 acre-feet, or some 30 per cent of the water delivered to the district by the C-BT, moves through the rental market every year. Annual rentals require no more than a postcard to the NCWCD to shift the water to a different use or location. Sales of water on a permanent basis are also relatively frequent and straightforward, and a number of property agencies specialise in brokering such transactions.

The price of shares of C-BT water has varied widely. The average price for permanent rights (all expressed in 1980 dollars) was US\$99 per acre-foot equivalent in 1961; US\$504 in 1970; US\$2 895 in 1980; US\$2 445 in 1981; and about US\$900 in 1985. In the rental market, prices rose slightly from 1961 to 1983, but at less than the rate of inflation. The predominant annual rental price is US\$5 to US\$7 per acre-foot. Prices are slightly lower in the early part of the irrigation season than later on (Howe *et al.*, 1986). The rental market has been stimulated by cities within the district that regularly acquire water in excess of present demands so as to ensure sufficient supplies for the future.

The fact that C-BT water is a supplemental supply appears to have played an important part in its marketability. Different areas in the NCWCD have different demands for the additional water, and a mandatory, uniform assignment of water to land was not deemed appropriate when the district was established. Freely-transferable allotments allow farmers to recover water costs whenever they have no need for their supplemental supply (Wahl, 1989: 135-136).

California Water Bank: from emergency water trading...

After a severe drought period between 1987 and 1992, during which the annual water availability was less than half that of normal years, California responded by renewing its water management system. One element of that renewal was the introduction of water trading and water banking, which resulted in a marked increase in the efficiency of water use (Chakravorty and Zilberman, 1997).⁸

In California, agricultural and urban water demands peak in summer, while precipitation peaks in winter. Also, there is a large difference in rainfall volumes between North and South. In response, the State has developed an extensive system of reservoirs, canals and aqueducts. Water is collected in the North during winter months and transferred to the South in the summer. In the 1980s, water planners realised that they needed a continuous improvement in water efficiency at the farm, district, and state levels, as the only way to match supply and demand. Due to differences in crops and technologies being used, there were wide variations in the economic productivity of water use. For instance, rice requires large amounts of water, while strawberries need less, and also fetch higher prices.

The allocation of water in California follows the *prior appropriation doctrine*. Farmers are assigned water-use rights according to the time they apply (first-in-time, first-in-right). They are not restricted to a fixed quantity, so long as they put the water to a “beneficial use”. They can lose the rights, however, if they do not use them for a certain period of time. They are not allowed to trade the rights. Obviously, this system sets no incentive for the farmers to use water efficiently, but rather to apply for as much as possible, and to consume all of what they receive.

The drought of 1987-92 forced farmers to install new technology, but did not affect all farmers evenly. Farmers with senior rights had enough water to continue their businesses, but owners of junior rights were restricted. In response, trading of water within and between irrigation districts was introduced and a water bank was created. It bought water from districts with a surplus and sold it to those with a deficit. The total amount bought by the bank was 825 000 acre-foot at \$125 per acre-foot, and it sold 435 000 acre-foot at \$175 per acre-foot. The experience of the drought period showed that farmers will respond to scarcity and to economic incentives by using water more efficiently.

... permanent trading through “Water Links” in California

In 1991, the California Department of Water Resources established a state-wide water bank to permit both public and private users to buy and sell water on a one-year basis. The bank served as a middleman, with its offers to purchase at US\$125 per acre-foot and to sell at US\$175 per acre-foot. Transactions were facilitated by the fact that much of California's water supply is from storage and therefore could be sold and released as required. The 1991 California Water Bank also worked efficiently for other reasons (Howe, 1997):

- Private parties could sell water from private rights and water districts shares.
- Sale water could come from fallowing irrigated land, substituting groundwater for surface water, or selling water in storage.
- Offer prices per acre-foot were set on the basis of farm budgets for the various crops that might be fallowed (primarily alfalfa, corn, pasture, rice, sugar beets, and wheat), making it slightly more profitable to sell water than to plant crops.
- The selling price per acre-foot was set to cover the purchase price, the administration costs, and the cost of purchased water released to the Bay Delta to replace water pumped into the San Joaquin delivery system.
- About half the water purchased came from fallowing wheat and corn acreage in the delta area.

Bank purchases exceeded sales by 400 932 acre-feet because March rains dampened demand. Sellers were encouraged to “commit early” by guaranteeing them they would receive any higher prices negotiated later in the season. On the other side of the market, buyers were not required to guarantee their purchases, and many withdrew their offers when rain appeared. The excess water was purchased and stored by the State Water Project.

Towards the end of the drought, the US Congress passed the *Central Valley Project Improvement Act* which, among other things, allows water contractors to sell water outside their districts. Subsequently, process water trading was encouraged, mainly through two mechanisms. One is the *buy-back scheme*. The District would buy surplus water from farmers and sell it to those who needed additional water. The District in this scheme took the role of an intermediary (or market-maker). The other mechanism is the electronic water market. The Natural Heritage Institute collaborated with the University of California and the Westlands Water District to establish the first electronic water exchange in the state: *Water Links*. It works in a way similar to a bulletin board, where buyers and sellers can advertise and meet to agree on prices and volumes. *Water Links* was initially started for farmers in the Westland Districts, but by June 1997, already 23 other Districts wanted to join (Chakravorty and Zilberman, 1997).

Water trading along the Rio Grande and the west Texas plains

According to Jonish *et al.* (1996), there were 1 220 entities with active water rights along the Rio Grande in Texas in 1992. Irrigation districts and individual irrigators accounted for 88 per cent of water use (3 million acre-feet), while cities held roughly 8.8 per cent. The US irrigates roughly 900 000 acres in

the entire Rio Grande basin, compared to 1.1 million acres in Mexico. Agricultural users constitute the majority of water sellers in the region, while municipal users are usually the buyers. Lease (or term) prices per acre-foot range from US\$15-20 per year annually, compared to US\$300-700 per acre-foot for sales of permanent water rights. Analyses of transactions show that most spot transactions involve irrigation purchases, followed by mining and municipal use. The cost of spot mining transactions was 28 times greater than the price of irrigation or municipal spot purchases. The average spot transaction was 227 acre-feet, and totalled more than 87 849 acre-feet. Irrigation spot purchases predominantly occurred in the first and second quarters of the year. Municipal purchases mainly occurred in the last two quarters, after their surface water allocations had expired and irrigation demands declined.

Water markets have also developed in the West Texas Plains. All of the purchases made by the cities of Midland, Lubbock, Plainview, and Amarillo involved groundwater rights, with the majority of these purchases having been made in the last 22 years. Most of the purchases were found to be inventoried for use at a later date when current supplies become insufficient. Price per acre-foot of recoverable water for purchases after 1975 ranged between US\$9-13 (1984 prices). A critical analysis of the Texas Water Bank had found that its characteristics closely reflected a well-defined commodity market, unlike the California Drought Water Bank. However, it was found that hurdles still exist in making the Texas Water Bank a broad and effective market for water (summary from David Yoskowitz, 1997).

Australia: policy learning in nested institutions?

Inter-state water trading: Murray-Darling Basin Commission

As reported by Sturgess (1997), regulation of the Murray River began in 1917 with the establishment of the River Murray Commission. The Murray defines the political boundary between the Australian States of New South Wales and Victoria, and the basin also extends into Queensland and South Australia. The River Murray Waters Agreement, signed by the federal government and the state governments of New South Wales, Victoria, and South Australia, provided for the construction of storage facilities on the Upper Murray, in order to facilitate navigation and irrigation.

The River Murray Waters Agreement has been amended from time to time and in 1983, the River Murray Commission's brief was extended to include both environmental and recreational uses. In 1985, the geographic limits of the River Murray Commission were expanded to facilitate environmental and resource planning across the entire Murray-Darling Basin, and the Murray-Darling Basin Commission (MDBC), which now includes the government of the state of Queensland, was established in 1988. This has created some tensions between the broader environmental planning roles of the Commission and the narrower interests of those parties concerned with the regulation of the Murray River itself.

The Murray-Darling Basin Agreement is essentially a contract for capacity-sharing among the three States involved. The Commission does not own the water in the reservoirs, but rather operates the assets in accordance with the agreement, distributing water that is under the ownership and control of the respective states. It is obliged to meet user demand for water supply at an agreed level of security and quality, but is not responsible for the way in which the water is used once it has been allocated. In normal conditions, this involves staged releases from the Commission's storage during summer and autumn (December to May). In winter and spring, the seasons of highest rainfall, there is usually a surplus of water.

Under the current agreement, South Australia is guaranteed specific quantities of water from the Murray River each month, with an additional sum being allocated for dilution and losses. South Australia is also entitled to all flows in excess of these amounts that cannot be retained within the various storage facilities upriver. New South Wales and Victoria have equal rights to the rest of the water in the various tributaries and storage reservoirs, with a right to take their entitlements at times of their choice throughout the season (subject to operational constraints). This allows each state to pursue separate risk strategies, reflecting their somewhat different industry profiles. Late in a typical dry season, New South Wales is normally close to exhausting its share, while Victoria still has entitlements. Opportunities therefore exist for inter-state sales of water, although until very recently, this option has not been available to irrigators. Nevertheless, irrigators in the Murray-Darling Basin do trade irrigation water, as illustrated in the following cases.

Irrigation water trading: Goulburn-Murray Region, Victoria

The Goulburn-Murray Region is situated within the Murray-Darling Basin in Victoria State. Transferable water entitlements were first introduced in this region in 1987. At that time, temporary (or annual) transfers were introduced in gravity irrigation districts and for private diversions of water. Revisions of the legislation provided permanent transferability through the Water Act 1989, Water Regulations 1991 and 1994 and by-law (G-MW By-law No. 4). Trading was initially only allowed in local areas (for example, for diversions from the same stream). Opportunities were significantly increased in 1994, when trading between diversion licenses and district water rights and trading between district irrigators from different areas was allowed (Earl and Flett, 1998).

Permanent water trading has constantly increased since its introduction in 1991. Annual volumes have been around 10 000 megalitres a year between 1994 and 1996. Water is generally moving from sheep and cattle grazing enterprises to dairy farming ones. Research has shown that the majority of water being purchased is going to secure supplies, not to expand irrigation activities. Sellers of water rights are likewise not reducing irrigation activities, but are instead either selling unused rights or reducing their own security of supply.

Temporary water trading is strongly influenced by climatic conditions. In the years between 1989-94, allocations were unrestricted and there were generally good spring rains. Buyers of water rights were thus not looking for additional water, but cheaper water. Sellers were the recipients of prepaid water deliveries. They were selling at about half the normal price. In the 1994-95 trading year, a cap on maximum sales allocations was introduced. A very dry spring resulted in substantial temporary trading as buyers were seeking entitlements to increase their allocations. In subsequent years, the amount of temporary trade has been lower, since there have been better spring rains.

The Murray-Darling basin cap was introduced by the Murray-Darling Basin Ministerial Council in June 1995. It capped the extraction of water from the basin at 1993-94 levels of development. It allowed water use to fluctuate in accordance with climatic conditions, but clearly aimed to stop the long-term increase in water use. The authorities of the State of Victoria restricted temporary water trading opportunities in order to limit the uptake of unused entitlements. This was done in order to address two shortcomings: one is that actual water use has often been much less than the annual volume allocated to users. Entitlements or sales allocations were to be reduced to the lower level of actual use – a measure in line with the objectives of the Murray-Darling basin cap. The second problem is that entitlements have not been properly defined, and are often not compatible with each other.

Prior to the Murray-Darling basin cap in June 1995, Victoria had started a bulk entitlement conversion process. The aim was to improve the definition of the rights and obligations of water supply and power authorities in their management of water resources. The current rights to water were converted into clearly-defined bulk entitlements. Furthermore, a clear statement of how water is to be shared in droughts was developed. Irrigation authority entitlements were divided into two parts. The first is an aggregation of customer rights – these are not tradable by the authority. The second part is rights to water which is lost through delivery system leakage. These are potentially tradable by the authority. The bulk entitlement process also defines environmental provisions, including operating rules aimed at protecting and improving environmental values.

The Goulbourn-Murray system is planned to undergo some major improvements in the near future. As part of the bulk entitlement process, it has been proposed to:

- Clearly define entitlements or “entitlement products”. The aim is to establish one high security water entitlement product for each major supply system. The product would be fully tradable.
- Discontinue any special treatment to any user group. All users should be restricted in the same way during droughts.
- Make sure the security of the entitlement is independent of the authority.
- Introduce a second, lower-level, security entitlement. This should replace the sales and allocation entitlement products currently being used. This would provide an opportunity for users to adjust the proportion of high and medium security titles in a way appropriate to their activities.

Earl and Flett (1998) conclude that the market for water trading is working, but that it still has significant problems to overcome. The introduction of the Murray-Darling basin cap has significantly influenced entitlements and their values. The hydrology of the major systems is said to be well understood, and the institutional and regulatory framework required for long-term entitlement management to be basically established. The major challenge for the near future will be the change to the new two-product system at the farmer level. Overall “a realistic and compelling vision for the establishment of a full property rights regime and an effective water trading market in the Goulbourn-Murray Region has been developed”.

Irrigation water trading: Murrumbidgee, Murray-Darling Basin, NSW

The Murrumbidgee Irrigation Area (MIA) in the State of New South Wales (NSW) is one of the oldest irrigation areas in Australia, officially opened in 1912. Farmers mainly use their land for rice and wheat production, but horticulture is also common (mainly citrus and wine). In the 1980s, measures had to be undertaken to ensure that present and future water demands could be met. Losses caused by ageing water delivery infrastructure and increasing demands had led to this situation. Traditionally, water rights were tied to a specific parcel of land. The only way to obtain additional water rights was therefore to buy land. In 1982, water entitlements were issued, based on the total area of irrigable land at that time, and on future planting intentions. Each horticulture farm received entitlements of 12 megalitres per hectare. So-called “high security annual entitlements” were made available to farms at the beginning of the irrigation season, intending to guarantee that farms received 100 per cent of their entitlement in 97 per cent of years (Pigram, 1993; Samaranayaka *et al.*, 1998).

Initially, the scheme allowed water entitlements to only be traded on an annual basis. Later, multiple-year leases were introduced, and from 1989 onwards, permanent transfers as well. In 1992, the MIA Council of Horticultural Associations established a voluntary water pool. The arrangement allows farms to sell excess water to farms inside and outside the region. In 1997, there were no specific restrictions affecting the trading of water rights for horticulture farms in the region.

In 1997, a total of 22 horticulture farms were surveyed (Samaranayaka *et al.*, 1998), which used only about 62 per cent of their water allocation. This low usage was due to new irrigation technologies, and to farms abandoning (or postponing) expansion plans for which they had already received entitlements. About 90 per cent of farms sold excess entitlements, equal to 33 per cent of total titles issued. The revenue from sales was A\$ 1 447 on average – equal to 2 per cent of farm cash income. Some farms decided not to sell part or all of their excess entitlements because they felt they may need them, or because they were simply not interested in selling at that price. The price for water was A\$ 14 per megalitre on average, but prices as high as A\$ 22 were also recorded. The amount traded in the Murrumbidgee Irrigation Area (106 megalitres) was by far the largest of the four case study regions surveyed in 1997.

Irrigation water trading: New South Wales Murray Region

The Southern Riverina Districts Council (SRIDC) in the Murray Region of New South Wales set up a water exchange in September 1997. SRIDC is the representative body of the 2 500 shareholders of the Murray Irrigation Limited, which operate 1 800 farms. Murray Irrigation Limited manages the local distribution of water for its shareholders. Its entitlement is 1.52 million megalitres – about 67 per cent of the total New South Wales Murray entitlements. SRIDC and Murray Irrigation Limited have a close relationship, which has greatly assisted the setting up of the SRIDC exchange.

Two different types of transfers exist in the region: annual (or temporary) and permanent. Annual transfers were introduced in 1982, after a drought emergency and severe water shortages. Until 1992, these were limited to transfers between property owners belonging to Murray Irrigation Limited. In 1992, inter-valley transfers were also introduced. Permanent transfers within the districts then commenced in 1993. The SRIDC exchange was established in 1997, initially in order to trade annual water entitlements. Up to this point, the majority of irrigators had little or no experience with water trading. The aim of the exchange was to maximise the volume of water that can be traded. The mechanism used for trading was copied from the dairy industry's milk trade scheme: the SRIDC receives tenders from buyers and

sellers. Each of them then lists the amount of water needed (or available) for sale, and the price at which they are willing to trade. The SRIDC calculates the market clearing price in accordance with supply and demand. The water exchanges take place every two weeks, tender boxes are opened in the meantime.

The first exchange was held in October 1997. Access was restricted to Murray Irrigation Limited members. Demand was turning out to be much higher than supply, with over 6 000 megalitres of buyer tenders, and only 1 000 megalitres offered. When the market clearing price was set to A\$ 45, only 500 megalitres were traded. The price went up during the next exchanges, peaking at about A\$ 67. A maximum of only 1 000 megalitres was traded due to a shortage of sellers and the resistance of buyers to pay high prices. The short experience so far has shown the volatile nature of the temporary water market. While at the end of 1996-97 irrigation season the price for water would have been zero, it peaked in January 1998 at A\$ 67. One problem is the small amount of water being traded. This may be due to irrigators' inexperience with trading.

Chile: effects of radical innovation

Chile has been exceptional in encouraging the use of water markets in water resource management. To this effect, the legal framework for water management was changed radically (see earlier discussion). In a 1995 World Bank study by Hearne and Easter, four cases of water rights trading (the upper Maipo, the Elqui, the Limarí and the Azapa valley) were examined. But only two of these showed significant amounts of trading: in the Elqui valley, there were significant inter-sectoral transfers, although transactions were infrequent and constrained by the lack of adjustable gates. In the Limari valley, transactions were relatively frequent (Hearne and Easter, 1995: ix; see also Rosegrant and Binswanger, 1994; Bauer, 1997).

The Maipo Valley (upstream), Chile

The upper Maipo river rises in the Andes mountains and flows through semi-arid agricultural land to the community of Isla del Maipo near Santiago de Chile (Hearne and Easter, 1995: 44 FF). In addition to irrigation water, the upstream section supplies water for 4.5 million inhabitants in the Santiago area (and provides for various non-consumptive uses, such as hydropower or recreation). The first section of the Maipo is divided into 8 133 shares, each representing at least 8 litres per second at 85 per cent of the time, with a relatively high security during the summer months of December and January.

The water suppliers for most of Santiago's population [the Metropolitan Sanitation Works Company (EMOS)] owns 1 369 shares, representing a total of 5.2 m³/s at 95 per cent security required for public water supply. Because EMOS owns storage facilities, the water rights held in fact provide 10 m³/s at 95 per cent security. EMOS has estimated that it needs another 3 000 shares by the year 2020, but between 1990 and 1993 had only succeeded in buying 33 shares at an average of US\$10 000 each (Hearne and Easter, 1995: 45).⁹ There are water users in the valley with abundant allocations they might sell, notably the irrigators organised in the Sociedad del Canal del Maipo (SCM), which delivers water for free and covers its costs with revenues from a hydroelectric plant.

There are several reasons why three farmers are reluctant to sell water-use rights. One is the public knowledge of high prices paid in the Río Mapocho valley. The Potable Water Company *Lo Castillo* drove up prices in this valley by offering lower water service prices for customers which turn over any water rights associated with their land. Some prospective clients bought rights before asking *Lo Castillo* for service, thus driving up prices.

Among the reasons for the lack of trading in water rights is the cost of physically changing canal flows through fixed flow dividers. These changes make marginal water transfers quite expensive. However, temporary transfers of water rights (*i.e.* trading of water rather than water rights) is reported to have taken place among neighbouring farmers. Such trades involve a yearly use of a share of the river, with the buyer having to pay the annual fee to a water user association (WUA) such as the SCM (Hearne and Easter, 1995: 47). It can be assumed that "informal trading" among farmers, not involving a WUA is much more common than formal trades, where trades in whole annual shares are notified to the relevant WUA.

Given the trading of water among irrigators and the potential for trading water rights (shares in river water), it can be assumed that EMOS will be able to secure more water in the future, if demand in the urban area of Santiago de Chile increases.

The Limari Valley, Chile

The market for water use rights in the Limari valley is described as being “active” by Hearne and Easter (1995: 53). In the period from 1992 to 1995, there has been a rapid accumulation of water rights in the hands of some large fruit exporters, mainly multinational enterprises. Small-scale producers have entered into marketing and financing contracts with these firms. The small firms agreed to pay their debts in grapes, but low prices and poor harvests have forced them to transfer land and water-use rights to the large fruit exporters. The result was a concentration of rights in the hands of a few, coupled with the loss of rights for traditional farmers.

In the Limari Valley case, water-rights trading has resulted in oligopoly, with potential longer-term effects on the efficient allocation of water, the balance of (economic) power, and the potential for regional development, economic self-determination and diversification, and regional employment and welfare.

One-off trade in hydro power and irrigation: Maule River Basin, Chile

The Maule River in south-central Chile is the site of the country's first clash between irrigation and hydroelectricity interests, following the establishment of private tradable water rights in Chile. In 1983, the *Direccion General de Aguas* (DGA) granted non-consumptive rights to *Empresa Nacional de Electricidad* (ENDESA), the formerly state-owned power company. ENDESA was granted rights to build two dams in the Andean foothills – at *Colbun* and *Machicura*. The grants defined a guaranteed minimum flow to be released to irrigators downstream of the dams. When ENDESA was privatised in the late 1980s, *Colbun* and *Machicura* remained in a separate company. In 1984, ENDESA was granted even more rights, this time to build another dam up-stream called the *Pehueche Dam*.

Construction of the *Pehueche Dam* was completed in 1990, coinciding with a third consecutive year of drought in central Chile. The company closed the dam gates to fill their empty reservoir, and irrigators downstream found much of their water suddenly cut off. When the *Maule River Vigilance Committee* asked Pehuenche to release water, the company refused and the Committee went to court. The Committee argued that Pehuenche was consuming water by filling their reservoir, even though they only had non-consumptive rights to their water. The power company argued that the farmers should either wait until autumn or buy water rights from *Colbun*. The *Appellate Court in Santiago* and the *Supreme Court* both ruled in favour of the irrigators, stating that their property was threatened. The DGA followed the court's decision and ordered the Pehuenche to open its gates. Nevertheless, Pehuenche succeeded in filling their reservoir by taking two measures:

- They received water that ENDESA released from two upstream reservoirs.
- They agreed to pay *Colbun* to release water temporarily to irrigators. However, the Committee opposed this transfer, arguing that ENDESA still had no right to fill the reservoir, thereby using the water for “consumptive” purposes. The DGA rejected the Committee's protest, on the grounds that the irrigators' rights were no longer being injured.

Subsequently, the *Supreme Court* convinced all parties (ENDESA, *Pehuenche* and the *Vigilance Committee*) to drop their law suits, without resolution of the issues. All sides retained the right to argue their case and to adopt the same positions again in the future. Everybody agreed to do so, which was mainly because it was too late – the reservoir was already full. And the case of the Maule River was only the beginning of this type of conflict in Chile. The following summer, the *Vigilance Committee* went to court again over the first dam on the Bío Bío River.

The conflict between hydroelectric power generation and irrigation in the Maule River clearly outlines the problems that can occur when consumptive and non-consumptive water rights compete, and when conflict resolution mechanisms operate slowly in comparison to hydrological developments.

Equivalent instruments to tradable water abstraction rights

General remarks on instruments “equivalent to” tradable rights

All the case studies on tradable water abstraction rights focus on water for irrigation, with navigation (Murray), hydropower (Snake River), or urban use (Northern Colorado Water Conservancy District) being of secondary importance. Permanent, “one-off” water transfers through trading from typically low-value applications in agriculture to high-value uses in cities are also reported, but appear to be rare. While the reasons for this are not clear, it is worth considering alternative instruments with equivalent effects. One such alternative instrument has been applied in the German City State of Hamburg, in order to bring about a re-allocation of (unused) water rights.

Water tax case: Hamburg, Germany

The German Free and Hanseatic City State of Hamburg has enforced a groundwater licence fee since 1 July 1989.¹⁰ The fee is levied on the basis of abstraction rights held by water users, namely the maximum quantity of water a user is licensed to pump per year. In case the licence is exceeded, the fee is set on the basis of the actual use. Exemptions are made for some abstractions that do not require a licence according to the federal Water Management Act (*i.e.* for small quantities (< 10 000 m³/a), and for water used for heat pumps). A fee schedule is applied in the case of quantities between 10 000 and 20 000 m³/a. Public water suppliers pay a reduced rate and, in addition, benefit from a blanket reduction of 65 per cent.

Hamburg distinguishes between public water suppliers and all other users, and between “good” groundwater (*i.e.* from deep and relatively well-protected aquifers) and water from aquifers near the surface (≤ 35 m of depth) that might be polluted or affected by the tidal river Elbe (salt intrusion). When the fee was adopted, a first increase in the rates was already included. A second increase took effect on 1 January 1994 (Table 1). As of 1994, the blanket reduction for water suppliers was lowered to 60 per cent.

Table 1. **Groundwater Abstraction Fee Rates in Hamburg**

Groundwater Quality	“Good Groundwater” [in DM per m ³]			High-chloride groundwater near the surface” [in DM per m ³]		
	1989-90	1990-93	1994	1989-90	1990-93	1994
Public water supply	0.05	0.10	0.11	0.00	0.05	0.05
Other uses	0.10	0.15	0.17	0.05	0.10	0.10

Source: Kraemer, 1995.

The fee provided water users with a stimulus (especially since the possibility of future increases in the tax rates was clear) to review their water needs and their holdings of water rights, and to consider the potential for water savings and substitution. The “environmental effect” the fee was designed to achieve was not primarily a reduction in water *use*, but a reduction in the water *rights* held by users. Between 1989 and the end of 1993, more than one-third of all water rights (103.8 million m³) were renounced (and thus made available for re-allocation for public use).

The legislation was subsequently revised on the basis of these results. The focus appears now to be shifting towards setting incentives to save water (rather than to retire water rights), both directly through rate increases, and indirectly through the decrease in the blanket reduction allowed for public water suppliers.

Summary remarks on tradable water abstraction rights

Experience with water banking in the US highlights the importance of water storage capacities. While it is intuitively clear that storage is a necessary component for banking, it must also be kept in mind that storing meaningful quantities of water requires large-scale investments in physical assets (dams). Furthermore, one large dam (or a cluster of dams within close distance of each another) is a precondition for water trading in significant volumes. Trading is spatially restricted to areas downstream of such dams. Dams have usually been built with public subsidies and the capital costs associated with these assets are normally not borne by the water users. Operational costs tend to be low (gravitational distribution) and can sometimes be cross-subsidised from other services, such as hydropower. The Northern Colorado Water Conservancy District shows that a stable water supply provided to an area of variable water need can be a good background for the developing water trading regimes, especially where significant environmental concerns do not arise, or can be ignored. Because of the political (and financial) difficulties associated with building dams today, water banking based on dam storage would appear to be limited in scope to those areas where dams already exist.

An obvious alternative to water storage behind dams is storage in (groundwater) aquifers which can provide sufficient storage capacity for several years of water demand. The investment costs associated with the development of aquifers are much smaller than those for dams. However, aquifer storage normally entails higher operational costs, mostly for pumping. Where aquifers are not well-understood, conflicts can arise because the negative consequences of abstraction (and recharge) on other water users can take years to become visible. This is true also of environmental costs, such as a loss of water-dependent ecosystems on the land.

Little is known about the details of conflict resolution mechanisms in the various trading regimes. It appears that this function is usually located either in a public authority (licensing authority) or in a quasi-judicial committee that operates within the governance structure of water users associations or irrigators associations. A prominent example of this is the Tribunal da Águas in Valencia, Spain.

The most "lively" trading appears to always take place within the agricultural sector, with transfers from agriculture to other sectors (hydropower or municipal use) being relatively rare. Nevertheless, such inter-sectoral transfers are perhaps the most significant in economic terms as they can be expected to provide important added value. The switch from traditional regimes for allocating water to trading, to judge from the few cases studied here, appears to be facilitated by two things:

- A long-established and institutionally-stable regime, in which reciprocal knowledge, trust, and a system of conflict resolution has been built up (a "well-run" water users association), especially those where membership is largely confined to one sector (*cf.* Ostrom, 1990).¹¹
- A crisis or emergency, such as a drought, during which the members or participants of the regime need to co-operate in order to stabilise the economic basis of a whole region (common interests make a blocking position morally untenable), or in order to guarantee the survival of the allocation regime on which they depend.

The introduction of trading can highlight the degree to which water has been over-allocated, where more water rights have been created through traditional licensing systems than the amount of water which can be supplied. One reason for this can be a lack of understanding of the hydrological system; another can be a lack of co-ordination among different authorities. Where water rights are not actually being used (or exploited), the impression can arise that there is more water available for appropriation. The Murray-Darling basin cap was one way to address this problem; the water abstraction tax introduced in Hamburg fulfilled the same function.

All examples of water trading presented above appear to rely on traditional transaction technologies, with the notable exception of Water Links, the electronic water exchange in California. It remains to be seen if this is the harbinger of a wider use of modern information technology, and if the reduced communication costs can make water markets more attractive, compared to traditional allocation systems.

The scarce evidence on the price of water trades in Australia suggests that prices for *permanent* water rights may be 100 times higher than prices for *temporary* rights to use water. New South Wales, for instance, provides for trades of water allocations of varying security, which allows water entitlements to be traded

to suit the specific requirements of agriculture or other activities. Most trades, however (both by number and by volume), relate to temporary transfers (James, 1997). This observation appears to be consistent with the experience in the US, where somewhat lower price differentials between temporary and permanent rights are reported.

Tradable water pollution rights

In contrast to trading water abstraction rights, which can be expressed rather simply in volumetric terms, trading in permits to pollute water resources has to cope with a much higher degree of complexity. Water can be polluted by a number of substances (or classes of substances) which have very distinct effects on water-based ecosystems. The presence of two or more pollutants at the same time can lead to synergies, both positive and negative. Furthermore, most sources of pollution contribute not only one substance that is dangerous to the water environment, but several. Water pollution permits can contain long lists of substances and parameters that have to be observed.

It is not surprising, therefore, that there are no examples of trading systems in water pollution as such, but only in relation to individual substances or parameters (salt, organic oxygen-depleting substances, and nutrients). For these three parameters, examples of tradable pollution permits are presented below. These examples relate to pollution loads which have long-distance effects, rather than concentrations which might have acute effects. In relation to water pollution much more than with water abstraction, it is the precise location of a discharge that determines the environmental consequences.

Salinity trading

Salt pollution in freshwater systems affects the suitability of water for many purposes, such as irrigation or drinking water supply. It can also have significant environmental effects on relatively sensitive ecosystems that rely on brackish water, such as in estuaries. The concentration of salt ions is relatively easy to assess by measuring the electrical conductivity of water. Conductivity is not a specific indicator of toxicity, nor it is a suitable proxy for dangerous substances. It is however, a useful parameter when measuring the concentration of salts, the nature and origins of which are well understood.

Salt pollution usually originates in the mining industry (salt mines, but also mine water from coal mines, for instance) or the energy sector, where cooling by water evaporation leaves saline residues. Salt pollution can also occur naturally as a result of erosion or natural dissolution of salt deposits. Where salt concentrations (rather than loads) trigger problems, dilution by fresh water can provide a (temporary) solution.

Although salt pollution rarely reaches levels where corrective action has to be taken, the examples of where it does can be instructive. Chloride pollution of the international river Rhine, for instance, provided the stimulus for developing the multilateral system of the riparian states for managing economic and environmental aspects of the river.

The most prominent examples of salinity trading come from Australia, with the inter-state trading case in the Murray-Darling Basin, and the more market-oriented approach in the Hunter River in the State of New South Wales. In both cases, the concern is for reducing and “managing” salt pollution to reduce harm. This is in contrast to the German case of the river Weser, where the right to pollute a tributary with effluents from salt mining was effectively “bought off”.

Inter-state salinity trading case: Murray-Darling Basin, Australia

A system of tradable rights for discharges of salt into the river came into force in 1992 as part of the Murray-Darling Basin Salinity and Drainage Strategy (James, 1997: Section 10.3). The regimes are administered by the Murray-Darling Basin Commission on behalf of the states of New South Wales, Victoria and South Australia (see above). The salt pollution rights are not freely-traded by industries or individuals, but are exchanged between the governments of the participating states within a constrained strategic framework.

In this regime, “salt credits” can be earned by investing in capital works to manage salt entering the river system, or to enhance river flow. In 1997, New South Wales had a credit of 6.15 units, and Victoria a credit of 5.92 units. South Australia and the Commonwealth of Australia have also earned credits, but these are unlikely to be used to offset debits. In South Australia, salinity has been reduced by 50 EC units.

Credits are tradable between the States, but are generally applied within each State to offset debits from drainage entering the river system. New South Wales is introducing its own system of tradable salt pollution permits for all dischargers contributing to salt pollution in the Murray-Darling and for all water users diverting flows from the Darling Basin. The system is separate from the Murray-Darling Basin Salinity and Drainage Strategy, and will involve only a small number of salt credits, so that trading is not expected to be significant (James, 1997: Section 10.3).

Salt pollution trading case: Hunter River, NSW, Australia

Water in the Hunter River in the State of New South Wales (NSW) and many of its tributaries have naturally high salt concentrations (James, 1997: Section 10.2; EPA NSW 1994). The salinity problem is exacerbated by discharges of saline waters to the river by coal mines, power stations, and other industries, as well as by drainage systems in the river basin. Salinity becomes a problem mainly during low flow. During high flow, the river is less sensitive to discharges, and irrigation usually ceases under these conditions. An annual loss of A\$ 10 000 for each unit increase in salinity, measured in terms of electrical conductivity (EC), has been estimated for the river basin, resulting mainly from reductions in agricultural yield and increased costs of water supply and treatment.

The Environmental Protection Agency (EPA) of New South Wales granted salt discharge licenses to 11 coal mines. The licences specified a limit on the maximum allowable increase in conductivity in the river of 700 EC units after the discharge, as well as a limit on the maximum allowable increase in conductivity of 40 EC units caused by each discharge. Pacific Power, which operates two hydropower stations in the Hunter region, was permitted to discharge up to 700 megalitres per day of relatively saline water from Lake Liddell to the Hunter River, when the river flow was less than 2 000 megalitres per day at a specific measuring point, on condition that the salinity level in the Hunter did not rise above 700 EC units. When the flow exceeded 2 000 megalitres per day, Pacific Power was permitted to discharge up to 700 megalitres per day with no salinity restrictions.

The EPA then decided to implement a system of tradable salt discharge credits on a trial basis in 1995, which was developed from the existing licensing system. The scheme was designed to eliminate discharges during low flow periods and to limit total discharges during high flows to meet target salt levels. The scheme has been initially limited to coal mines and Pacific Power. It is expected that other sources will be brought into the scheme in the longer term.

The limit on the maximum allowable increase in conductivity caused by an individual discharge (previously 40 EC) during high flow was subsequently removed, and replaced by salt concentration objectives for the receiving water. Henceforth, each discharger would be allowed a specified percentage of the total allowable salt load. During low flow, a stricter “interim receiving water conductivity threshold” for saline discharges in the upper reaches of the river was implemented, and a new threshold of 500 EC units (previously 700 EC) was established. Within this regime, any discharger can trade pollution credits with any other source(s). Because the total allowable load is calculated in relation to conductivity levels, it is also related to river flows.

For the implementation of the regime, the river is divided into three sectors – upper, middle and lower. Each licence holder is associated with a relevant sector. Water flowing in the river is divided into “blocks”, with each block consisting of the body of water that passes a specific monitoring point (Singleton) during any 24-hour period. Each credit entitles the holder to discharge 0.1 per cent of the total allowable discharge for each block. After determining the total credits available, a fixed proportion is allocated to individual mines and to Pacific Power. The EPA retained 20 per cent of the credits as an environmental buffer. A bonus entitlement may be earned by dischargers, if they purchase a special release of water from the upstream storage operated by the Department of Land and Water Conservation.

Trades must be for whole credits and for whole blocks. Holders of credits are free to trade with other credit holders, but all trades must be registered with a Credit Register and a River Register. The EPA reserves the right to refuse approval of a trade if it detracts from the effective environmental operation of the scheme.

The number of credits allocated to each discharger was determined by means of a “merit formula”, which takes into account the environmental management of each discharger, its output and use, as well as operating difficulties represented by gross mine water that needs to be discharged. The environmental performance “score” is derived from the water management plan, currently in effect, and based on the saline water storage capacity, the level of discharge and receiving water monitoring, the discharge control capability, and the recent record of compliance with licence conditions.

During discharge, monitoring is required at discharge points for volume and conductivity. Real time monitoring must also be conducted upstream and downstream of the discharge point in the Hunter River or its tributaries. The permit holder is responsible for these functions. All monitoring data must be submitted to the EPA. Permit holders must prepare and submit quarterly reports to the EPA, and must keep their records for two years. The EPA conducts regular audits of the scheme.

Gilligan *et al.* (1996) document initial experience with the scheme. In general, conductivity levels in the river remained within the target limits. Due to extended dry periods, it was decided to test the scheme for another year. In the second year, only about 35 per cent of the total potential discharge opportunity was used, and the stored waste water was discharged during a flood flow period.

Thus far, several dischargers have offered to sell their entitlements, but only one trade has taken place. This is because dry conditions have created water shortages, rather than a need to discharge. In addition, due to uncertainty about long-term needs, as well as about the value of entitlements and arrangements for longer term allocations, most mines are not prepared to trade. There have been some difficulties with data collection and monitoring, and operation of the system is not yet fully automated. The EPA considers that the scheme is better than the previous system of licences, and expects it to operate more fully after 1997 (James, 1997: Section 10.2).

Buying off salt pollution rights in the River Weser, Germany

The largest river basin in Germany that is not international is that of the river Weser with its main tributaries Fulda, Werra and Aller, and Thuringia, Hesse, Saxony-Anhalt, North Rhine-Westphalia, Lower Saxony and Bremen as riparian *Länder* (or States in the Federal Republic of Germany). The Weser used to be an “international” river shared between the Federal Republic of Germany and the German Democratic Republic. At that time, it was the subject of negotiations between these two countries over the issue of salt pollution. Since German unification, however, it has become a good example of river basin co-ordination within Germany.

In 1992, the riparian *Länder*, building on earlier co-operation among the Western *Länder* involved, concluded the Administrative Agreement on the Creation of the Working Committee for the Protection of the Weser (*Arbeitsgemeinschaft zur Reinhaltung der Weser* – ARGE Weser). The *Länder* agreed that their Ministries of the Environment, as the supreme water management authorities, should co-ordinate water management activities and the implementation of water law, especially in relation to pollution control. By agreement, the ARGE Weser decides on pollution limits, water quality objectives, emission standards and issues recommendations concerning water management plans, as well as on monitoring programmes for water quality and quantity. Each *Land* is responsible for the collection of data and information on its territory that may be needed by the jointly-established Weser Water Quality Unit. The latter may be asked to collect these on a *Land*'s behalf in which case the *Land* in question has to bear the cost. In this way, the river basin approach is combined with the sovereign responsibility of the *Länder*. Other than sharing the operational budgets of joint river basin institutions, there are no financial transfers among the *Länder*, unless there are separate agreements.

Another agreement for controlling salt pollution of the Werra and the Weser has been concluded in addition to the general co-operation agreement described above. The Administrative Agreement on the Granting of Subsidies by the *Bund*, the Federal Republic of Germany, and the *Länder* for Measures to

Reduce Salt Pollution in Werra and Weser of 1992, is highly specific in that it concerns changes to the Thuringian salt mining industry following German unification. It is also temporary. It is a good example, however, of the way in which the *Länder* may effect financial transfers in order to finance investment on the basis of priorities established at the river basin level, and to buy off or “trade in” perceived rights to pollute.

The agreement provides for 79.52 per cent subsidies on the cost of measures to reduce salt pollution of the Werra, to an annual average of 40 kg/s. The volume of subsidies is DM 116.5 million for a total investment of DM 146.5 million over five years. The costs are shared between the *Bund* (50 per cent) and the riparian *Länder*, except for Saxony Anhalt (10 per cent each). In addition, the *Bund* pays half of Thuringia's share, so that its total contribution is 55 per cent. The administration of the subsidies is handled by Thuringia following a detailed technical concept and a time schedule; annual reports including audit reports are sent to all contributors.

Summary remarks on salinity trading

Experience with salinity trading so far has been rather limited, and the examples show such trading to be highly intertwined with traditional environmental management systems. In all cases, the number of participants (traders) is small, trades are rare or restricted to times of low flow, and are highly specific.

The establishment of trading regimes in this case is facilitated by the fact that salt pollution can be measured relatively easily, and on a continuous basis. In effect, monitoring the behaviour of the market participants is relatively cheap. Nevertheless, the examples given above may be useful in designing tradable permit systems for other high profile pollutants, for which continuous analysis may be possible.

Trading of organic pollution rights

A more challenging aspect of trading in water pollution permits is presented by organic pollution. Such pollution consists of a multitude of different substances containing carbon, any one of which may be present at concentrations below critical levels. Such substances can originate from human wastes (*e.g.* sewage), but also in industrial effluent (*e.g.* food and beverage industries), as well as from rainwater run-off. Organic pollution can be controlled (but not fully eliminated) by treatment, and the ability to release such pollution into recipient water bodies typically has a significant impact on the cost of treatment.

(Almost) all organic pollution is naturally degraded or “metabolised” by biological mechanisms in natural water systems, consuming oxygen in the process. When oxygen is consumed, the level of oxygen dissolved in water decreases. In extreme cases, especially during periods of low flow or in warm water, the water can be deprived of oxygen to the point that fish and other life in rivers and lakes die. This is not a slow process but can often “hit” a river as a consequence of a single pollution incident, such as storm water over-flow being discharged. It is therefore vital to control overall pollution with oxygen-consuming substances, and to ensure sufficient levels of dissolved oxygen in waters.

As illustrated in the following case study, the experience so far has not been very encouraging. However, it appears that this is largely due to a lack of understanding about abatement technologies (and their costs) when the permit trading systems was established. It can even be said that the lack of trading in the Fox River case is one indication that effective innovation incentives have been created as the result of the trading system.

Organic “point source” trading case: Fox River (Wisconsin)

In the US, the State of Wisconsin established the legislative basis for what may be the first operational water-pollution permit market. The Wisconsin Department of Natural Resources approved the trading of rights to discharge into the Fox River as early as 1981. Point sources of water pollution can trade rights to discharge wastes that increase biological oxygen demand (BOD). The Wisconsin programme was aimed at providing flexibility for point sources, such as paper mills and municipal wastewater treatment plants, in meeting state water quality standards. Sources that reduce discharges containing BOD below

permitted amounts are allowed to sell the excess reductions to other sources. The pulp and paper mill effluent guidelines suggested that substantial costs would be incurred to meet the stringent limits required to meet the water quality standards because of the large numbers of dischargers concentrated in a few miles of the state streams. Analysis showed potential cost savings from trading of about US\$7 million. In actual fact, the effluent guidelines now appear to have far overstated the needed expenditures. Costs in addition to those needed to meet the national point source requirements were not incurred and, with one limited exception, no trades have yet occurred (Carlin, 1992; see also O'Neil, 1983; O'Neil *et al.*, 1983).

Under this system, permission to trade will only be granted if the discharger meets certain preconditions:

- the plant acquiring the rights must be new or growing, or at least unable to meet the discharge limit despite working efficiently;
- every firm has to prove the increase in water pollution is necessary;
- traded rights must have a life of at least one year, but no longer than the seller's discharge permit expiration date.

The Fox River programme includes 21 parties: five mills and two towns in each of the three administrative segments into which the river stretch was divided into. In a 1992 EPA Report, Carlin judged the trading to have been disappointing (Carlin, 1992, pp. 6-29). He stated three reasons for the limited activity:

- Dischargers developed a variety of compliance alternatives not contemplated when the regulations were drafted.
- There ... remain questions about the vulnerability of the programme to legal challenge, since the *Clean Water Act* does not explicitly authorise trading...
- The State imposed severe restrictions on the ability of sources to trade.

The amount of trading in the Fox River has not only been disappointing, in an even more extreme judgement of the system (Holm-Müller, 1995: 111), the programme was judged a complete failure.

Trading of nutrient pollution rights

Nutrients (*i.e.* nitrogen and phosphorous) are not in themselves dangerous to water or water-based ecosystems. In fact, they are necessary components of plant life. That is why they are applied as fertilisers to enhance plant growth. They also appear in domestic sewage in significant concentrations and loads. In water bodies, they stimulate plant (algal) growth, which consumes oxygen and can thus lead to fish kills.

In many respects, the logic of nutrients trading follows that of trading in organic pollution permits. However, since agriculture is an important source of the former, there is scope here for trades between point and non-point (or diffuse) sources. In the following discussion, one example is presented relating to the Hawkesbury-Nepean River, in New South Wales (Australia). This example is one where "trades" (in the form of intra-firm allocations) affect point sources only. The next example, of the Tar-Pamlico Basin in the US state of North Carolina, is also primarily one of point source trading, but does offer the prospect of trading between point and non-point sources. Such trading is more pronounced in the case of Lake Dillon, and might become so in the future in the Cherry Creek Basin in Colorado (US) case study, discussed at the end of the Section.

Nutrients trading case: Hawkesbury-Nepean River (NSW)

The Hawkesbury-Nepean river system in Australia is experiencing significant environmental stress as a result of high phosphorus and nitrogen loads (James, 1997: Section 10.4). Sewage treatment plants are a major source of nutrients, although discharges occur also from other point sources and non-point sources in the basin. In response, the New South Wales EPA has developed a "bubble licence" regime, incorporating a number of sewage treatment plants within the Hawkesbury-Nepean river system, all belonging to the water services operator for the Sydney area (Sydney Water). This bubble licence is the first of its kind in water quality management in Australia.

Through the bubble licence, the regulator controls the aggregate load generated from multiple sources within the bubble, rather than controlling emissions or effluents from individual sources. The advantage is that the operator has the flexibility to search for cost-effective solutions, while ensuring that the overall discharge targets are achieved.

Licence conditions associated with the scheme have been applied to Sydney Water's licences from 1 July 1996 for the 1996-97 licence period. The bubble licence is underpinned by a strong regulatory framework. The aim of the scheme is to reduce high nutrient loads within the creeks and the main stem of the river. It is estimated that the three sewage treatment plants currently contribute around 60 per cent of the phosphorus load, and around 75 per cent of the nitrogen load in the section of the Hawkesbury River at a particular measuring point.

The aggregate load limits for phosphorus and nitrogen are to be reduced over an eight-year period. Load targets specified for the year 2004 should result in a reduction of 83 per cent in phosphorus loads, and around 50 per cent in nitrogen loads. Interim targets should ensure continuous environmental improvement. While annual load limits will restrict the load discharged to the environment, additional regulatory control in the form of maximum concentrations are to be set for each sewage treatment plant to ensure that any operating difficulty at a particular plant is promptly recognised and rectified.

It is estimated that cost savings from the bubble licence scheme, compared to uniform discharge concentration limits, will be in the order of 10-20 per cent. The scheme is also expected to provide an incentive for Sydney Water to investigate alternatives to traditional sewage treatment technologies, which could result in additional savings in abatement costs (James, 1997: Section 10.4).

Nutrient trading: Tar-Pamlico River (North Carolina)

The US State of North Carolina has designated "nutrient sensitive waters", including limits on total phosphorous and nitrate discharges into such waters, among them the Tar-Pamlico Basin. In this case, Department of Environmental Health and Natural Resources in 1989 created a basin-wide bubble for nutrient pollution (nitrogen and phosphorus), which covers about a dozen sewage treatment plants and one industrial discharger, and which is administered by the Tar-Pamlico Basin Association as the broker for trades. Membership in the Association is voluntary, but if a discharger chooses not to join, it comes under the normal individual permit system. If the Association's members (jointly and severally) fail to meet the pollution targets in 1992 through 1995, charges would be imposed (Anderson and Snyder, 1997: 153-156).

The Tar-Pamlico scheme was initiated after a series of fish kills led local residents, fishermen, and environmental groups to push for State action to regulate nutrient discharges. The North Carolina Environmental Management Commission classified the Tar-Pamlico as "nutrient-sensitive waters". Upon completion of watershed modelling, new and lower aggregate effluent limits were established that would have to be met by the Association (Carlin, 1992).

The Association facilitates trade among member pollution dischargers, and with non-Member farmers who reduce field run-off and collectively reduce the discharge of phosphorous and nitrogen to the Pamlico estuary. Point sources accounted for only 15 per cent of the total nutrient load in the watershed, with the majority coming from agricultural and other non-point sources. Any member of the Association can reduce nutrients internally, trade within the group, or pay a fee of US\$56 per kilogram to a fund that goes toward non-point source reductions. The funds generated from effluent charges are then used to reduce nutrient loads from non-point sources. Farmers in the region are paid to adopt management practices that reduce nutrient runoff. The transactions among these point and non-point source polluters represent a distinguishing feature of the Association's procedures (Riggs and Yandle, 1997).¹² The "bubble" emission limit was 425 000 kilograms per year by the end of 1994 for the Association as a whole, but emissions have always been lower than this. The association is allowed to bank its excess abatement capacity for later use when plants expand. The overall discharge of nutrients into the basin has been reduced by 28 per cent, even though the average discharge of effluent has actually increased by 18 per cent.

An implicit system of marketable permits is also at work within the Association. Members of the Association have varying marginal nutrient abatement cost functions, leading to substantial gains from trade when members transact with one another (Riggs and Yandle, 1997).

Point/non-point nutrient trading: Lake Dillon (Colorado)

Non-point sources play a major role in water pollution. Virtually all areas in the US have water quality problems because of non-point sources, and half the States regard non-point sources as a significant cause of degraded water quality. One innovative approach taken in a new programme in Colorado is to tie non-point sources into existing pollution discharge permit systems (NWCCG, 1984; Kashmanian *et al.*, 1987; Elmore *et al.*, 1988; Carlin, 1992).

The Dillon Reservoir is one of the main tourist attractions of Summit County, as well as the source of 50 per cent of Denver's water supply. Surrounding the Dillon Reservoir is a recreational community that developed during the 1970s and early 1980s. Four municipal treatment plants, sixteen small treatment plants, one industrial plant and numerous non-point sources all discharge wastes into the reservoir. All of the point sources were subject to strict discharge limits. According to analysis prepared for the EPA, the municipal treatment plants were generating nutrient loads which were becoming difficult to manage. Human activities accounted for about one half of the total phosphorus load, and, of this, between 50 and 75 per cent came from non-point sources, with the rest coming from point sources. The municipal wastewater treatment facilities faced incremental control costs for phosphorous that were many times those of non-point sources, providing a rationale for trading. In effect, the fast growing phosphorus pollution of the reservoir generated the need for a watershed management programme, which was subsequently founded in 1984.

In the Dillon Water Quality Management Plan, the goal of limiting the total phosphorus load was set at 10 162 pounds. Every point discharger was then granted limited annual discharge rights and a growth margin to accommodate some development. The allowed point source dischargers to increase their discharges above a given limit by point/non-point source waste load allocation system investing in low-cost phosphorus removal from non-point sources. Due to the uncertainty related to the control of non-point sources credits for non-point pollutant reductions were only traded for point loads at a 2:1 ratio. This means that a point source had to reduce two tons from non-point sources (that existed before 1984) before it could increase its own discharge by one ton. Economically, such a trade can still be interesting for the discharger: the marginal cost for removal of one pound of phosphorus from a wastewater treatment plant is estimated US\$860, while the average cost of non-point source control is US\$119.

Until 1988, no trading was approved by the basin management authorities because critical loads were not exceeded. It was expected, however, that trading would eventually take place for two reasons:

- The waste load allocations for point sources are not sufficient for future growth.
- Even if cheaper reduction technologies for point sources are being developed, there is still a significant cost-saving potential in trading.

More recently, however, there have also been a few trades proposed between non-point sources. Because of a slowdown in growth in the region, as well as substantially-improved point source removal efficiencies, point sources have not yet faced a significant need to trade with non-point sources (Carlin, 1992).

Nutrient trading: Cherry Creek (Colorado)

Cherry Creek Reservoir is an important recreation area near Denver that, until recently, was experiencing strong development pressure and rapid population growth. In the mid-1980s, the Denver Regional Council of Governments developed a management plan to prevent eutrophication in the reservoir. The plan identified phosphorous as the critical effluent, projecting that by 1990 loadings would have to be limited in order to maintain water quality standards established by the Colorado Water Quality Commission.

The Cherry Creek trading programme allows certain dischargers of nutrients into the Reservoir to earn phosphorous reduction credits, through the control of non-point source phosphorous discharges. The programme requires that non-point sources reduce their loading by 50 per cent on their own, before point sources can earn credits for any reductions. Because growth has slowed, phosphorous loads remain below established limits, and trading has not yet materialised (Carlin, 1992).

Summary remarks on nutrient trading

Overall, the experience with nutrients trading has been most successful within the conceptual framework of a “bubble” over point sources. In such a context, because nutrient abatement is largely dependent on up-front investments in treatment systems, trading becomes a tool for allocating and optimising investment. In all cases, the system of tradable nutrient pollution permits is underpinned by strong (and pre-existing) regulatory regimes which provide a framework, including sanctions on individuals for overall failures in pollution abatement.

Nutrients are not normally a pollution problem of short duration or local extent. This is because their levels in effluents and initial receiving waters have usually already been reduced to levels where no immediate effects occur. Instead, they often have effects over long distances (such as in estuaries or marine basins far away from the average point of discharge), or they affect the long-term quality of water bodies, such as lakes used for drinking water production (*e.g.* Lake Dillon).¹³ Because of this, they remain a focus of water quality management and may even become a more interesting field for applying tradable permits in the future. In effect, as a result of lower concentrations, the point of discharge becomes less relevant,¹⁴ and the annual load becomes more important.

Equivalent instruments to tradable water pollution rights

Quality objectives as traditional “bubbles”

Water pollution control is, in many instances, based on environmental quality objectives (EQO), which describe a desired characteristic of a water, usually as a chemical or physical value (parameter) or as a biological quality. In principle, EQO based on chemical concentrations can be considered as “bubbles” because total (annual) emissions or loads can be calculated on their basis. Tolerable loads might then be traded among polluters. The same may apply also to physical parameters, for instance to describe thermal pollution or “heating” of waters. Biological quality objectives are “holistic” in the sense that they capture the effects of many polluting substances (including any synergies) on whole ecosystems, as described by “lead” organisms.

EQOs are usually embedded in authorisation procedures establishing discharge conditions for individual polluters. Authorisations (licences, permits, consents, etc.) are normally established on the basis of uniform emission limit values (ELV), which may be technology-derived, as well as EQOs in a “dual approach” ensuring that both, ELVs and EQOs are respected. The case study on the control of Pentachlorophenol in the river Irwell in England serves as an example of an EQO applied without trading among polluters, and without reference to ELVs.

Quality objectives for Pentachlorophenol: Irwell Basin (England)

Pentachlorophenol (PCP) is a dangerous substance which (in the European Union) is controlled both under Directive 91/173/EEC (which regulates marketing and use but from which exempts the textile finishing industry) and under Directive 76/464/EEC on dangerous substances in the water environment (specifically, its “daughter” Directive 86/280/EEC). The latter requires Member States to set emission controls so that an environmental quality objective (EQO) of 2.0 micrograms per litre as an annual mean value is attained in the receiving water (see Edwards, 1996).

In the late 1980s, monitoring of waters in the river Irwell basin in East Lancashire, England, UK, revealed that a number of rivers contained PCP levels of 3-21 micrograms per litre, and thus, were in excess of the EQO. The area has an important textile finishing industry where PCP is used as a rot-proofing agent to prevent fungal and bacterial growth on fabrics. This application was identified as the most important source of PCP contamination in the area. Within the Irwell basin, 16 textile finishing companies discharged PCP in their trade effluents, all of them to sewers and thus not directly into receiving waters.

At that time, such indirect discharges were controlled by means of a trade effluent consent or permit issued by the sewerage operator which, for dangerous and controlled substances such as PCP, incorporated conditions based on the best technical means available (BTMA) determined by Her Majesty's Inspectorate of Pollution (HMIP). Discharges from the sewage treatment works to receiving (natural) waters were regulated by the National Rivers Authority (NRA) through a "consent" procedure specifying conditions necessary to meet established EQOs downstream of the discharge. (HMIP and NRA have since been merged into the Environment Agency.) Taking dilution into account, concentration limit values of PCP in effluent from sewage treatment works were set at between 2-6 micrograms per litre. The values were significantly more stringent than emission limit values established by Directive 86/280/EEC in relation to industrial plants engaged in the production of PCP.

The industry reacted by changing working practices to minimise losses, switching to off-site incineration of PCP solutions after use, establishing closed cycle processes, treating effluent with activated carbon or other methods, and in many cases ceasing to use PCP altogether. These measures reduced PCP concentrations in the Irwell Basin to well below the EQO. Once these measures had been taken, "grey cloth" imported from Asia which has to be cleaned before finishing emerged as an important source of PCP, with certain batches containing quite large quantities of PCP. Education of Asian cloth producers in the use of alternative, environmentally-friendly specifications therefore seems to be necessary as a follow-up strategy.

Summary remarks on tradable water pollution rights

The cases of tradable water pollution rights presented above contain a number of lessons. One of these stems from the observation that trading has been applied effectively only to pollution covered by a single chemical or (in the case of electrical conductivity) a single physical parameter. This does not mean that only pollution by identical substances is traded, as the parameters used often refer to classes of substances, such as soluble salts or substances oxidisable by bio-chemical processes. However, single elements or substances were the object of trading or regulation in the case of nutrient pollution or with Pentachlorophenol.

A second lesson concerns the location issue. In all cases, the locality of discharges to the recipient water is of little or no consequence. This is one necessary condition for trading to function without adverse effects on water quality and the stability of water-based ecosystems. Historically, water pollution was often significantly higher than it is today, at least in the highly-industrialised countries. With today's level of sewage and effluent treatment, there are few instances where releases result in acute eco-toxic effects on water bodies (apart from accidental releases, such as during the fire on the Schweizerhalle site of Sandoz on 1 November 1986). Pollution, and pollution trading, takes places at levels well below those where acute and local effects may be observed. Instead of concentrations, the focus today is on loads of pollutants entering the river system. The causality of any damage involves a common "sink", such as an estuary or lake at the end of the river, where pollution has long-distance effects that may also be delayed in time.

As a third lesson, this observation points in effect to a promising possibility of combining two instrument in water pollution control:

- limit values for concentrations, both in effluents and receiving waters; and
- tradable rights for loads.

Discussion and Conclusion

Opportunities and limitations

As is evident from the case studies presented in this report, there are a number of characteristics which facilitate or limit the emergence or establishment of tradable permit regimes in water resource protection and management. Among the key factors favouring trading are:

- In many cases, water use is rather homogenous within one area, notably in the case of irrigation districts. In such cases, trading patterns may be a long-established feature of water allocation with

many market participants, such as members of an irrigation association. These participants may trade frequently, such as annually with even more frequent “informal” transactions, involving differing and sometimes marginal quantities of water, or leases of water, or water rights. Where such regimes exist, the scope for increasing allocative efficiency may be limited, and no public policy action for market development may be necessary.

- Water is a variable resource, with significant differences in availability from year-to-year, or between seasons. Water users have adapted to this changeable environment, for instance by crop diversification or the construction of storage facilities. Differing profiles of risk-acceptance emerge in such circumstances, which can be an important factor in creating benefits from trading, as well in generating a subjective readiness to do so. In water management, there is rarely a “stable baseline”, and changes in resource availability may actually induce trading. This is contrary to the generally-held assumption that a reliable supply of a resource should be a component of an optimal framework for trading and market development.
- Within water management, there are a number of distinct possibilities available for defining tradable rights. This is obvious where quantities of water (of similar quality or from the same source) are traded, but it also applies to some parameters of water pollution and various water-based resources.
- In many cases, the infrastructure for effecting water transfers has been built up with subsidies, so that trading does not have to provide the revenues necessary for the initial capital investment. This may facilitate trading and reduce the potential for conflict, but it may also inhibit trading by masking the actual cost of water provision.

On the other hand, a number of factors inhibit water trading, or the establishment of tradable permit regimes in water management:

- Water is heavy, with a high weight-to-value ratio. In consequence, water transfers are often uneconomic because of the cost of pumping and the necessary physical infrastructure. In principle, water can be traded most readily downstream of a storage facility, and within a homogenous area of use.
- Most water uses entail significant impacts on other water users, imposing external environmental and resource costs, as well as social externalities. Thus, changes in water management induced by trading may affect many others than just the initial buyers and sellers. The resulting conflicts are usually managed within the context of Water User Associations, rather than as part of (regulated) markets.
- The effects of water use – be it for water abstraction, water pollution, or the use of water-borne resources – are not independent of where and when this use occurs. Any trading which shifts water use in space or time has to account for the consequences on other water users or ecosystems. While this does not make trading an impossible proposition, it may constitute a severe restriction in certain situations.

The relative importance and consequences of all the above factors need to be assessed in each case where existing water trading is analysed, or where the establishment of tradable permit regimes is being contemplated.

Strategies for introducing tradable permit regimes in water management

Based on the evidence (notably from the evolution from Water User Associations towards formal market structures), strategies and sequences for the introduction and promotion of water trading may be designed (cf. Smith, this volume), making use of various “instrument mixes”.

The first step would be to shift from technology controls on water use or water pollution (which are actually rather rare), and towards controls on emission. At the same time, water use permits should be linked to environmental quality objectives, and water pollution permits should be based on loads, rather than on concentrations. Care must be taken, however, to protect the water environment and other water users from the acute effects of excessive concentrations of pollution. The first step requires the emergence of effective controls on the quantity and quality of the water abstracted, the effluents discharged,

and the receiving waters themselves. The monitoring system would need to be designed to provide data of such quality that they can be used in court for the resolution of any conflicts arising from the use (or abuse) of tradable water rights or water pollution rights.

As a second step, trading may be allowed among water users with similar characteristics. This step necessitates the definition of tradable rights (covering all water available, only water in excess of existing permits, or water no longer available as part of reduction scenarios – similar definitions can be found for water pollution). The rights need to be allocated initially, and a mechanism for allocation be established. The definition and allocation of tradable rights needs to take account of the economics of water use and pollution control, as well as to anticipate the responses of water users to the establishment of a tradable permit regime. There must be sufficient room and opportunity to correct errors and to “fine-tune” the allocation of rights, as well as the trading rules. The role of environmental and market regulators needs to be defined, and rules for enforcement and sanctions need to be established.

Once a tradable permit regime is functioning, its continued ability to meet environmental and resource requirements must periodically be evaluated, including its ability to respond effectively to new (emerging) challenges. The regime should also be adaptable to account for changes in the boundaries of the system, such as an expansion of the territorial coverage, the emergence of unforeseen long-distance effects, etc.

Compatibility of tradable permit regimes in instrument mixes

In (almost) all cases, water resource protection and management in any given area has evolved into specific institutions, and given rise to locally-adapted habits, customs, traditions, rules or laws, which may be enforced by means of a variety of different policy instruments. Usually, water management makes use of “instrument mixes”. The issue therefore arises whether tradable permit regimes are compatible with any or all of the pre-existing instruments, and how tradable permits can be integrated into the water management, monitoring, and enforcement systems already in place. The following considerations are based in part on the case studies presented in this contribution, and in part on various other papers contained in this volume.

Compatibility with taxes and charges

Taxes and charges on water abstraction and water pollution are widely-used policy instruments in water resource protection and management. In addition, there are normally also financial charges for water services, notably water supply (incl. storage) and sewerage services. Water abstraction taxes or charges (sometimes in the form of administrative fees) are normally based on the water quantities taken from the environment. Water pollution taxes or charges, on the other hand, are normally based on the pollutants contained in effluent reaching the water environment.

One of the problems with tradable permit regimes in water management is that, in some cases, the holders of rights are unaware of their entitlement, or cannot be encouraged to trade. Imposing a tax or charge on “sleeping” water rights may effectively induce owners to sell their unused water rights. A combination of tradable permits with taxes or charges can have a number of other positive effects:

- It can help “sleeping” holders of rights to learn about the entitlements they own, thus reducing the cost of entry into the market.
- It may impose a cost on speculative banking of rights, with its attendant risks for water resource protection and management over time, and could have positive consequences for the acceptance of tradable permit regimes.
- It can earn an environmental (or resource) rent, that can be employed to carry out water management functions that are in the public interest, thereby redressing persistent environmental or resource externalities.

- It can induce the owners of rights to consider whether or not they need the whole entitlement, or only a part of it, thereby inducing them to place fractions of their entitlements on the market. This would lower the minimum quantity (and value) of trades and supports the formation of marginal prices (market creation).
- By imposing a cost on the difference between the number of rights held and the actual usage (which contributes to supply security), taxes and charges can set an additional incentive to save water or to reduce pollution, as any unused right then carries a price. In this way, a combination of tradable rights and taxes or charges can stimulate innovation and even facilitate the “claw-back” of unused rights for new auctions (thereby further lowering barriers to entry).

Taxes and charges, if applied in combination with environmental standards expressed in limit values for concentrations or loads of pollutants, have a significant capacity-building effect, both within the environmental enforcement agencies and with polluters (Kraemer, 1995). This effect is mainly due to the information flows necessitated by economic instruments, as well as to the incentive function. In light of the above list it would seem plausible to conclude that tradable permit regimes and taxes or charges can be mutually-reinforcing.

Compatibility with environmental quality objectives

Environmental quality objectives (EQO) are one of the most widely-used instruments in water resource protection and management. As noted in the earlier section on “equivalent instruments” (*i.e.* equivalent to tradable water pollution rights), EQOs are conceptually equivalent to “bubbles” and are not only compatible with tradable permit systems, but may even be considered a prerequisite in some cases. However, EQOs are not normally applied on their own, but are usually in conjunction with a range of other instruments of monitoring and enforcement, with which they are compatible. This does not mean, however, that these other instruments, some of which are addressed below, are themselves always compatible with trading within “bubbles”.

Compatibility with technology-derived standards (BAT)

Technology-derived standards, usually applied to pollution abatement and control, are often seen as being inefficient, because they require the same abatement standard from every polluter without regard to marginal abatement costs (or benefits). In addition they are often seen as being incompatible with tradable permit systems. The assumption behind this assessment is that technology standards (a prescription on abatement technologies to be used) and technology-derived standards [*e.g.* pollution abatement parameters such as emission limit values or a reduction percentage set with reference to the “best available technology” (BAT)] are static, and cannot therefore accommodate the economically- and environmentally-beneficial changes in pollution abatement that tradable permits might induce.

In fact, the approach of using technology-derived standards aims at establishing a dynamic process of continuous improvement in environmental performance. In principle, environmental (emission) standards are expected to follow technological development, and to provide for an ever higher level of environmental protection. By setting standards on the basis of BAT or some similar definition, regulators force all polluters with similar process or abatement technologies to meet the same standards for new installations. Normally, existing or “old” installations, may be operated under previous, less stringent standards until the end of their licence period, which should coincide with the end of their economic or technical life. Any installation built (and operated) later would then have to conform to the BAT in force at the time.

In principle, it would be feasible to combine dynamic technology-derived standards with tradable permits, as in the following scenario: at a time when the best available technology changes, old installations are given a derogation period during which they may continue operation, provided they hold permits to pollute. Such permits, which could be discounted each year by 10 per cent of the original entitlement for instance, might then be sold by those polluters who install better pollution control equipment (or shut down operations) to other polluters who choose to continue operating their “old” installations. Such systems could be introduced each time there is an improvement in the BAT, or when a significant difference develops between the BAT and the (more stringent) “state of the art”.

Compatibility with established principles of environmental policy

Tradable permit systems appear to be compatible with the polluter-pays and the resource-user-pays principles, in that they impose a cost on water pollution or water resource consumption, or generate a reward for pollution abatement or resource savings. Nor should there be any fundamental problems with the principle that pollution effects should be prevented at the outset, rather than mitigated later by compensation measures (prevention principle). Some countries also subscribe to the “minimisation principle”, according to which pollution should be reduced even if no risk is demonstrated, as long as this is economically-bearable. This principle is best applied through emission limit values, which may be based on technology standards and, as discussed above, which may be combined with tradable permit systems.

The compatibility of tradable permit systems with the principle that pollution should be reduced at source (rather than after dilution) appears to be weak. However, a successful tradable permit system would direct resources at those sources of pollution where abatement is most cost-effective, resulting in an optimal level of pollution reduction at source. In this sense, tradable permit systems *are* compatible with this principle, if “source” is understood to mean the sum of all sources (of similar characteristics).

An important tenet of environmental policy is that action should be taken to reduce risk as soon as there are reasonable grounds to assume that a pollutant may be dangerous or hazardous (precautionary principle). According to this principle, action may be taken if a risk is demonstrated in the absence of a conclusive proof of causality (which may be impossible to provide at the level of certainty usually expected in the natural sciences), and before the associated risks can be fully assessed in terms of probability and likely damages (in nature, value, and variance). The precautionary approach may call for significant changes in pollution control as a consequence of new scientific insights. Care must be taken in the design of tradable permit systems to ensure that they provide the necessary flexibility to react to new challenges, since they might otherwise be incompatible with the precautionary principle.

Tradable permit systems within instrument mixes

It is obvious from the previous discussion that tradable permit systems are, on the whole, compatible with most existing environmental policy principles and instruments. It should therefore be possible to integrate tradable permit systems into existing instrument mixes in a strategy of gradual improvement in environmental monitoring and enforcement practices. It should be noted, however, that successful tradable permit systems must be based on the same broad foundations of environmental (or water) management as other instruments. Notably, there must be effective monitoring of water qualities and quantities and of ecosystems, but also a well-functioning system for settling disputes.

Overall assessment

Overall, there are large number and a wide variety of examples involving tradable permits for water resource management and water pollution control. However, in spite of their number, existing applications remain restricted to very few countries, and each case is highly specific and not very amenable to broad generalisations.

Successful trading regimes tend to be based on pre-existing institutions, which have incorporated the instrument of tradable permits into their operations, in order to achieve either “one-off” re-allocations of water rights or permanent (dynamic) efficiency gains from continuous trading. They also tend to be intertwined with traditional regulatory regimes, to ensure both their effectiveness and their integrity, by providing a (sometimes threatening) back-up of potential regulatory intervention.

The most successful markets for water as a resource can be found in irrigation, where the number of participants tends to be high (hundreds or thousands) and trades in water are frequent, recurring, and for small volumes. Thus, marginal efficiency gains can be realised.

Such efficient markets can be threatened, however, by growing monopoly power in cases where single entities hold or obtain a large proportion of the water rights available in a given region. This problem tends to be aggravated where water rights are permanent rather than temporary. Trades of

permanent water rights (as distinct from trades in water) tend to be much rarer, and with much higher prices. The number of market participants buying and selling water rights is normally very small, and there are often only one (or a few) active buyer(s) and seller(s) over many years.

Markets in water pollution permits are still in their experimental phases, but experience is accumulating, primarily in Australia and the US. So far, there has been little discussion about the effects of such trading on the effectiveness of environmental regulation. In particular, there appears to have been little discussion to date on the possibilities of combining emission standards with tradable pollution permits, or on the monitoring and adjustment needs where tradable permits are combined with environmental regulation using environmental quality objectives.

It appears from the case studies and the descriptions of equivalent instruments that there are a large number of institutions which use trading systems internally, even if they do not carry the labels “tradable permit”, “right”, or “market”. These are common property management regimes, with sometimes elaborate governance structures and allocation systems, which use economic principles and instruments as part of their internal operations. Collectively, they are often referred to as “Water User Associations”, and are sometimes dismissed as out-dated, inefficient, and inflexible community organisations, that would be better replaced by markets. The example of Chile reveals, however, that markets depend on the presence of a number of contextual factors, such as effective and fast conflict resolution mechanisms. If the right context is there, community-based regimes that are embedded in local customs and social constraints can be both effective and efficient.

NOTES

1. Various general or theoretical publications not relating directly to the cases covered in this contribution are listed separately at the end.
2. Information on Scottish salmon fishing was provided by Clare Coffey, Institute for European Environmental Policy, London.
3. The Industry Commission (1992) of Australia has reviewed the country's experience with tradable permits relating to water resources. Other surveys include those by Delforce *et al.* (1990), Pigram *et al.* (1992) and James (1997). The Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ 1995) has produced guidelines for the implementation of property rights in water.
4. Several case studies relating to Chile have been collected by Hearne and Easter (1995: Annex).
5. The following priorities for leasing water have been established by the "Committee of Nine": 1) existing canal companies that own storage space; 2) agricultural users who have traditionally used rental pool water; 3) new agricultural users; and 4) any other user (such as a power company).
6. Case study description largely follows Wahl (1989: 133-135).
7. Case study description largely follows Wahl (1989: 135-136).
8. For a description of the federal water banking operations in California during the 1976-1977 drought see Wahl (1989: 136-138).
9. Hearne and Easter suspect that EMOS may be unwilling to pay for water rights, hoping that, as a public water supplier, the government will interfere in water markets on its behalf, should severe water shortages develop.
10. Act relating to the Imposition of a Fee for Groundwater Abstraction [Gesetz über die Erhebung einer Gebühr für Grundwasserentnahmen] of 26 June 1989, Gesetz- und Verordnungsblatt I, p. 115 (Hamburg).
11. The example of irrigation water trading in the Gouldburn-Murray Region of Victoria, Australia, shows, by contrast, how difficult it can be to "construct" a water trading system where no traditional regime can be used as a basis. The contrast is provided by the Murrumbidgee Irrigation Area in the same river basin.
12. Riggs and Yandle (1997) give further sources on the Tar-Pamlico basin case.
13. The international Lake Constance in the Rhine basin is another example.
14. For instance, the European Community's Urban Waste Water Treatment Directive allows for averages of pollution reduction established at the regional level to be used instead of emission limit values. This opens up an opportunity for nutrient pollution trading.

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Chapter V

TRANSFERABLE FISHING QUOTAS: EXPERIENCE IN OECD COUNTRIES

by

Paul Wallis

Fisheries Division, Directorate for Food, Agriculture and Fisheries
OECD, Paris, France

Introduction

Tradable fishing quotas have been used to manage fisheries in OECD countries since the 1970s. Commonly referred to as individual transferable quota (ITQ), this management instrument has been used to manage fisheries in Australia, Canada, Iceland, Netherlands, New Zealand and the United States. This paper provides a broad overview of the application of ITQs and attempts to focus primarily on the quota trading systems employed.

The difference in design in ITQ systems is marked. While the theories underpinning the systems are relatively simple, transposing these into workable policies has been quite different in different countries and in different fisheries. The discussion in this paper is therefore very much in general terms, with the use of selected examples.

Description of the approach

Background and where ITQs are applied

Traditionally, fish stocks have been managed under regimes of “open access” (*i.e.* the rights of access to the fishery are open to everyone and anyone). There is no incentive among fishermen to conserve the resource for future use, since no fisherman has exclusive use. This phenomenon results in the dissipation of profits and increased exploitation rates, pushing the size of the fish stock below a size that produces the maximum sustainable yield. Low profits lead to economic problems and social disruption. Over time, the economic and social problems will prompt demand for government policies which can create market distortions and further hinder sustainable use. Excessive effort in the fishery also means that capital and labour – factors of production that could be more profitably employed elsewhere in the economy – are occupied in a “race-to-fish”. Net national welfare suffers as a result.

Governments have intervened to try to address this problem. Often they have tended to restrict the activities of certain inputs (*e.g.*, numbers of vessels, numbers of fishers). However, these measures often fail to limit the expansion of fishing effort. For example: a limit on vessel numbers can be circumvented by increasing vessel size, power and catching capacity. Management through output controls, like setting a limit on the total allowable catches (TAC), appear to offer a solution to biological problems. But economic uncertainty and resultant problems often remain.¹

Management instruments that define and allocate the rights to catch fish create incentives for maximising economic, biological and social outcomes. Once fishers own the right to catch a certain amount of fish, they have a private incentive to maximise its economic value and to preserve its biological status. A common example of such a property right is the *individual transferable quota*.

Table 1 gives an indication of the number of species or species groups that are managed by ITQs. The specific fisheries involved use a range of gear types: abalone fisheries consist of harvesting by divers, oysters by dredging, and cod by large trawl nets. The fisheries managed with ITQ include inshore fisheries, with many smaller-sized operators, through to large factor trawlers with fewer operators.

Table 1. **Individual Transferable Quotas in OECD Countries**

	Number of Fish Species or Species Groups ¹
Australia	4
Canada	14
Iceland	16
Netherlands	4
New Zealand	33
United States	4

1. Annex I provides a list of the species and species groups in OECD Member countries.

What is traded and under what conditions?

In all cases, the ITQ relates to a flow from a resource stock, not to the resource itself. What is traded is the right to catch a certain quantity of fish, at a certain geographic location, during a certain period of time (normally one year). The authority of quota is required to harvest the yield from the stock. This authority may be acquired by fishers in many ways (discussed below), but there are two main trading possibilities:

- trading of the *perpetual* right by sale and purchase of the quota; or
- trading of the *annual* right to harvest that is generated by the quota.

In New Zealand, the ITQ is the perpetual right and each year it generates Annual Catch Entitlements (ACE) which are the annual harvesting right. In Iceland, the TAC share is the perpetual right and each year they generate annual vessel quotas. In the United States and in the Netherlands, quota is either sold (the perpetual right) or leased (the annual right) between vessel owners. For example: in the US surf clam fishery, a vessel is not required to own quota to harvest the animals, but quota must at least be leased from another vessel owner (OECD, 1993).

The separation of the perpetual right and the annual harvesting right raises interesting issues. While this is not the appropriate context to explore them one comment could be made. The separation of these rights into the hands of different actors could conceivably undermine the resource stewardship incentives created by the quota. This is because the users of the quota are not the same as the owners, and thus, they do not have the same long-term stake in the health of fisheries resource.

Who trades?

Fishing companies and individual vessel owners normally own and trade quotas. The concentration of ownership in the hands of these participants is due to two main factors: initial quota allocation; and the need for a fishing license or permit to actually use the quota. Initial allocation issues will be discussed later in this paper. Fishing licenses or permits require a person to meet a whole range of additional conditions related to resource management. For example, in New Zealand, commercial fishers need the authority of an ITQ right (acquired by buying ITQ or ACE), and a fishing permit which would specify where the fish may be landed and what information should be furnished to the monitoring agency. In addition, the fisher must comply with fishing regulations regarding minimum legal mesh sizes and minimum legal fish sizes.

In Iceland, the restrictions are more explicit: only people who own fishing vessels with a valid fishing license can hold quotas. In addition, that person's total holdings of quota must not exceed the capacity of the vessel in question. In effect, the set of possible quota holders is severely restricted (OECD, 1997b). In the Australian southern bluefin tuna fishery, quota transfers may only be made to other commercial operators using the same or other specified gears (OECD, 1997b).

A common restriction: foreign ownership

Fisheries are often perceived as valuable and strategic industries because, like agriculture, they are linked to sensitive issues related to food self-sufficiency. The right to exercise sovereignty over fisheries resources in an exclusive economic zone is still a relatively new possibility for coastal states.² Foreign control or interests in fisheries therefore tend to be carefully regulated – quota ownership is no exception. Foreign interests in quota, and even annual harvest rights, are usually prohibited or limited to levels to ensure foreign “control” does not result.³

Why has the trading arrangement been designed in a particular way? How is the system administered?

Quota trading systems are usually designed to meet the needs of the main users – the fishers. Other demands from society also exist and, although the ITQs should create resource stewardship incentives; in every case, governments set the overall catch limits under which ITQs are allocated.

Quota systems deliver the most benefits when their markets are relatively free. The benefits of improved economic performance can only be realised if quotas (the perpetual right) are allowed to move and end up residing in the hands of those who can make the most profit from them. Where annual harvest rights (*e.g.* quota leases, ACE) are used for multispecies fisheries – fisheries with a number of species that, because they swim together, are harvested together – a more flexible system will help fishers obtain the quotas (either by buying or leasing, etc.) to cover any shortfall in their own quota holdings. Without such flexibility, fishers will be more inclined to dump fish.

There is a considerable diversity between the trading arrangements in different countries and different fisheries. Most are characterised by transfers between fishers whereby information on the transfer – quantity, date, buyer and seller details – are recorded and sent to a central registry authority. The information recorded in the New Zealand quota registry is shown in Box 1. This register, and the separate register of Annual Catch Entitlements, are public registers, open for inspection. Information on prices is also collected, but is not publicly-available. In New Zealand, an attempt was made to establish an electronic trading floor for ITQ, but this failed in 1988 after it was unable to become self-funding. Actual trading now takes place face-to-face or through advertisements in newspapers (Boyd and Dewees, 1992).

Box 1. Information shown in the New Zealand Quota Register

- Total Allowable Catch.
- Total Allowable Commercial Catch.
- Quota allocated to each person.
- Every transfer of quota (number, date, names of transferor and transferee).
- Aggregate holding of quota by each person.
- Caveats or loans registered over quota.

In the other country that has made widespread use of ITQs, Iceland, transfers of quota are registered with the Directorate of Fisheries. In 1998, the Icelandic Parliament passed legislation that established a Quota Exchange. The Quota Exchange is an auction market for trade in catch quotas and is financed by a

tariff on registered bids and offers of sale. The legislation requires that all catch quota trades be made through the Exchange. The only exceptions are for: i) catch quota transfers between vessels owned by the same parties; and ii) for transfers that involve an exchange of equal value. The Exchange commenced operation on 1 September 1998.

In the case of the Australian southern bluefin tuna fishery, trades (temporary or permanent) are made through the Tuna Boat Owners Association of Australia – a stand-alone industry organisation – or through brokerage firms.

Although the government has responsibility for controlling the overall national TAC, Producers' Organisations (POs) manage quota transactions in the Netherlands. The government created a framework for the establishment of POs in 1993. Under these arrangements, individual fishers assign management of their quota (and their "days-at-sea"; an additional management measure) to a group management board. While the fishers retain ownership of their quota, it is made easier for them to buy, sell or lease quotas and "days-at-sea" in the event of surpluses or shortages. Under this system, fisheries are required to submit a plan for their fishing activities to the board to show how they plan to operate during the season. Fishers also make an undertaking to sell their fish only at auctions.

What approaches are taken to reporting, monitoring and enforcement?

The interest of fisheries management authorities resides in ensuring compliance with catch limits and ensuring that those catching the fish have the correct ITQ holdings. Attention thus tends to be paid to ensuring that knowledge of precisely who has the quota authority. Anderson (1992) points out that, compared with other management systems, there are fundamental differences in the construction and operation of a suitable monitoring and enforcement system to support ITQs. Transferability of quotas means that it is necessary to keep track of the amount of quotas actually owned by individuals.

In New Zealand, fishers have obligations to balance their stocks against their ACE (annual catching rights generated by quotas) and report on what they catch and what happens to it. A quota balancing regime is operated to ensure that catch does not exceed the catch limits. A commercial fisher must hold sufficient ACE to cover catch before going fishing – it is an offence to take fish in excess of ACE holdings. For shellfish and "at risk" stocks, the commercial fisher must acquire sufficient ACE to cover excess catch by 15 days after the end of the month in which the fish is caught. For the majority of stocks, the commercial fisher has until the end of the fishing year to balance catch with ACE or satisfy a demand for the deemed values of the fish.⁴ Failure to do so may result in the responsible fisher being prosecuted. A fisher is liable for deemed values for any catch in excess of ACE held on a monthly basis. Davies (1992) emphasises that the administrative tasks required under ITQ management are enormous (compared with other management systems), given the increased monitoring requirements.

Producers' Organisations in the Netherlands, with the use of fishery plans and a more co-operative approach, are intended to increase the degree of compliance with quota regulations. Prior to 1993, tensions had been rising between the government and fishers over non compliance with catch limits (OECD, 1997a). POs can fine members who do not obey the rules. These sanctions, which aim at reducing profits, vary according to the gravity and frequency of the transgressions. In some cases the management boards have not been willing to monitor and sanction fishers. In these situations the government's General Inspection Service carries out the necessary role.

In Australia and New Zealand, the introduction of ITQs prompted a change in the way enforcement was conducted. Effective enforcement of the ITQ system required specialised investigators, auditors and accountants, since the focus is more likely to be in the area of "paper fraud". The skills of traditional fisheries compliance officers have not been superseded, but have been supplemented (OECD, 1997b). Crothers (1988) emphasises that industry compliance with quota management was ensured by policing product flow from fishing vessels at off-loading points, through processing to the point of sale, essentially a fish "paper trail".

Particular implementation problems – obstacles to successful implementation (including any problems of compatibility with other existing policies)

Introducing ITQs is not an easy process. OECD (1997a) identified the initial allocation of quota as one of the most serious problems in implementing this form of fisheries management instrument. There are often concerns about reductions in catching opportunities, quota consolidation and elimination of small scale fishers. Part of the concerns relate to the fact that quotas are often introduced “at last resort” when other management options have been exhausted. Thus, ITQs are often introduced into overcapitalised and over-fished fisheries – when reducing catching rights and the resultant rationalisation in the fishing industry is inevitable.

Initial quota allocation

Overcoming fishers' opposition skews the debate on implementation. Quota allocation is usually done in such a way to reflect the interests of existing fishers. These interests, while not well defined in terms of realisable property, are nevertheless recognised by governments in most OECD countries. Also, for a quota system to work, it must have the support of the fishers. What better way to obtain support than to allocate quotas to current participants?

The methods of recognising these interests in allocation are many, but generally reflect the fishers' catch histories. In the Canadian Pacific halibut fishery, the following formula was used to determine quota allocations: 70 per cent of a vessel's allocation was based the vessel's historical catch between 1986 and 1989, and 30 per cent was determined by the vessel's overall length. In Iceland's demersal fisheries, the TAC share allocation equated to the vessel's average share of the total catch during the three years prior to the introduction of the quota system in 1994. In New Zealand, 1986 quota allocations were based upon catch history and “commitment and dependence” on the fishery. The resulting allocations in many cases were more than the estimated sustainable harvest levels from the fisheries. To reduce catch limits to sustainable levels, NZ\$ 42.3 million was spent removing 15 800 tonnes of catch allocations. As Sharp (1997) notes, the buy-back scheme was an important part of the transition to ITQs and a way of dealing with the economic and biological problems created by previous forms of property rights and their associated incentives.

Quota allocation is usually made to vessel owners. Fish workers, deckhands, etc., are not normally allocated quotas. In Iceland, only people who own fishing vessels and a fishing license may hold quotas. Furthermore, the total holdings of a vessel, may not exceed the capacity of the vessel in question.

Existing management instruments

Existing fisheries management controls related to inputs (*e.g.* mesh size limits, vessel size limits) tend to be retained when ITQs are introduced. Output controls – like competitive TACs, individual (non-transferable) quotas, individual vessel quotas – tend to be replaced. Technical measures are retained because of management objectives additional to just limiting catch rates. In the Netherlands, for example, three types of input control are used in addition to ITQs (*i.e.* engine power licenses, shrimp permits and entitlements for the coastal zone, derived from the EU vessel file registration).

Additional measures are often implemented in response to perceived needs to:

- manage fishing capacity (*e.g.* the EU's Multi-annual Guidance Program);
- protect juvenile fish (*e.g.* technical measures like mesh size limits); and
- mitigate effects on the wider ecosystem (*e.g.* incidental seabird by-catch).

In relation to the first two of these points, OECD (1997a) notes that these additional measures do not necessarily lead to improved resource conservation of the fish stock.

How has the interface between suppliers and demanders in the tradable permit market been maximised?

Only two countries have established a quota market in an institutional sense: Iceland's Quota Exchange and the Producers' Organisations in the Netherlands. New Zealand established an electronic trading floor in 1986 but, as it was unable to become self-funding, it ceased operating in 1988. Other countries see the provision of information on quota ownership alone as being sufficient, and leave the formation of any market to the industry participants. Quota trading in these countries tends to be conducted face-to-face, or by brokers.

In the Netherlands, quota exchanges take place through the POs. In Iceland, the quota exchange system performs a similar function on a national scale. Box 2 outlines describes the respective functions of the Icelandic Quota Exchange and the Directorate for Fisheries.

Box 2. The roles of the Icelandic Quota Exchange and the Directorate of Fisheries

The Icelandic Quota Exchange:

- Records purchase bids and sale offers.
- Services as a forum for trade in catch quotas.
- Handles payment settlement between buyers and sellers.
- Provides information on trading on the exchange.

The Directorate of Fisheries:

- For sale offers, establishes that the catch quota may actually be transferred from the vessel in question.
- Or purchase bids, establishes that the catch quota may actually be transferred to the vessel in question.

How has the need for a publicly-visible market price been handled?

Market price information is only publicly-available in a systematic manner in Iceland. The quota exchange is required under law to publish information each day on trading, including trading prices, total quantity and total volume by species. The exchange is also required to publish information that it thinks would be useful to establishing effective price formation. In the Netherlands, the POs manage the intra-group ITQ exchanges – information on buy and sell offers are posted to each member by the PO.

In other countries, prices for ITQs are not publicly-available in a systematic way. Quota brokers are common in New Zealand and Australia. In New Zealand, while prices are collected, they are not made publicly-available. As discussed above, the primary function New Zealand's ITQ and ACE registries is only to make available information on ownership and financial interests.

What have been the effects of the programme on third parties, and how have these effects been limited?

Quotas are not normally allocated to other parts of society that have an interest in the resource such as recreational users and environmental groups. This is the entry point to another debate: should those who receive the exclusive rights to the commercial use of a fish stock under ITQs be required to pay a return to the rest of society in the form of royalties or resource rentals? Most countries that operate ITQ management recover costs associated with fisheries management (*e.g.* Australia, Canada, Iceland and

New Zealand). However, none of these countries charge royalties or resource rentals to extract a return for society from fishers' use of the resource. The beneficiaries of this policy are those fishers who receive the initial quota allocations for free. Once the quota is sold to another fisher, the benefits from this free allocation are realised – having been capitalised into the sale price – and it then becomes difficult on equity grounds to justify the introduction royalties or resource rentals.

In New Zealand, the process for setting of catch limits attempts to provide for the views and use patterns other stakeholders. This is done through consultation mechanisms and through catch right allocations. When setting TACs, the government is required to consult with all interested parties and provide for the input of Maori, before setting the TAC for any stock. There are two types of catch limits – the total allowable catch (TAC) and the total allowable commercial catch (TACC). The total allowable catch (TAC) is set first. From this the government quantifies the TACC for a particular fishing year, making allowance for recreational and Maori customary non-commercial fishing interests and all other sources of fishing. This includes the quantity required for research and the amount taken illegally each year. Based on this allowance and the available scientific data the government decides what the TACC should be. ITQ catching rights for the year are then derived, as a proportion of the TACC.

How have the legality and the efficiency of the permit trading system been protected?

The creation of the private property in ITQ systems is normally accomplished through public statute. ITQs are seen as valuable private property and their introduction is normally empowered by carefully designed legislation.

OECD (1993) notes that "ITQs... are a repository for substantial rents. This phenomenon creates incentives for fraud and black market activity. As fisheries rights increase in value, the effort and skills required to protect those property rights will need to be upgraded". Legislation underpinning ITQ systems therefore tends to have scope for the application of stiff penalties. In Iceland fines of up to IKr 250 000 have been recorded, and possibilities exist for the confiscation of catch and equipment (OECD, 1993). In New Zealand, selected offences under the Fisheries Act can attract penalties of up to 5 years in prison or a fine of up to NZ\$ 250 000, in addition to the forfeit of quota and property.

What has been done to minimise the transaction costs of the system?

Attempts to minimise transactions costs creates a tension with the objective of flexibility in an ITQ system. Flexibility is especially important in multispecies fisheries for ensuring that fishers are not faced with incentives to dump fish. A flexible system will make it easier for a fisher to obtain catching rights to cover his or her catch. Attempts to ensure flexibility have tended to result in complexity and have raised the transaction costs for the administering authorities and fishers. Box 3 contains a description of the various methods used in ensuring flexibility in ITQ systems [taken from Squires *et al.* (1998)]. These methods are in addition to the trading and leasing options available to fishers.

In New Zealand, a deliberate attempt has been made to simplify the system, while attempting to preserve some of its flexibility. The *Fisheries Act* 1996 removes opportunities for banking and borrowing or quotas, catching against another's quota, and leasing of quotas (replaced by trading in ACE). These opportunities were removed in an effort to reduce compliance and administrative costs.

Due to the increased flexibility which facilitates the leasing of quotas between group members, the Producers' Organisations in the Netherlands may have increased transactions costs. However, the additional flexibility, and the positive results it provided (*e.g.* improved utilisation of the total quotas), are deemed to more that outweigh the costs of this mechanism.

How has the design of the system dealt with changes in the membership over time, if any (additions, subtractions)?

In most cases, new membership is contingent on acquisition of existing quota rights. Those fishers wishing to leave the fishery sell their quota. If there is an increase in the overall catch limit, new catching

Box 3. Flexibility in ITO Systems

Borrowing of quotas: exceeding quotas in one year may offset against next year's allocation (5 per cent in Iceland; 10 per cent in New Zealand in the past).

Banking of quotas: under-harvested quota in one year may be harvested in the following year (20 per cent in Iceland).

Fishing against another's quota: allowing for entry into an arrangement with another fisher for catch to be counted against that fisher's quota holdings.

Retroactive trade in quota: Allowing fishers to acquire catching rights after they have landed in the fish, while ensuring a final reconciliation by the end of the year.

Deemed value: payment to the government of an imputed value to compensate for their catch caught without quota. The value charged is designed to remove the profit from unauthorised catch, but remove incentives to dump the fish.

By-catch trade-offs: quotas in other species exchanged, using agreed exchange rates, for other species.

rights are allocated to existing quota owners and not made available to prospective new entrants. For example, Icelandic quotas are expressed as TAC-shares; an increase in the TAC increases the TAC-share and quota for each existing fisher.

Some countries have additional impediments. For example, only vessel owners may hold quotas in Iceland. In the main the impediments tend to be economic: quota becomes very expensive. Quota values capitalise future profit streams and expectations about returns on sunk costs. High quota values make it difficult for prospective new entrants into the fishery.

In what way do the rules of operation encourage technological change or other elements of dynamic efficiency?

The expression of ITQs as a transferable catching right provides incentives to enable fishers to introduce efficiencies to maximise their returns. Rather than trading units of effort (*e.g.* days-at-sea, vessel size), the focus is on the unit of output from the fishery. In this sense, the ITQ management does not constrain developments in efficiency of input factor use. The extent to which there actually have been observed improvements in efficiency is discussed below.

Have the economic costs or the environmental effectiveness associated with the system been evaluated? With what results?

Much of the debate regarding the use of ITQs has tended to take place in the normative sense. Empirical work to support the normative assertions regarding the usefulness of ITQs as a management instrument has not been as forthcoming. Proponents of ITQ systems have tended to construct markets and describes incentives in a normative terms, without necessarily drawing from empirical evidence to support their fabrications.⁵

The OECD Fisheries Committee recently completed a study (hereafter "the OECD study") that assessed the performance of fisheries management instruments in Member countries (OECD, 1997a). An analysis of the performance of individual quota management was included in this study and, in effect, represented a step towards reconciling empirical evidence with normative assertions. There was sufficient information to analyse IQ (non-transferable quotas) and ITQ performance in 42 different fisheries. This section draws heavily from that study.

"Individual quotas are shown to be an effective means of controlling exploitation, of mitigating race-to-fish and most of its attendant effects, of generating increased profits, and of reducing the number of participants in a fishery." "(Individual quotas) present serious problems with initial allocation of quota and with enforcement and compliance." OECD (1997a).

Environmental performance

The OECD study found that evidence of improved resource conservation was mixed. In the main, IQs and ITQs were observed as being effective in limiting catch at or below the total catch limits set by the fisheries authorities. Of the 31 fisheries for which information was available, catch in 24 of the fisheries was maintained at or below catch limits. In three fisheries, previous problems with catches exceeding limits were eliminated under ITQs. There were eight cases where surpassing catch limits was a problem, often due to insufficient monitoring and enforcement. For example, in the Canadian herring fishery, high demand and prices for herring roe, difficulties in weighing herring, and collusion between buyers and sellers, resulted in catch limits being exceeded. In Canada and Australia, illegal fishing was the cause of over-fishing in some high value fisheries.

Some of the resource conservation problems may relate to the fact that the overall catch limits were set at too high a level. According to the OECD (1997a), 24 fisheries experienced at least temporary declines in stocks and near collapses in the case of some pelagic fisheries. Information problems are often a reason for problems in setting the appropriate catch limits. For example, when ITQs were introduced in New Zealand, there was little information on the status of the stocks and such information is still lacking for many fisheries. In the case of the orange roughy stocks, the productivity of the stock was over-estimated and catch per unit of effort declined markedly in the late 1980s.

Sissenwine and Mace (1992) concluded that there was little evidence to support the assertion that the status of New Zealand's fish stocks had improved since the introduction of ITQs, primarily because of the dearth of information. They state: "The general conclusion is that TACs are not closely tied to the best available assessments of the fisheries resources, nor are catches strongly controlled by the TACs. Some valuable stocks have probably declined in abundance. To date, the track record of ITQ management with respect to conservation is not good." Furthermore: "While ITQs are not responsible for the problem, they have done little to resolve it." Annala (1996) suggests the situation is becoming more positive. Based on 1994 stock assessments, of the 149 fish stocks that are actively fished in New Zealand's EEZ:

- 13 (8.7 per cent) were estimated to be below the biomass size that would produce the maximum sustainable yield (B_{MSY});
- 13 (8.7 per cent) were estimated to be above B_{MSY} ;
- 42 (32.2 per cent) were estimated to be at or near B_{MSY} ; and
- the status of the remaining 75 (50.3 per cent) was not known.

Theory suggests that there will be problems with the by-catch of other species and high-grading. By-catch problems can be acute in multispecies fisheries because fishers face the problem of reconciling their catches with their quota holdings (Squires *et al.*, 1998). In addition, there is the problem of by-catch of non-market or undesired species. Of the fisheries surveyed in the OECD Study, by-catch was reduced in a few fisheries, but increased in nearly as many. However, the analysis was only able to assess the change in seven fisheries (OECD, 1997a).

High-grading involves the discarding of less valuable fish, and the retaining of the more valuable fish, in order to maximise the value of the catch against quota. OECD (1997a) found evidence of high-grading in 10 fisheries. There was no evidence of high-grading in Icelandic fisheries. Additional measures like on-board observer coverage appear to reduce the incidence of high-grading (*e.g.* Canadian Northern shrimp fisheries). Similarly, on-board observers may have assisted the reduction in Alaska groundfish discards, which declined from 24 per cent to less than 10 per cent in the sablefish fishery after ITQs were

introduced (Buck, 1995). It could be expected that additional restrictions might reduce the catch on lower value fish (*e.g.* minimum mesh sizes to reduce catches of small fish). However, OECD (1997a) did not find evidence to support this contention and, indeed, high-grading was reported in eight fisheries with mesh size restrictions.

Economic performance

With regard to economic performance, OECD (1997a) notes that introducing quotas is expected to reduce and remove “race-to-fish” and its resultant problems including fleet over-capacity and excess effort. Evidence from the OECD study indicates that, in 12 fisheries, there was an elimination of the pre-existing “race to fish”. For example: the Canadian halibut fishery season changed from being a few days long (under competitive TACs and limited entry) to being able to run for the whole year under quotas. There was a clear contrast between quota systems that were transferable and systems that were not. When transferability was introduced into the quota system for the Canadian halibut fishery, the number of vessels dropped by 20 per cent to 353 vessels.

As quota values reflect the discounted stream of future profits, positive quota values attest to the generation of resource rents and the creation of wealth. Of the 42 fisheries in the OECD study, there was evidence of creation of resource rents and wealth in 22 of these. There was no contrary evidence from the other fisheries surveyed. There were increased profits, or at least increased cost-effectiveness, in 23 of the fisheries surveyed. Box 4 summarises information presented on the performance of ITQs in Icelandic fisheries (OECD, 1997b).

Box 4. **Economic Impact of ITQs on Icelandic Fisheries**

Herring: increased technical efficiency; significant improvements in catch per unit of effort.

Capelin: major improvement in economic efficiency; substantial reduction in fleet and effort; substantial improvement in catch per unit of effort.

Demersal: significant economic rents generated; reduced harvesting costs; higher quality catch.

A recent study of productivity and profitability in Icelandic fisheries supports these results. Valsson and Klemensson (1998) show that productivity growth was considerably higher in the years after the introduction of the ITQ system. During 1980-1990, the average annual growth in productivity was labour productivity increased by 0.74 per cent; in 1991-1995, the labour productivity increased by 1.53 per cent per annum. A similar trend of improved productivity growth after introduction of ITQs is evident with capital. For 1980-1990, annual capital productivity fell by 0.13 per cent; in 1991-1995, it grew by 1.3 per cent. The authors also construct a multi-factor (capital and labour combined) productivity index and estimate that during 1980-1990, this index grew by an average annual rate of 0.56 per cent. During 1991-1995, the index grew by 1.48 per cent per annum. While these improvements in productivity certainly support the hypotheses associated with ITQs, the authors point out that other factors also affect this index – technological progress, fishers’ skills levels, natural conditions and legal and institutional factors.

In Australia, there is mixed evidence regarding economic performance under ITQs:

- despite several reductions in catch limits in the Southern bluefin tuna fishery, the fleet remained profitable, with the cost of catching a tonne of tuna falling by 25 per cent and a resource rent of A\$ 6-7 million being earned in 1986-87 (Geen and Nayar, 1989); and
- in the South-east trawl fishery, return on capital fell in the Danish seine sector, but increased in inshore and offshore sectors (OECD, 1997b).

Evidence from the OECD (1997a) study strongly supports the view that employment is reduced under IQ and ITQ programs. Of the fisheries surveyed, employment fell in 19 and rose in four. This effect is likely to be intensified in a transferable (ITQ) context, where changes in ownership provide increased opportunities for rationalisation of fishing effort. New Zealand fisheries provided a counter-example: in the eight years following the introduction of ITQs, employment in the catching sector increased by 34 per cent and in the processing sector by 16 per cent (OECD, 1997b). A possible reason why these positive trends emerged could be the replacement of foreign chartered vessels with New Zealand vessels. As employment on foreign chartered vessels operating in New Zealand's EEZ is not included in employment statistics, the net impact on total employment in the catching sector may quite possibly have been neutral or even negative.

The salient issue in the debate on employment effects of ITQs is well summarised by Buck (1995): "... a shift from many short-term, seasonal jobs to fewer, long-term jobs, may lead to more stability...". "It comes down to a value judgement as to whether a boom-and-bust economy is better than a smaller but more stable economy..." The use of ITQs will expose operators to the long-term economic viability of the fishery, whereas other management measures (and financial support policies) may mask its more correct biological and economic capacity. Often the blame placed upon ITQs in relation to employment effects is merely a realisation of the failure of previous management regimes – ITQs have merely made the problems transparently and painfully obvious.

Administrative performance

Monitoring and enforcement issues have been briefly discussed above. OECD (1997a) states that: "Higher enforcement costs and or greater enforcement problems occurred in 18 fisheries (of those surveyed) to five that experience improvements. Enforcement proved particularly difficult in high value fisheries, in multispecies fisheries, and in transnational fisheries." In addition, "Under-reporting of catch and data degradation was documented in 12 fisheries, but improvements were made in six fisheries".

Administration costs in the United States surf clam and ocean quahog fishery fell drastically when ITQs were introduced (Buck, 1995). The previous management system had very stringent management regulations which resulted in high enforcement costs. The ITQ system resulted in a considerable reduction in monitoring requirements. However, according to Buck (1995), "the National Marine Fisheries Service estimated that the increased monitoring and enforcement costs to cover additional landing ports and vessel observers for the halibut and sablefish IFQ (Individual Fishing Quota) program would be approximately US\$2 million annually, to counter high-grading and by-catch concerns, and to deal with the large fleet and area covered". The number of participants appears to be an important factor in determining whether the enforcement costs are bearable by the government. In Canada, for example, the high cost of enforcement has meant that ITQs have only been applied in Atlantic fisheries where the number of vessels or participants is relatively small (OECD, 1997b).

The creation of property rights encourages a perspective that illegal fishing (*e.g.* catch exceeding quota limits) is a form of theft from other fishers. In several countries this had led to increased co-operation between enforcement authorities and fishers (OECD, 1997a). In the British Columbia halibut fishery, ITQ holders were activity involved in enforcement and assisted in identifying offenders. In 1993 and 1994, quota owners in New Zealand voluntarily paid for added enforcement in the rock lobster fishery.

Social performance

Theory suggests that the ITQs will place additional pressure on small-scale fishers, and prompt their exit from the fishery. As quotas can be bought and sold, it is argued that the larger companies will use their size to buy up significant proportions of the total quota. OECD (1997a) found little evidence to support this contention: two of the fisheries surveyed showed a reduction in the numbers of small fishers, whereas five showed an increase. However, there was evidence of consolidation of quota ownership – OECD (1997a) documented consolidation of quota in 12 fisheries, but five showed contrary evidence.

Shortly after the ITQ system was introduced in New Zealand, there was a significant change in the structure of quota ownership. Some fishers left the fishery; others sold their quota to processors in exchange for on-going contracts to lease quotas from them. In 1987, the top three quota holders held 29 per cent of the quota by tonnage – by 1994 they held 44 per cent. This partially reflects the economies of scale in fish processing and the rationalisation of capacity amongst larger companies (OECD, 1997b). New Zealand has placed limits on quota concentration, with the more stringent limits relating to inshore stocks (*e.g.* red rock lobster).

In Iceland, the ownership of fishing enterprises has changed in recent years, due to the economic pressures created by lower catches – enterprises have merged and efficiencies have increased. The resultant concentration of quota ownership has created some concerns. In April 1998, the Icelandic Parliament passed legislation specifying limits on the degree of quota concentration. Under this legislation no fishing enterprise may:

- control more than the equivalent of 12 per cent of the annual quotas for cod and haddock (for other species the maximum limit is 25 per cent); and
- control, in value terms, more than the equivalent of 10 per cent of the entire quotas allocated for all species.

Creation of class divisions are evident in Icelandic fisheries. As mentioned above, only vessel owners with a valid fishing license may own quotas. In addition, fishers working on these vessels are increasingly being required to pay for part of the costs of leasing quotas. Fish workers' incomes are calculated as a share of the catch value, after costs like quota leasing. Collusion between the vessel owner and the buyer of the landed fish can be used to conceal the subtraction of the leasing price prior to payment for fishing crews (Eythórsson, 1996). This is also alleged to have occurred in the case of vertically-integrated fishing companies, where the landed price is a purely notional value that can be fixed by the company. Eythórsson (1996) notes that fishers in Iceland refer to this as a "feudal" system. Low income fishermen are referred to as the "tenants" while quota owners are the "lords of the sea." To address these concerns, in March 1998 the Icelandic Parliament established a Pricing Authority for Catch Prices. The Authority monitors fish prices and the settlement of the shares paid to fishers, with the aim thereby encouraging proper and fair settlement of catch shares. It is too early to assess the success, or otherwise, of this new authority.

"In a broad sense, an ITQ program increases the relative bargaining strength of whomever controls the ITQs due to the wealth ITQs represent and the necessity of ITQs as a prerequisite to fish" [Squires *et al.* (1995)].

In the United States, Buck (1995) notes that fishing companies have attempted to change the basis for setting wages. In the surf clam and ocean quahog fishery, crew shares have been reduced in most cases. There are counter-examples, however. Some are optimistic and consider that crewing jobs will often be better paid under an ITQ system. The Deepsea Fisherman's Union of Seattle supported the creation of the of ITQs in the Alaska halibut and sablefish program for this reason.

Concluding comments

This paper has attempted to summarise some of the characteristics of individual transferable quota (ITQ) systems in OECD countries. The degree of sophistication of the systems perhaps reflects perceived need within each country for institutionalised trading systems. Two countries have made widespread use of ITQ systems: Iceland and New Zealand. Iceland has just established a quota exchange to facilitate the

trade in quotas. New Zealand operated an exchange in 1986 and 1987, but it ceased operation when the exchange could not become self-funding. The Producers' Organisations in the Netherlands provide a institution for quota trades between fishers to be facilitated. Systems in other countries rely on direct trades and brokerage firms.

The number of participants in the ITQ markets is quite limited and trade normally takes place between vessel owners. Reasons for this include the initial quota allocation procedure and restrictions on quota ownership. The systems often have quite complex arrangements in order to address the problems in multispecies fisheries. In addition to options for trading and leasing, there are numerous ways of transferring part or all of ITQ rights, in order to cover catches. In one country at least, the administrative and compliance costs were sufficient to encourage a simplification of the system.

There has been resistance to the use of ITQs, and there will continue to be so. The primary reason is that, when applied to an overcapitalised and over-fished fishery, ITQs will result in a rationalisation of fishing activity. Governments have tended to ease the transition by allocating quota to existing participants and choosing to overlook the rights of the rest of society in obtaining a fair return for the free allocation to a select group of individuals of what was previously common property. This may not be possible for other systems where the possibilities forgone by (or the costs to) the rest of society are more easily conceived and understood.

NOTES

1. OECD (1997a) found that: "TACs management clearly results in over-capacity, shortened fishing seasons and fluctuating landings." Furthermore: "over-exploitation was generally not prevented in fisheries managed by TACs."
2. This right is enshrined in Part V of the United Nations Convention on the Law of the Sea 1982. The Convention entered into force on 16 November 1994.
3. For example, the Magnuson-Stevens Fisheries Conservation and Management Act 1996 mandated the preparation of a report, to be completed by 1 October 1998, to investigate, among other matters: "mechanisms to prevent foreign control of the harvest of United States fisheries under individual fishing quota programs..." Source: Magnuson-Stevens Fishery Conservation and Management Act, US Public Law 94-275.
4. Deemed values are set at a level to remove the economic benefit of harvesting fish other than with the authority of ACE.
5. See: i) Dubbink, W., and van Vliet, M. "The Netherlands: From ITQ to Co-management: Comparing the Usefulness of Markets and Co-management illustrated by the Dutch Flatfish Sector". In OECD (1997c); and ii) Annala (1996).

Annex 1

**FISH SPECIES OR SPECIES GROUPS THAT ARE MANAGED
WITH TRADABLE QUOTA SYSTEMS**

Australia

South-east trawl, Southern bluefin tuna, pearl oyster (West Australia, Northern Territory), Southern abalone (Tasmania and Victoria).

Canada

Offshore lobster, Midshore snow crab St. Lawrence, Scotia Fundy herring 4WX, Offshore groundfish, Midshore groundfish, Atlantic Newfoundland inshore, S. Gulf of St. Lawrence trawl 4T < 65', Scotia Fundy inshore trawl, Offshore scallop, Northern shrimp.

Iceland

Capelin, cod, dab, deepwater shrimp, Greenland halibut, haddock, Icelandic scallop, Icelandic summer-spawning herring, long rough dab, nephrops, offshore shrimp, plaice, redfish, saithe, witch, wolffish.

The Netherlands

Sole, plaice, cod, whiting.

New Zealand

Barracouta, blue cod, bluenose, alfonsino, spiny (red) rock lobster, elephant fish, flatfish, grey mullet, gurnard, hake, hoki, hapuku and bass, john dory, jack mackerel, ling, blue moki, oreos, orange roughy, paua, packhorse rock lobster, red cod, scallop, dredge oyster, school shark, gemfish, snapper, rig, squid, stargazer, silver warehou, tarakihi, trevally, blue warehou.

United States

Mid-Atlantic surf clams and ocean quahogs, South Atlantic wreckfish, sablefish, halibut.

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Part II

**OBSTACLES TO THE IMPLEMENTATION OF TRADABLE
PERMITS: LESSONS FROM SPECIFIC PROJECTS**

Chapter VI

OBSTACLES TO THE IMPLEMENTATION OF TRADABLE PERMITS: THE CASE OF NORWAY

by

Per Schreiner

ECON Centre for Economic Analysis, Oslo, Norway

Norwegian proposals to create domestic tradable permit systems

Emissions of SO₂: an industry proposal

Acid rain is considered a serious problem in Norway. A tax on fossil fuels of NOK 17 per kg is the main instrument currently used to reduce SO₂ emissions, but this tax is combined with regulation of the sulphur content of several fuels. The tax rate is probably well above the marginal damages, but the high rate is justified by the lack of broad coverage of the tax.

If one could activate abatement measures in industries that are tax exempted in a cost-efficient manner, the marginal cost of abatement might be brought down to about NOK 5-10 per kg. However, the cheapest potential is in processing industries that compete in international markets with competitors who face few or no restrictions on their emissions of SO₂. A tax on emissions in processing industries is therefore anticipated to lead to significant reductions in their production in Norway.

An international protocol sets a cap for Norwegian emissions of SO₂ in 2000 and thereafter. The Confederation of Norwegian Business and Industry in 1994 became concerned, partly that some measures would be introduced that would hurt the processing industries, partly that it would be necessary to activate costly measures in other industries to compensate for the continued emissions from the processing industries. They therefore engaged ECON to work with them to prepare a proposal on tradable permits for SO₂ emissions (ECON 1995). The main features of this proposal were:

- All emissions of SO₂ would be covered by system.
- The authorities set an annual cap for total emissions of SO₂ in Norway.
- Within this cap, companies could partly buy and partly receive for free an allocation of emission permits. Permits would be issued for several years in advance.
- The free allocation of permits would be based on emissions in a base period (grand-fathering).
- The permits would be valid for a fixed amount of emitted SO₂, or for distributing fuels or raw materials that lead to a similar amount of emissions of SO₂, when used.
- At the end of the year, companies would have to deliver permits (which would then be invalidated) for the quantities emitted or sold.
- The permits could be traded between companies or be banked for later years, but not used before the year for which they were issued. In case of closure or lowering of production, companies could sell permits in their possession, whether they were bought or received for free.

Some emissions would have to be monitored by smokestack measurements, but most would be monitored by keeping track of the sulphur content of inputs and outputs (mass balances). Correspondingly, the obligation to deliver permits would mainly be placed on importers and wholesalers, just as is the case in the taxation of petroleum products. That is why it would be simple to include all emissions in

the system. The sellers of products that contain sulphur are assumed to include the cost of permits in their product price when the products are used in Norway. Selected buyers of products may be given preferential treatment by allocating them permits free of charge. When products containing sulphur are sold outside of the country, or when sulphur is permanently “fixed” in products or waste, permits would be refunded.

A fundamental condition for cost-efficiency of a permit system is that trades in permits not influence marginal damage costs of the emissions. This condition is not generally fulfilled with SO₂ emissions, as damage costs depend on the location of the emission. However, most of this damage is related to acid rain, so only a few, small regions with high concentrations in the air would pose a problem in Norway. Less than 10 per cent of the sulphur in the precipitation in Norway derives from Norwegian emissions. The remainder stems from neighbouring countries. A redistribution of Norwegian emissions, therefore, influences only to a negligible extent the marginal costs in various localities.

Emissions of greenhouse gases

Introduction

In June 1998 the Government stated that CO₂ taxes, already high by international standards, would be the main instrument to reduce Norwegian emissions of greenhouse gases. However, the Storting rejected a proposal to include all CO₂ emissions in the tax base.

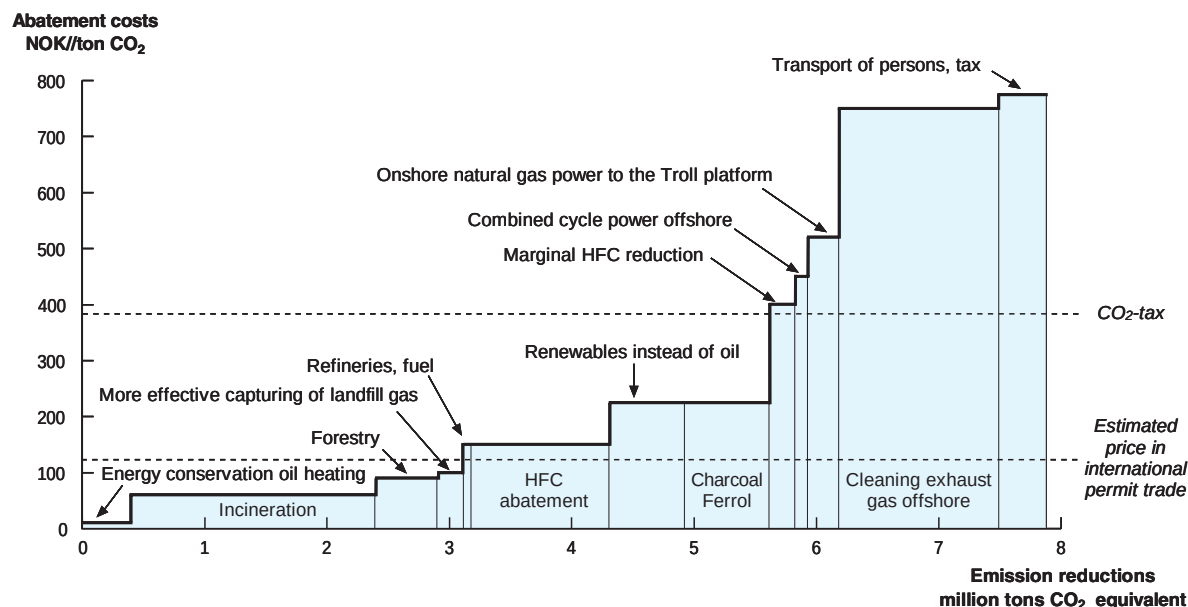
The Government had estimated that the proposed amendments to the CO₂ tax would reduce emissions by little more than half a million tons of CO₂ equivalent. In comparison, a newly adopted tax on waste is expected to reduce emissions of methane by another 1-1.5 million tons of CO₂ equivalents. In the proposal, the CO₂ tax rates were to continue to vary significantly, from NOK 384 per ton CO₂ for oil and gas extraction, and on some oil products, to NOK 100 per ton on the bulk of mainland emissions. To smooth the transition to a tax regime for the industries that are currently exempt from the tax, the Government proposed to compensate the fishing fleet, air transport, and some enterprises in the processing industry for their increased tax payments over a period of ten years.

The Government acknowledged that the wide variation in the tax rates would not be cost-efficient. It intended to compensate for this by using additional instruments that would reduce emissions by some 2-3 million tons. In total, the planned measures would reduce Norwegian emissions by less than half of what is required by the Kyoto Protocol.

This is to demonstrate that: i) in spite of statements to the contrary, new taxes are likely to play a minor role in the additional reduction of the emissions of greenhouse gases; ii) the Government, with the support of the Storting, intends to cover about half of the Norwegian commitments by using the flexibility mechanisms allowed for in the Kyoto Protocol.

An internationally-harmonised CO₂ tax imposed at a rate high enough to stabilise global CO₂ emissions would not stabilise Norwegian emissions. One important reason for this is that there will be steep growth in the emissions from extraction and transport of oil and gas that are (for the most part) exported (to countries that switch away from coal, and thereby reduce their emissions of CO₂). In addition, the costs of implementing further measures to reduce emissions of greenhouse gases are high, partly because of the high CO₂ tax already in place. (See Figure 1 – the CO₂ tax is increased somewhat since this figure was developed.) Norway is in an exceptional position in that as much as 70 per cent of stationary energy use is generated by renewables (hydro-electric power). This implies that there is little potential for reducing emissions cheaply by modernising or refuelling existing electricity-generating plants. At the same time, the potential for expansion of hydro power is also limited.

A consequence of high expected costs is that the importance of cost-efficiency of the overall climate policy is generally accepted. Furthermore, all parties, including most environmentalists, realise that part of Norway's obligations to limit emissions will have to be satisfied by measures in other countries under some mechanisms for international flexibility that are allowed for in the Kyoto Protocol. A domestic tradable permit system would facilitate a link to an eventual international system.

Figure 1. **Costs and Potentials of Climate Measures in Norway**

Source: ECON report 24/98.

ECON proposal

This proposal is being developed by ECON, with the support of the Confederation of Norwegian Business and Industry, Norsk Hydro and Statoil. It is an outgrowth of the SO₂ proposal presented above. A major share of all greenhouse gases covered by the Kyoto Protocol is to be included, and it is intended to replace all other measures to curb emissions of greenhouse gases, including the existing CO₂ tax. This is to be achieved by combining the regulation of major point sources with regulation of importing and wholesaling of products. Different gases included in the Kyoto Protocol are weighted together by their Greenhouse Warming Potential (GWP).

Important economic interests are involved in the CO₂ emissions trading system, as the expected price of permits related to the product price is much larger for CO₂ than for SO₂, and the quantities are also much larger. Another difference is the smaller potential for abatement of CO₂ emissions than for SO₂ emissions. With a high price on emission permits, the emissions of SO₂ per produced unit can be reduced substantially. That is so far not the case for CO₂. When modern techniques are employed, further reductions have to come via changes in the product mix. Then, changes in relative prices become important.

Labour Party proposal

This proposal covers only those industrial activities not covered by the existing CO₂ tax. Thus, it is intended to supplement the CO₂ tax system, and it was developed as an alternative to the Government proposal to extend the tax to these industries. All permits are to be allocated free of charge, but the number of permits would be limited. The proposal contains an indication that industries would have to cover "their share" of the reduction of greenhouse gases. That would imply a reduction of their emissions by 40 per cent compared to 1990. If they were not to achieve such a reduction, they would be allowed to acquire permits from abroad.

Main features of the proposals

Domestic permits are derived from allowances

Previous programmes of tradable emission permits have taken one of two forms: trading in *credits* or trading in *allowances*. Credit trading allows emission reductions above and beyond a pre-specified legal requirement (a source-specific cap) to be certified as tradable credits. Credit programmes focus on specific sources or projects. In the US, the only country with extended experience with trading systems, allowance trading systems have performed well, whereas credit-trading systems have generally not performed to expectations.

The Norwegian proposals are built on trading of permits that derive from allowances. Allowance trading starts by defining an aggregate emissions cap. The emissions authorised by this cap are then either allocated or auctioned as emission permits to eligible parties. The permits are entitlements, based on a contract with the authorities. They are not to be considered to be property rights.

A permit allows for the one-time emission of a fixed amount of greenhouse gases (tons, or fractions of a ton, of CO₂ equivalents as measured by the GWP). This is distinct from a source-specific cap which regulates the maximum aggregate amount of emissions in a certain time period (usually a year), but is kept in place for several periods. A permit that is not used can be banked (saved) and used later. When emissions, instead, are regulated by individual caps, emissions lower than the cap allows cannot normally be traded or credited for later transgressions of the cap. That is only possible when Emission Reduction Credits (ERCs) are issued by the authorities, after proper case-by-case documentation of over-fulfilment of emission reduction obligations.

The Kyoto Protocol defines national obligations as the average over the five years 2008-2012. This averaging allows for flexibility of timing of abatement measures, and it smoothes out the impact of particular economic and natural occurrences. Still, few countries would like to leave it to the last year to make sure that they will meet their obligations. Therefore, countries will most likely want to regulate their emissions year-by-year.

This can be achieved in a tradable permit system by letting each permit have a validation date. Each year the number of permits that are validated corresponds to the target emission level that the authorities have set for that year. This target can be the average set in the Kyoto Protocol, or it can be set high for the first year, and decline throughout the period, so that the overall average is the same. Saving permits for later years can be unrestricted, whereas borrowing permits from future years has to be severely restricted, or ruled out, to avoid undermining the credibility of the system.

Assuming an annual regulation period at the end of the year, emissions over the year are recorded for each polluter, who is obliged to submit permits for the emissions for which they are responsible. A corresponding number of permits are then destroyed. The sanction for not being able to submit a sufficient number of permits would be set so high in relation to the market price of permits that it would not pay to be "short".

Permits granted for abatement abroad

The system would be domestic, but an important motivation is that it would readily lend itself to benefiting from the four co-operative implementation mechanisms that the Kyoto Protocol authorises: bubbles, emission trades, joint implementation and the Clean Development Mechanism. For Norwegian companies, knowing that their abatement costs generally are much higher than the costs of their competitors in other countries, it is important that they be able to fulfil their obligations in Norway by acquiring permits abroad. If an international market then develops with a uniform price of permits, Norwegian firms can expect to be able to compete on a level playing field.

With international emission credits available, and with unrestricted access to international trading, the cost of activities abroad would set the price of permits. The domestic price of permits would then be the same as the international price. Ideally, marginal abatement costs would tend to become equal in, and between, all countries. This would promote global cost-efficiency in restraining the emissions of

greenhouse gases. Even in a less-than-ideal situation, the ability of enterprises to acquire permits by investing abroad would dampen domestic prices of permits. High transaction costs (for instance, fees that are not cost-related, or restrictions on the activities within the flexible mechanisms) would reduce global cost-efficiency.

Comprehensive coverage

- Universality as a guiding principle

The great advantage of a tradable emission permit system is that it can combine equal marginal abatement costs with preferential treatment to certain branches of industry, in order to avoid distorted competitive conditions. Differentiating tax rates, which is the established procedure in Norway, induces different marginal abatement costs across branches.

However, this cost-efficiency is obtained within the trading system only. The problem remains of how to make marginal costs equal across systems, and to keep them equal. The “simple” way to achieve overall cost-efficiency would be to let the system cover all emissions of greenhouse gases. Then, one could be reasonably sure that the marginal cost of reducing emissions would become equal across sources and gases.

This is not possible, however. Some emissions cannot be monitored well enough, or there is lack of will to include them because of the subsequent fall in revenues from the SO₂ or CO₂ taxes. That is why the Labour Party proposal would include only industries which are currently exempt from the tax. In this case, overall cost-efficiency is difficult to achieve. One can set the total cap for the tradable system so that the estimated permit price initially is equal to the estimated marginal abatement cost outside the system. However, this equality can only be maintained by regularly changing the cap to take into account changes in patterns of demand and technology.

- Monitoring emissions: combined “downstream” and “upstream” systems

In most cases, there is no need to measure emissions of greenhouse gases at the smokestack; using mass balances that keep track of the composition of inputs and outputs is both cheaper and more precise. As long as a company has a record of well-kept and undisputed accounts, reliable calculation of permit liabilities should cause no more difficulties than the calculation of tax liabilities. As a consequence, permits can apply directly to emissions at the source (downstream); or they can apply to the distributors of fuels and other products (upstream), as is the case with the current Norwegian SO₂ and CO₂ taxes. Tradable permits can therefore control any emissions that can be taxed.

In a “downstream” system, there is in practice a lower limit on the size of the source that can be regulated. Household emissions and emissions from most vehicles cannot be captured. In Norway, these represent more than half of total CO₂ emissions. In principle, nearly all emissions can be controlled in an “upstream” system. However, both the major enterprises and authorities may find it preferable to maintain the direct contacts developed over years of environmental regulations, based on “command and control” approaches. Authorities can then keep themselves updated on initiatives and technological options, and large companies will prefer to manage permits and emissions themselves, rather than being exposed to the pricing policies and strategic behaviour of their suppliers. As a matter of fact, since the large companies in processing industries generally import their own raw materials, this does not make much of a difference, since it is only relevant for their use of fossil fuels, whether for energy or feedstock.

The ECON proposals combine both “downstream” and “upstream” approaches. This is not expected to cause any administrative or institutional problems. The Labour Party proposal only applies to the processing industry, which consists of large units, and is conceived as a “downstream” system.

In the “downstream” part of the system, one can choose whether to regulate individual point sources or enterprises as a whole. The implications of this choice are not explored. The Labour Party suggests restrictions on the sale of permits in the case of plant closures in single plant communities. This would imply regulation at the plant level.

- Limitations of monitoring

Past emission trading programmes show that high quality monitoring is essential to assure compliance. All parties, government, business and environmental groups, must have confidence in the measurement and monitoring of the individual quantities that are regulated by permits.

The proposed SO₂ trading system is designed for 100 per cent coverage. Still, in the proposal it is suggested that to reduce administrative costs, one may choose to exempt a few very small sources (coal, coke and wood used as fuels) that are not expected to expand in use.

Some emissions covered by the Kyoto Protocol can be estimated satisfactorily on a national basis, but not well enough for each single source. The main exemptions are laughing gas (N₂O) and methane (CH₄) from agriculture, where emissions depends on cultivation procedures (not simply on the use of fertilisers etc.). Methane from landfills may also cause a problem, but it is assumed that calculation methods that are reliable enough to monitor individual landfills can be developed. Table 1 reviews the sources and gases that may be included in a national tradable permit system for greenhouse gases. In total, these comprise well over 80 per cent of Norwegian emissions.

Table 1. Sources and Gases That May Be Included in a National Tradable Permit System

ectors and number of sources	CO ₂	CH ₄	N ₂ O	CF/SF ₆	HFK
Energy Production					
Petroleum extraction (32 fields) ¹	Yes	Yes ³	Yes ³	—	—
Oil refineries (3 plants)	Yes	Yes ³	Yes ³	—	—
Industry					
Pulp and paper (25 plants) ²	Yes	Yes ³	Yes ³	—	—
Chemical raw materials (30 plants) ⁴	Yes	Yes ³	Yes	—	—
Cement etc. (5 plants)	Yes	—	—	—	—
Metals (30 plants)	Yes	—	—	Yes	—
Other production	Yes	Yes ³	Yes ³	—	—
Other Sectors					
Agriculture	Yes	No	No	—	—
Transport ⁵	Yes	—	—	—	—
Service sectors	Yes	—	—	—	Yes
Households	Yes	—	—	—	—
Waste landfills (200 in operation)	—	Measuring/ incentives good enough?	—	—	—
• Involves a major loss of tax revenue. • Involves a moderate loss of revenue. • The costs and accuracy of monitoring methods need to be investigated further. • Includes production of fertiliser, petrochemicals, and onshore processing of oil and gas. • Aviation, sea freight transport, offshore supply transport, fisheries.					

Overall cost-efficiency

To reduce the cost of controlling the emissions of greenhouse gases, new technologies will have to be developed, and considerable capital investment will be needed. There are long lead times before such initiatives can yield results, and the pay-back period may be long. Predictable and transparent framework conditions increase the chances that enterprises will spend resources to undertake these investments, and not resort only to short term measures. The total cap on emissions from sources participating in the tradable system (and other crucial elements of the control programme) should not be changed often. In addition, the rules for changing them must be known to, and accepted by, the major participants.

A major concern will be to equalise marginal abatement costs, both inside and outside the system. Even if the ideal is a universal scheme, some emissions have to be controlled by other measures. When a tradable permit system is developed, estimates have to be made of the marginal costs ($\approx$ price of permits) for various levels of the total cap on emissions from sources participating in the scheme; as well as for sources outside the scheme. In setting the overall cap for later years, these marginal costs should be monitored. They may turn out to be quite different, both because the original estimates were mistaken or because of new developments in technology and demand patterns.

If the tradable permit system is nearly universal, the differential between the marginal costs of abatement inside and outside the system is of little concern. If, however, for some reason, major emissions remain outside the scheme, discrepancies between marginal costs can become large. Still, one should hesitate before tampering with the scheme. Adjusting the cap of the system should wait for this major readjustment. If, however, the system is put in place, and emissions are regulated well before 2008, one might consider some adjustment connected to the transition from pre-Kyoto to Kyoto obligations.

The issuance of permits

- Both by auction and by free allocation

Two considerations are guiding the choice between auctioning and allocating for free ("grandfathering"). The first one seeks to respect the Polluter Pays Principle (PPP) in its broadest sense (*i.e.* the need for polluters to pay the full value of any environmental externalities that they generate). The second seeks to protect the competitive position of Norwegian industry. Strict application of the PPP, and not considering transition costs or the transient aspect of the loss of competitiveness of some industries, implies that all permits should initially be sold. However, in the current Norwegian proposals, the free allocation of permits to offset the undesirable impacts on companies exposed to competition from abroad is a central element.

In principle, free allocation should be used only to offset a *temporary* distortion of competitiveness of some industries caused by Norway moving ahead of other countries in introducing economic instruments to curb emissions. It would not be cost-efficient to give preferential treatment to industries that *permanently* lose their competitive edge because of the Kyoto Protocol. At a maximum, it should therefore be a question of slowing down the transition process. However, the political reality may be different, and the distribution between auctioned and freely allocated permits is for the politicians to decide. A key issue in Norway will be whether to allocate permits for free to planned gas-fired power plants.

- Timing

The timing of issuing permits should be guided by concerns about confidence, predictability and liquidity. To facilitate company planning, permits for future years must be in circulation. A company should have the opportunity to choose between solving its emission problem either by investing in abatement or by acquiring permits. As investments may have a life span of at least ten to fifteen years, permits should be available for a similar period. Liquidity is also important as a means to reduce possibilities to exploit market power in the permit market. Confidence that the framework conditions will not be changed at short notice; and that the number of permits to be issued for a certain year will not be changed after trading has started, are crucial for the active participation of major companies in the trading scheme.

The sale of permits also has to be organised so that the market is cleared. That can be achieved by regular auctions, combined with second-hand markets. The auctions themselves can be conducted in a similar way to the sale of treasury bonds. Bidders would be served from the highest bid downwards, until all permits available for sale are sold. The organisation of such auctions needs little attention. What needs more emphasis is what quantities to offer of the various vintages, and how often to hold auctions. Some permits should be sold for the last year of the period, so as to develop a future market, whereas some should be sold only in the year for which they become valid, in order to bring liquidity to the spot market. The respective shares should be decided in consultations with industry and market experts.

- Criteria for the allocation

The free allocation of allowances to pollution sources is a contentious issue, since these allowances can be very valuable. That calls for clear rules that yield little room for discretionary decisions by the authorities. Even so, disputes over the allocation could delay implementation. For example, in the US sulphur trading programme, the distribution of allocations became a political issue, leading to numerous special rules for individual sources. In an earlier US programme, it had also led to failure of the system when agreement could not be reached on the method of allocation.

The (free) allocation should be made according to criteria that are not related to current emissions, otherwise the incentive effect of the tradable permit system will be distorted. In the “upstream” part of the system, where wholesalers are obliged to submit permits, free allocations may be given to emitting companies to compensate them for any permit costs that are added to the price. All Norwegian proposals assume that the initial allocation will be based on emissions experienced in some prior base period. It is necessary here to go back some years in time, in order to avoid punishing companies that started early to reduce their emissions by giving them small allocations. The ECON proposals for Norway would provide an opportunity for such firms to apply for adjustment on the grounds of special circumstances.

Since the allocation of permits may go on for an extended period of time, keeping the base period fixed may contribute to an undesirable freezing of industrial structure. In particular, there is a concern that a “grandfathering” principle will discriminate against new activities. Therefore, various ways to develop a dynamic base, by using a lagged moving average of some years, need to be explored.

The major justification for allocating permits free of charge is that it gives an incentive for emitters to reduce their emissions without at the same time giving them an incentive to move production out of the country (carbon leakage). If a company receives permits for an extended period of years (in the case of the US sulphur scheme, permits were granted in perpetuity), and is allowed to sell them when closing down business, this will give them a strong incentive to move their production and to sell their permits.

Therefore, in the ECON proposals, the allocation period is no more than ten years. Presumably, the returns from selling permits will not be high enough to compensate for the capital loss that most often will be incurred when closing down a profitable operation. This is to be combined with a clear presumption that companies will regularly be allocated new permits, so that as long as the need for preferential treatment prevails, they will always have a stock of permits for ten years ahead (the Labour Party proposal would restrict the sale of permits in the case of closure or downsizing).

When the total cap on emissions from the tradable system has to be reduced (because the environmental goal one seeks to achieve is set higher), the full adjustment should not be made on the number of permits that are sold. Permits which are already allocated for free should not be tampered with, since this would seriously hurt predictability and confidence in the scheme. So, the first opportunity to reduce the number of freely allocated permits is the annual allocation of permits for the tenth year. Because the last year of the Kyoto Protocol is 2012, the question is whether a new stricter Protocol will be in place by 2002. If it is not, the Norwegian authorities could anticipate a new agreement and reduce permit allocations for 2013 and subsequently.

Banking and trading of permits

Banking (saving) of permits is essential for the smooth operation of a permit system. It is especially significant for industries in which major capital expenditures must be made, since it allows individual sources some flexibility in the timing of these investments. Holders of permits should not have to risk that permits will expire – this would only cause a rush to use emissions at the end of the year. In addition, some companies would like to reduce uncertainty by complementing their reliance on the permit market with a “hoarding” strategy.

Reducing the stocks of permits, for instance at the peak of a business cycle, may bring annual emissions above the average annual cap. This does not cause any real problem for greenhouse gases, since it is the *stock* of gases in the atmosphere, not the *flow* of emissions that causes problems. With SO₂ it is similar, as long as local concentrations in the air stay well below safety levels. Neither should the fluctuations

caused by changes in the stock of banked permits cause problems in relation to international obligations. The Kyoto Protocol applies to a five-year average, and presumably future protocols on acid rain will be based on averaging.

The potential savings from trade in permits are large because of the wide variation in marginal abatement costs. Little effort is therefore assumed to be necessary to get the trade going. Brokers and trading intermediaries can be expected not only to enter, but also to create secondary emission permit markets. That is what happened when the Norwegian electricity market was deregulated. The main responsibility of the authorities is therefore to provide proper framework conditions for the permit trading:

- A reliable system of registration so that there is no ambiguity concerning the title to permits.
- Credible monitoring and control of emissions and the submission of permits.
- Avoid interference and regulations that may increase transaction costs.
- Supervision and regulation to avoid accumulation of market power.

The permits would exist as entries in a public register only – no paper is needed except for information. The authorities would recognise as legitimate owners of permits only those who have title to them (according to the register). Therefore, those who acquire permits to cover their obligations must see to it that the trade is recorded in the register. On the other hand, some traders would consider the permits as financial objects, and would acquire them only in order to sell them later. To reduce transaction costs, they would probably avoid the registration fees by not recording the trade in the register. The relation between the buyer and the registered owner would then be a civil-law case, of no concern to the authorities.

Restrictions on transactions would most likely be proposed in connection with closures and down-sizing, as is evident in the Labour Party proposal. The proposals from ECON strongly advise against such restrictions, since they will seriously impair the cost-efficiency of the system. Neither should there be any restrictions against third party investors or speculators. The openness and liquidity of the market stands to benefit from the participation of such investors.

A major concern when creating a market in a small country, like Norway, is the possibility that major actors will be able to exploit their market power. This worry disappears when there is easy access to acquire permits abroad, as is assumed in ECON's greenhouse gas system. That is an additional argument for facilitating the access of individual enterprises to activities that can create permits abroad. Still, some attention needs to be given to the organisation of the market. In general, it should be regulated like any other financial or commodities market.

Institutional and legal requirements

The operation of a tradable emissions system rests on five institutional functions:

- Issue and allocate for free some permits.
- Sell (auction) other permits.
- Keep a register of who has title to the permits.
- Monitor emissions and collect the corresponding number of permits.
- Enforcement and sanctions.

All these functions can be fulfilled by existing institutions. Except for possible transition problems, responsibilities and administrative loads would increase only to the extent that the trading system adds to the number of emission sources being controlled, and is not simply a substitute for existing instruments.

The Norwegian Pollution Control Authority would issue permits and be responsible for the free allocation. It would further monitor emissions and see to it that permits are submitted on time. The Tax Directorate would be responsible for the sale of the permits, since large sums of money are involved. It would also be responsible for enforcement and sanctions, based on reports from the Pollution Control Authority. Preferably, sanctions would be in the form of high fees for excess emissions. The Central Securities Depository should be able to maintain the register. For that, however, the permits would have to be defined as "securities" by the Ministry of Finance.

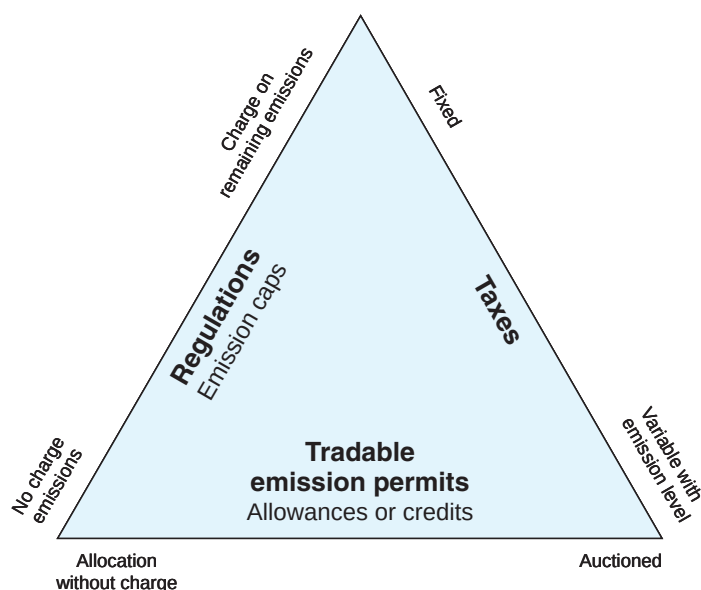
It is important that major elements of the permit trading scheme be sanctioned by law. In that way, enterprises which consider investing considerable sums to benefit from the scheme, can get some assurance of stability, since a law cannot easily be changed. In addition, a law is probably necessary to regulate emissions of greenhouse gases, since it is not clear that they fall under the law that currently regulates pollutants in Norway.

Trading systems as part of a framework of environmental instruments

Overview

There are no clear distinctions among the instruments used to combat undesirable emissions. This complementarity is illustrated in Figure 2. When all emission permits are sold (auctioned), the trading system is similar to a tax, where the rate is adjusted to bring emissions close to a desired level. When all permits are allocated free of charge, it is similar to a traditional “command and control” system, except that the criteria for the allocation may be different from the criteria for setting the caps. When there is a charge on remaining emissions after an emissions cap is satisfied, the charge may be so high that the cap is not an efficient constraint. In this situation, one is close to a tax system.

Figure 2. **Tradable Emission Permits in Relation to Other Environmental Instruments**



Source : OECD.

Criteria for the choice of environmental instruments

Below is a list of criteria that environmental instruments ought to satisfy:

- *Steering efficiency*: Achieving accurately the desired reduction in emissions.
- *Cost-efficiency*: Reduction in emissions is achieved as cheaply as possible.
- *Global efficiency*: Domestic reductions lead to lower global emissions.

- *International flexibility*: Opportunity to benefit from the mechanisms for international flexibility that are introduced in the Kyoto Protocol.
- *Fair distribution*: The burden of abatement is shared in a way that is considered fair.
- *Predictability*: Consumers and producers should get to know their obligations in good time, and be reasonably insulated against surprises.
- *Transparency*: Implications of the system should be readily available and understandable.
- *Dynamic effectiveness*: Incentives should be provided for technological innovation.
- *Polluter Pays Principle*: Application of the broad definition of the PPP should generate new revenues that can be used to reduce distorting taxes.

A system of tradable emission permits scores well on all of these criteria, as the comparisons provided below illustrate.

Combination of several measures

The main obstacle to a universal trading system in Norway is probably concerns about lost tax revenues. The current CO₂ tax rate for several branches of industry is considerably higher than the assumed equilibrium price of permits (Table 2). A universal trading scheme would therefore imply a substantial loss of revenue for the government, even if the free allocation of permits were restricted to emissions that are now exempted from taxes.

However, the legitimacy of a green tax regime rests on the condition that there be a close link between the tax and the undesired externality. The emphasis on revenue therefore corrupts the legitimacy of the existing SO₂ and CO₂ taxes. On the other hand, the use of oil products, presently taxed at the highest rates, induce other externalities that can be taxed instead; and in oil and gas extraction, the authorities have considerable freedom to influence revenue generation.

Table 2. Norwegian Carbon Taxes (Autumn 1998, US\$¹)

	Per litre, kg or SM ³	Per ton CO ₂
Gasoline	0.12	51.8
Mineral oils		
Light fuel oils, auto diesel, etc.	0.06	22.7
Heavy fuel oils	0.06	22.7
<i>Sectors with a reduced tax rate:</i>		
Paper and Pulp	0.03	11.3/9.7
Herring meal and oil	0.03	11.3/9.7
<i>Sectors exempted from tax:</i>		
Air transport		
International shipping		
Domestic sea transport of goods		
Fishing and catching		
Supply fleet		
Coal and coke		
Coal	0.06	24.7
Coke	0.06	18.8
<i>Sectors exempted from tax:</i>		
Coal and coke for energy in Cement and Leca production		
Coal and coke for processing (Ferro alloys, Carbide, Aluminium)		
Oil and gas in the North Sea		
Oil	0.14	54.5
Gas	0.14	61.9

. At the rate of NOK 7.41 per US\$.

A tradable permit system can be very flexible. It may function in combination with other instruments, such as taxes, command and control, and information. On the other hand, such a combination of instruments is not advisable unless it can be designed so that the various elements supplement each other in a predictable way to achieve a desired outcome. That is rarely the case, except where “information” is used as a supplement to other measures.

Two cases are likely. First, the authorities may want to supplement an assumed low price of permits (or a low tax rate) with “command and control” regulations, in order to achieve further reductions in emissions. Second, they may want to supplement the permit system with taxes, in order to provide additional revenues. In both cases, the cost-efficiency of the scheme is destroyed. As a tradable permit system can cover most emissions, it is therefore better to extend that system than to add additional instruments. If the authorities need to increase revenue, they should instead tax some other undesired externality, or choose a tax for purely fiscal considerations.

Comparison with “command and control” (and voluntary agreements)

One essential benefit of a trading system over “command and control” is dynamic efficiency. The former approach allows those who are responsible for emissions the flexibility to determine abatement technology options themselves, including the timing. Since emissions sources are the ones that would make the decision to implement emission-reducing technologies and processes, the possibility to trade allows them the greatest potential to achieve efficiencies.

In “command and control” systems, emissions are generally allowed to occur free of charge. This contradicts the (broad definition of the) Polluter Pays Principle. For some reason, this is not generally regarded as a subsidy to industry that violates international trade agreements.

In addition, it can be difficult to achieve the desired level of emissions in a “command and control” system. For instance, in Norway there is no system to keep track of the sum of emissions allowed by the individual licences (caps), and to keep this sum within the desired limit. Second, even where such a system existed, actual total emissions at any point in time would be well below this level. The individual caps have to allow for cycles in production and emissions, and all emitters will not be at the top of the cycle at the same time. Consequently, either emission control becomes unnecessarily strict, and thus expensive, or the sum of caps has to be set well above the desired level of total emissions.

Only about 40 per cent of Norwegian emissions of greenhouse gases stem from large point sources which can be efficiently controlled with “command and control” measures or voluntary agreements. Regulating the content of pollutants in fuels can reduce some emissions from small sources, such as households and vehicles. This approach has no relevance for the emissions of CO₂, however, since it is impossible to remove CO₂ from these products.

“Command and control” implies extended contacts between emitters and the authorities. Often this leads to mutual confidence and respect, but the basic asymmetry of information cannot be overcome; the system therefore invites companies to spend resources on lobbying and to exert political pressure.

Comparison with taxes

In principle, tax systems and permit trading systems are very similar:

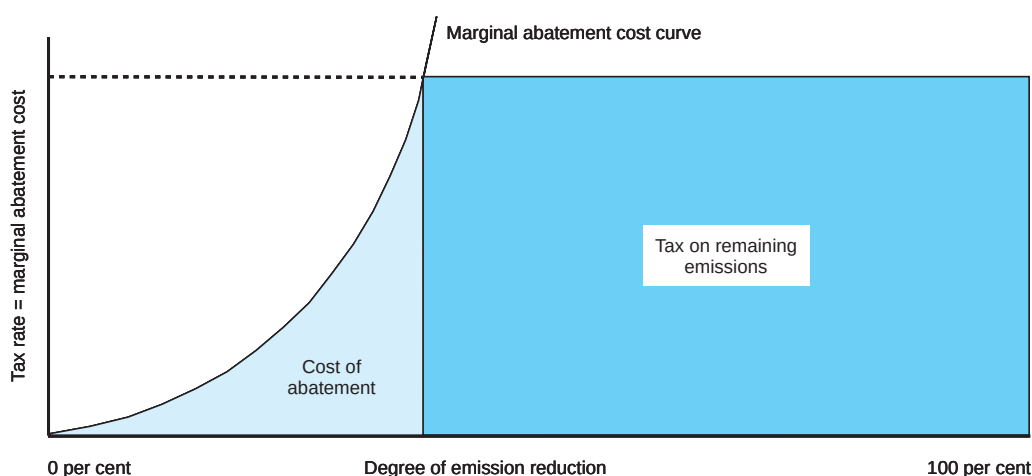
- The polluter pays both for abatement costs and for remaining emissions.
- Both instruments yield approximately the same revenue when applied to the same sources.
 - The price of permits will vary with changes in demand, technology and competition from other producers or products.
 - The tax rate can be adjusted to yield emissions close to the cap that is decided. It will then equal the market price of permits. That is next to impossible to achieve in the short term, but over time, the rate can be adjusted to reflect changes in demand, technology and competition from other producers or products.
- How the revenue can be spent is independent of how it is derived.

- Preferential treatment of particular industries can be achieved with both instruments.
 - In a tax system, one uses differential tax rates, exemptions and refunds or cash transfers (subsidies).
 - In a tradable permit system, some permits are allocated free of charge according to similar rules.

However, there are also important differences:

- The great advantage of a tradable permit system is that it allows for preferential treatment without destroying cost-efficiency. Tax exemptions nullify economic incentives to reduce emissions, whereas incentives to reduce emissions are maintained when tradable permits are allocated free of charge. The need for compensatory measures is illustrated in Figure 3. If other countries do not rely on taxes to reduce emissions, the financial burden on Norwegian companies will be much higher than that of their competitors, even when their abatement costs are the same (assuming that the levels of all other taxes facing the two sets of firms are comparable). This is particularly pressing with CO₂, since the scope for reducing emissions per unit of production is very limited. Then the right hand rectangle becomes large, compared to the abatement costs.

Figure 3. The Relationship Between Abatement Costs and Tax Payment

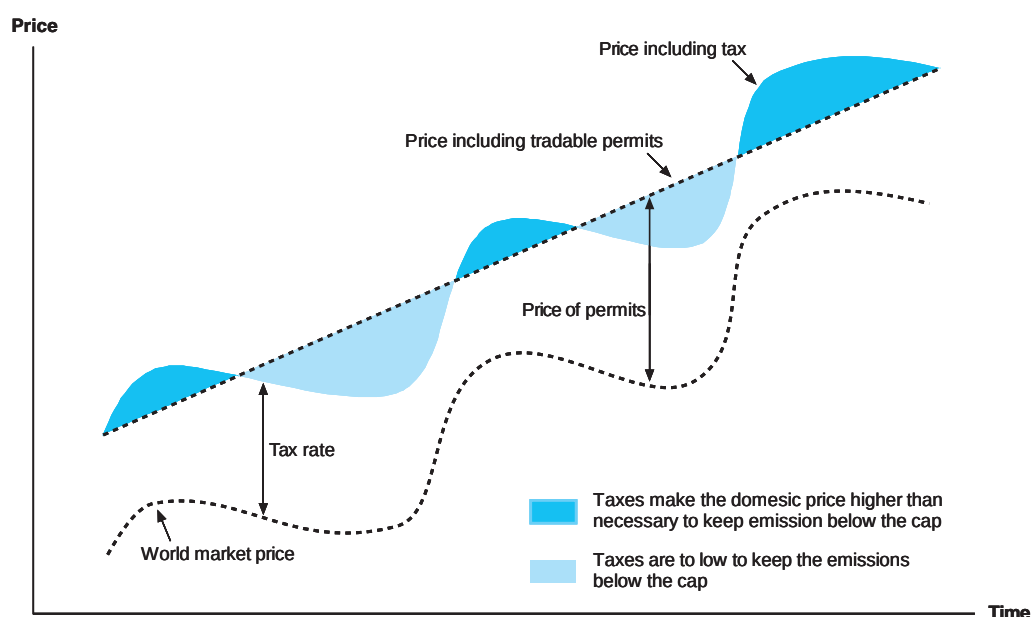


Source: ECON report 331/95.

- Subsidies (cash transfers) are not a realistic alternative for preferential treatment of certain industries, since they are sensitive to the rules in GATT/WTO and ESA. Permits allocated free of charge correspond to the emission caps that are the standard instrument in environmental policies in most countries. They should therefore not be in conflict with these rules.
- Tradable permits have high control efficiency – total emissions will be close to the number of permits issued, at least over a period of years. This reduces the need for safety margins to take care of forecasting errors. With taxes, the authorities have to guess what the emissions will be given a certain tax rate. Frequent adjustments of the rate are politically and administratively cumbersome.
- Control efficiency also reduces need for safety margins to make room for growth and cyclical variations in production and demand.

- Flexibility and self regulation yields a more optimal adaptation:
 - Economies of scale of abatement projects can be exploited more easily, and premature scrapping can be avoided.
 - This may reduce government revenues, but it reflects a more cost-efficient accomplishment of the obligation to reduce emissions.
 - Flexibility over business cycles is cost-efficient.
- Tradable permits tend to dampen price fluctuations of goods that are subject to regulation. This is illustrated in Figure 4.

Figure 4. **Different Price Trajectories With Taxes and With Tradable Permits**



Source: ECON Report 331/95.

- Predictability and confidence is important for industry. There is distrust within the business community of the fiscal authorities, particularly related to taxes. Even if a tax is called a “green” tax, in practice it is decided upon from a revenue-raising perspective. The fate of both the SO₂ and the CO₂ taxes illustrate this point. A tax rate (in Norway) is set for a maximum of one year at a time, and it introduces an element of uncertainty into business calculations that is difficult to hedge against. The price of permits is also uncertain, but that particular market uncertainty can be hedged by buying futures, etc. Economists often overlook the importance of distorted decisions in business because of perceived uncertainty.
- The incentive to initiate technological development should, in principle, be the same for tradable permits and taxes. However, the perceived risk that reduced emissions may be met by higher taxes to maintain a given revenue, may dampen the willingness of companies to take such initiatives when taxes are used.

Obstacles to implementation of the proposals

Introduction

Obstacles to the introduction of tradable permits systems in Norway (for SO₂ and CO₂) are linked more to a lack of legitimacy and general acceptance, than to institutional or practical hindrances. To a large extent, resistance is caused by a lack of familiarity with a new concept, and one can overcome this through consultations and co-operation with stakeholders.

The main obstacle, however, seems to be concern about loss of revenue from existing SO₂ and CO₂ taxes. Such a loss would come about if permits are allocated free of charge to more industries than those which are presently exempted from the taxes. It is noted that this approach is *not* part of the current proposals. However, the loss of the opportunity to impose a particularly high CO₂ tax on the petroleum industry would reduce revenues considerably. So would the increased cost-efficiency in the abatement of emissions of SO₂ that a trading system would bring about.

Second, much of the debate is centred on the possibility of allocating some permits free of charge, and on why one should give preferential treatment to industries that would be particularly hurt by these measures to reduce emissions. The surprising thing is that the arguments against preferential treatment were stressed only after the idea of tradable permit systems was introduced. The existing system of differentiated tax rates has largely been ignored in this subsequent debate.

A climate of suspicion and distrust would be a major obstacle to the introduction and good functioning of a trading system. Systems are not credible if they are considered to be vehicles for evading, rather than complying with, environmental targets. The system must be legitimate, predictable, and transparent. This applies both to suspicions held by the authorities or by the environmental community, as well as to mutual mistrust between trading partners. Without a broad sense in the society of ownership in the trading system, business expectations that the system may soon be changed will reduce their interest in engaging in it.

Academic criticism

Preferential treatment distorts industry structure

Some economists from academia and research have expressed doubts about the arguments typically used to promote tradable permits in Norway. They agree that a permit system without free allocation is very similar to a tax system with a uniform tax rate, but maintain (without analysis) that a trading system will be more complicated. Their main concern, however, is that the allocation of permits free of charge is not cost-efficient, and that the trading system cannot maintain cost-efficiency and reduce carbon leakage at the same time.

Their objection to the allocation of free permits is partly based on a general objection to preferential treatment for specific industries, and partly on a belief that the revenues from "green" taxes can be used to reduce other distorting taxes or to support industry in other ways. However, they have been much more vocal against the introduction of a tradable permits system than they have been against the differential rates in the existing CO₂ tax system. One reason for this may be that they see the introduction of free permits as a set-back in the struggle for a complete application of the Polluter Pays Principle, and they may be afraid that once granted, a free allocation will be difficult to reverse.

These critics also maintain that preferential treatment distorts the future structure of industry, and increases the costs of keeping emissions within the cap. When some industries are exempted, more expensive measures in other industries and with consumers must be implemented. In this calculation, these economists do not put much emphasis on transition costs. One should be reluctant to try to avoid transition costs, they say, except possibly to dampen those costs related to particularly abrupt changes.

Further, these critics are not concerned about carbon leakage, arguing that the responsibility of Norwegian authorities is to ensure that Norway fulfils its obligations as cheaply as possible. Nor are they concerned about the possible loss of legitimacy of climate policies if people should discover that the net impact on global emissions of Norwegian efforts becomes nil or negative. Another argument is that the impact of Norwegian policies on global emissions is already negligible.

The proponents of a trading system have two basic arguments. First, preferential treatment may be rational in order to avoid a temporary migration of industrial production to countries that are lagging behind Norway in their climate policies (carbon leakage). If Norway wants to be a “front-runner” in the introduction of active climate policies (Green Tax Commission NOU 1996:9), instruments must be chosen that help to convince other countries to follow the Norwegian example. That will not be the case if emissions are simply transferred to other countries. Such transfers would probably also reduce the domestic legitimacy and support both for the measures taken and for an active climate policy in general.

Second, one must realise that politicians will not completely refrain from giving preferential treatment, no matter how much economists may argue. In such conditions, a certain allocation of free permits is much more cost-efficient better than granting tax exemptions or reduced tax rates. As to the possible beneficial use of revenues to reduce other taxes, there is no significant difference in the revenue between a tax system and a tradable permit scheme when allocations free of charge correspond to present tax exemptions or refunds.

Allocation of permits free-of-charge does not prevent carbon leakage

If tradable permits are allocated as a flow of permits in perpetuity, the critical economists point out that incentives for enterprises to move production is the same as if they have to expect a tax whose rate is adjusted in step with changes in the price of permits. The profit rate is significantly different, but at the margin companies will stand to profit the same amount by moving. In the one case, they can sell their permit allocations for an amount corresponding to the discounted value of future permits. In the other case, they save the same amount in discounted future taxes.

The main justification for the allocation of permits free of charge is to avoid distortion of the competitive position of some domestic industries. If other countries introduce corresponding measures exempting their own industries, however, this justification disappears. Therefore, the free allocation should be for a limited period of years. After that, most of the profit of selling off permits will be eroded by the capital loss involved in moving or abandoning production equipment, so that there would be little incentive to move profitable production elsewhere. The period can be rolled over as long as deemed necessary, so that enterprises can expect to always have at their disposal permits for a period of at least this length of time. Ten years would probably be long enough for enterprises to follow a cost-efficient emissions strategy, and at the same time short enough to not present an incentive to move profitable production activities to another country.

To the extent that they admit the need to accommodate politicians' demand for ways to offer preferential treatment to some industries, the critical economists also recommend cash transfers or compensatory reductions in other taxes and charges. To this, the proponents respond that such measures will meet objection from Norway's trading partners. Whereas granting free emission permits is the standard procedure in environmental policies in all countries, subsidies, and preferential tax rates not directly connected to a particular disadvantage are generally considered to distort competition, and are therefore typically disallowed in trade agreements.

Government objections

The present Government has generally been against the introduction of tradable permits systems. The main argument is that the allocation of permits without charge leads to loss of revenue, and is also a deviation from the (broad definition of the) Polluter Pays Principle. It is further argued that an allocation of permits based on past performance would reward poor performers. However, the Government also expresses concerns about higher administrative costs and the potential of big companies to exploit market power in the small Norwegian permit market. There is also a general dislike of the idea that anyone should be able to buy a license to pollute.

The Government arguments strongly support the use of economic instruments in environmental policies, but maintain that allocating something free of charge would effectively turn the Polluter Pays Principle “upside down”. However, it is not generally correct to say that free permit allocations would make the recipients “free-riders” on the efforts by the rest of the society, or that this would reduce the

incentive for innovation (loss of dynamic efficiency). When the permits can be sold, the incentive to reduce both present and future emissions is the same for all participants. Only when the total allocation of permits is so large that there is no need for the participants in the system to reduce their emissions will the price of permits be zero. In this case, the incentive to reduce emissions will disappear, so that the whole burden of reducing emissions has to be born by the remaining sectors in the society.

In the Labour Party proposal, the number of permits would correspond to about 60 per cent of emissions in the base period. That should imply a significant price for the permits, even if all permits were allocated free of charge. A proper procedure would be to set the number of permits so that the estimated price of permits equals the estimated marginal abatement cost outside the permit system. Then one would achieve cost-efficiency, despite preferential treatment of the recipients of permits.

What is surprising is that, in spite of the arguments against a preferential treatment of some industries, and with the emphasis on cost-efficiency, the Government does not propose either a uniform rate of the CO₂ tax, or a tax on emissions of greenhouse gases other than CO₂.

Scepticism of environmentalists

Broadly, the Government position described above also reflects the attitudes of the environmentalists, but perhaps with a certain lag. There has been a traditional prejudice against economic reasoning and economic instruments in environmental organisations. This prejudice is based on the principle that nobody should be allowed to “buy the licence to pollute”. Command and control regulations, where public environmental authorities have power to force industry to carry out the measures, have usually been preferred.

However, there is a perceptible change occurring in this position, as it becomes clearer that all environmental problems cannot be addressed at the same time, and that it is necessary to make choices. There is growing support for the (broad definition of the) Polluter Pays Principle, and resistance to trading systems is now primarily limited to the allocation of permits without charge. There is also a resistance within environmental organisations to allowing Norwegian companies to acquire permits abroad to cover their obligations in Norway. (This resistance is partly based on concerns about “hot air” in the climate change policy debate.)

In spite of proponents’ arguments about the control efficiency of trading systems, there remains a basic suspicion that they are devices for industry to escape from their responsibilities. There is a lack of confidence that monitoring and control will be efficient, and consequently, that emissions will be excessive.

An additional factor is general antagonism towards power-intensive electrochemical and electro-metallurgical industries. This antagonism has long history in Norway. Partly, it is a question of general distaste for industrialisation and economic growth; but more specifically, these industries are associated with heavy pollution and with extensive exploitation of hydro power that has historically implied big dams and dry water courses. Furthermore, in the past, the authorities have provided these industries with very favourable economic conditions, which are often considered to be subsidies. In spite of significant improvements in recent years, these industries therefore still struggle with a public image problem.

Business hesitation

Hesitation is the result of lack of familiarity

The business community, both through the Confederation of Norwegian Business and Industry and because of the involvement of some major companies, has been broadly involved in developing and promoting tradable permit systems. Therefore, business is not expected to present a significant obstacle to the introduction of a system of tradable permits, even if it is an unfamiliar and untested system. The general experience is that hesitation of business people is linked to lack of knowledge of a new instrument, and to uncertainty about the implications for their own company. When they learn more, and when basic uncertainties are resolved, they tend to become proponents.

From the outset, therefore, there is a preference within business for “command and control” systems. The experience here is that a reasonable dialogue has developed between companies and the Pollution Control Authority. Objections from the companies are listened to, and regulations and requirements are in general considered to be sensible. However, as emission standards have been tightened, there has been a growing awareness that this system is not cost-efficient, and that the economic burden of emission control may become unnecessarily high. This is part of the reason why the business interest in trading has been growing.

Another factor is the interest of government authorities in extending emission taxes, which the Norwegian companies sense will put them at a disadvantage *vis-à-vis* their competitors in other countries. The prospect of permits being allocated free of charge is therefore somewhat attractive. On the other hand, if a large share of the permits is to be auctioned, business sector support for trading would be weaker.

As business leaders learn more about trading systems, they realise that a system can be designed in many different ways – “the devil is in the detail”. So they involve their companies in the shaping of the systems.

A major attraction of a trading system of greenhouse gas emission permits is the potential link to international markets based on the flexible mechanisms of the Kyoto Protocol. Companies foresee that they can buy permits abroad to cover their obligations in Norway, and that the price of permits in Norway will equal the international price. If this link is not established, or if the possibility of acquiring permits abroad is restricted, this will create a severe obstacle to the functioning of the market for greenhouse gas emission permits market. For SO₂, this is not so important, since the estimated differences in marginal costs between countries are smaller.

Confidence is essential

For a trading system to function properly, participants must trust that the rules of the game will remain stable over long periods of time. Cost-efficiency is achieved when companies can choose between investing in abatement and selling/buying permits. If buyers do not trust that permits will be accepted in the future, the price of permits will not correspond to marginal abatement costs, and the system will not serve its function. In some segments of the business community, there is a basic distrust of the authorities. So there is, for instance, a suspicion that once they have invested in permits, the value of the permits will be reduced, or that taxes will be collected on top of the trading system. Even if such a suspicion does not lead to outright rejection of the trading system, another result may be a lack of participation in it.

The allocation of permits must be fair

The allocation of permits free of charge involves significant transfers of income and wealth, particularly for greenhouse gas emission permits. Since the number of permits to be allocated is fixed, the allocation between recipients is a multilateral “zero-sum” game. This problem may be difficult to resolve, since major players might prefer to block a solution, rather than see a competitor gain. The allocation of non-tradable emission permits in a “command and control” system also distributes value, but each allocation is a bilateral “non-zero sum” game that is easier to resolve. In this latter situation, if one company strikes a good deal with the authorities, the competitor will not object, but will use this knowledge as an argument to improve its own deal.

So far, it does not seem that the allocation of permits will create major obstacles in Norway. One reason is the emphasis in the proposals on a rule-based allocation procedure, and on securing broad support for the rules before they are enacted. Initial allocations are expected to be based on historical emissions, with some possibility to apply for modification according to clear criteria related to major investments in abatement and/or to special business cycle situations.

Market power in permit markets

Norway is a small country and the permits markets will inevitably be thin. This raises the possibility that exploitation of market power might distort markets. With greenhouse gases, this should not be a problem, as long as there is a free opportunity to trade internationally. However, in developing the proposal for an SO₂ emissions market, concern about market power was seen as a major obstacle. The oil companies were especially worried that a competitor could acquire so many permits that their own sales of fuels with a sulphur content would be reduced. Since the estimated price of SO₂ emission permits is low compared to fuel prices, acquisition of permits would be a cheap way of reducing competitors' access to fuel markets. To avoid such an abuse, a mechanism was devised whereby the petroleum companies would voluntarily pay a fee that could finance the acquisition of permits to cover the sales of oil products.

Another dimension of market power was that some potential buyers of permits outside the oil and gas industry were afraid of dominance by the oil companies. In particular, companies engaged in offshore activities were expected to have ample profit margins and a growing demand for permits. The other companies were therefore afraid of being squeezed out of the market, or having to pay so much for permits that their overall profitability would be undermined.

In small markets such as Norway, care is needed to develop a well-functioning market. Confidence has to be built (*e.g.* by declarations across party lines that, once set, the rules of the scheme, including the number of permits to be validated each year, will not be tampered with). The scheme itself needs to be established by law, implying an element of stability and predictability as compared with a tax regime, where rates can be changed at relatively short notice. Some permits have to be issued to create a future market so that emitters can calculate the profitability of investing in permits versus investing in abatement. On the other hand, some permits also have to be kept for sale in the year of validation, so as to make sure that some permits will always be available on the market. To the extent that *international* trading becomes a reality, the importance of these considerations diminish in the design of *domestic* trading systems.

Chapter VII

OBSTACLES TO IMPLEMENTING TRADABLE POLLUTION PERMITS: THE CASE OF POLAND

by

Tomasz Zylicz

Warsaw Ecological Economics Center, Warsaw, Poland

Background

1989 marked the launch of a comprehensive reform of Poland's environmental policy. From the outset of work on this reform, the Economics Department (ED) in the Ministry of Environment (ME) has strongly recommended taking advantage of emissions trading to improve the cost-effectiveness of the new policies. The adequacy of simple forms of emissions trading as an option for former centrally planned economies was widely discussed at political as well as academic fora (Zylicz, 1990). The idea of Tradable Pollution Permits (TPP) was officially introduced into the *National Ecological Policy* (Ministry, 1990a, b), and approved by the Parliament in 1991. Subsequently, TPPs were included in the new Environmental Protection Act, drafted in 1991.

The pace of the reform process has therefore been quite rapid. Prospects for implementing TPPs seemed bright. The 1991 bill, however, has never been passed, because of other political priorities which overshadowed environmental concerns after 1991. For many years, environmental reforms were slowed down.

The preference for TPPs, declared officially in 1989-1991, was not an obvious choice. There are not many environmental policy instruments which are more controversial than tradable pollution permits. Praised by many, emissions trading is seen by others as an ethically-ambiguous solution that, at best, can be contemplated not sooner than the 21st century. This was the most frequent message heard by Polish Government officials from their Western European counterparts.

Like conceptual frameworks for general economic and political reform, ideas for an environmental policy reform were also developed by dissident academics prior to 1989. TPPs were early recognised as a promising instrument, not only because of their economic characteristics, but also because of political advantages of letting the market constrain the power of bureaucrats.

A small (yet gradually growing) group of environmental economists studied the merits of TPPs and their applicability in a non-market or semi-market economy. A radical departure from the Soviet-type centrally-planned systems was not anticipated soon, but it looked possible that TPPs could be usefully applied in Central and Eastern Europe (CEE). Some key issues known from the literature were analysed to identify circumstances under which TPPs might provide the benefits of which they are theoretically capable. It was argued that the instrument could be applied – subject to some administrative oversight – even under circumstances that are very far from a mature competitive economy. Detailed arguments were discussed in the local academic literature (Zylicz, 1987), explained in non-technical terms in the most popular Polish environmental monthly magazine (Malawski, 1987), and also repeated in an environmental economics textbook (Zylicz, 1989). In the fall of 1989, TPPs were endorsed by a Polish-American workshop aimed at providing input to policies of the first post-communist government, and attended by a

number of Polish environmentalists (Bochniarz & Kassenberg, 1989). The main argument was that, even in thin or hardly-existing markets, substantial cost savings could be achieved, if at least one low-cost polluter and a neighbouring high-cost polluter were allowed to comply with certain regulations jointly.

First of all, the TPP concept had to confront the command-and-control approach. In the late 1980s, this was relatively easy, as the central planning ideology had been severely eroded in Poland.

Nevertheless, striking a balance between “command-and-control” based sectoral and/or regional approaches and financial (macroeconomic) approaches was a major challenge facing environmental and industrial policies. On the one hand, any discretionarism would have to be avoided, in order to provoke the necessary psychological breakthrough in the attitudes of industrial managers in the former centrally-planned economies, who were used to the excessive paternalism of state administration. Since the very beginning of the reforms initiated in 1989, these managers have been seeking individual or sectoral subsidies, exemptions, tariff protection, etc. To yield to such pressures would have led to the instant decay of the emerging market system. On the other hand, the over-reliance on financial measures – in the absence of well-established market structures and entrepreneurial behaviour – led, in Poland, to an economic collapse much deeper than initially expected (and, perhaps, deeper than was absolutely necessary).

Environmental policy-makers must therefore seek to reconcile the need for sectorally- and regionally-customised policies (typically implemented by “command-and-control” approaches) with the need for financial rigor. The answer cannot simply point to the positive OECD experiences, since OECD countries could combine discretionary measures with the functioning of well-established markets. Economies in transition do not operate under the same circumstances.

It is precisely here that TPPs provide an option to gradually integrate market choices into sectoral and regional physical plans. They can be applied both at regional and sectoral scales. Successful examples of regional implementation of these instruments are provided by numerous “bubbles”, often the only practical and cost-effective way to meet ambient standards in many non-attainment areas in the US. The textbook example of a successful sectoral programme is the phasing-out of lead additives in US refineries. Finally, combined sectoral/regional effects were observed in cases where industry-specific pollutants (*e.g.* volatile organic compounds released from laundries) were tackled on a municipal basis (Howe, 1994; HIID, 1996).

Perhaps an even more difficult challenge was to confront TPPs with the system of pollution charges that were already in operation in Poland (since the 1980s).

The rationale for emission charges – or, more specifically, for *Pigouvian taxes* – stems from the fact that in a perfect-information, perfectly-competitive, world the same socially-desirable effect can be achieved either through setting emission quotas, or through a per unit charge. If the marginal costs and benefits of abatement are known, then the point where they are equal defines the “socially right” level of emissions. This level minimises the sum of total costs of environmental damages (or foregone benefits), plus the total costs of protective activities. Environmental policy-makers can reach this level by simply not allowing emissions to be higher than optimal. Under standard microeconomic assumptions, the same effect can be achieved through establishing a charge (corresponding to the level of the marginal abatement costs and marginal benefits) per unit of emission released into the environment. The equivalence of both solutions is only a formal one. In the real world, however, the superiority of one system over the other is contingent upon specific assumptions (Weitzman, 1974). For practical purposes (Adar and Griffin, 1976), there is often a strong *a priori* preference for the quantity-based approach. Unfortunately, neither the seminal paper of Weitzman, nor the subsequent discussions have ever been referred to in the East European and Soviet literature dealing with emission charges.

There is a way to combine environmentally-safe, quantity-based, regulations with a flexible market response. This involves the use of TPPs, and it perfectly conforms with Herman Daly's (1984) principle of separating scale and allocation decisions in environmental policy. To regulate the environment implies deciding on the extent to which a given resource (*e.g.* the carrying capacity of the atmosphere or an aquifer) should be used, and what portion of the resource should be allocated to any of its potential users. Only the second problem is suitable for market regulation. When it comes to the question of scale, the market may easily fail as a guide, and the solution should be arrived at by means of a public debate,

followed ultimately by the imposition of an administrative standard. This theoretical background (*i.e.* letting the market determine the allocation, but not the scale), coupled with its cost-reducing potential, make TPPs a preferred instrument of environmental policy.

It should be stressed that the quantity-based approach was not seen just as being a more understandable and acceptable option for those who have normally been exposed to non-market mechanisms. In particular, it was not that the preference for marketable permits might result from CEE managers' familiarity with physical rather than financial targets, as interpreted by some analysts (Levin, 1989). Neither was it just a preferred tool in an economy with "soft" budget constraints. It was advocated for as an instrument which combines the certainty and effectiveness of a standard with the flexibility of a market incentive. While it is true that its viability was first demonstrated in a developed market economy, it may – unlike charges and fines – also work well in a poorly-developed institutional context. Of course, here, its cost-saving potential cannot be fully utilised.

The Polish environmental policy reform

The general context for implementing TPPs in Poland in the early 1990s has been outlined above. The main challenge thus seemed not so much to depart from command-and-control approaches, but rather to persuade policy-makers and stakeholders that, under the particular circumstances in Poland at the time, TPPs were a preferred alternative to the existing pollution charges.

There was an impression in the early 1990s that Poland, as well as other economies in transition, could very quickly implement advanced market solutions in environmental policy. Post-communist Europe was thought to have an opportunity to learn from the Western experience and to avoid some of its mistakes. One study of economic instruments concluded with a statement whose optimism was, however, somewhat premature:

"Market-based environmental policies are an ideal tool for nations nurturing new market-based economies. Employing market-based approaches to environmental policy from the outset can avoid some of the costly mistakes made in the US, and propel Eastern Europe into the forefront of environmental protection" (Dudek, Stewart, and Wiener, 1992).

The assumption about "the outset" was not an obvious one. Although it is true that environmental reforms in CEE ought to be comprehensive, this did not mean that their architects might be able to "start from scratch". The countries of the region had had extensive legislative systems which could not be ignored without putting environmental objectives at risk (at least in the short-run). There were too few analyses in the early 1990s stressing the importance of legal and political constraints on what seemed to be "appropriate" policy choices (Godard, 1994).

Here is how the main government policy document described the role for economic instruments at the time (Ministry, 1990b).

"Economic instruments perform a number of useful functions. The most important task of these instruments is to minimise overall costs of environmental protection through an efficient differentiation of control requirements. Namely, those agents with the lowest abatement costs should be given the most stringent requirements. In practice, only marketable permits can perform this role in an environmentally-safe way. Theoretically, the same goal can be achieved by setting effluent charges at sufficiently high levels; however, in practicality nowhere is this solution relied upon due to political and methodological obstacles. Another function performed by economic instruments is to raise funds to be spent on environmental management needs. Charge systems throughout the world serve this purpose – and their implementation in Poland is expected to do the same. Charges, as well as other economic instruments, may also create incentives for environmental conservation and improvement, but this is their secondary role."

Thus, the reforms envisaged a substantial role both for TPPs and pollution charges, the latter serving mainly revenue-raising purposes. While the document proved successful at arguing for substantial pollution charge increases (since 1991, the charges have corresponded to 0.5 per cent of GDP and have provided more than a reasonable public share in environmental project financing) (Zylicz, 1998), it failed to speed up the introduction of TPPs.

Another important notion introduced in the policy document was that of responsibility for attaining ambient environmental standards. Under the current Polish law, no entities are actually responsible for attaining these standards.

“First of all it is necessary to clearly state that national ambient standards (*e.g.* concentrations of toxic substances in the air, water quality classification, soil contamination limits) are binding on government bodies at the regional (voivodship) or other levels legally responsible for environmental regulation. Achievement of standards – within the framework of timetables set in conjunction with government agencies (National Planning Commission, Ministry of the Environment, voivodship administrators) and formalised in Parliamentary resolutions or acts – is the responsibility of the government body specifically in charge of environmental permitting. Individual firms in turn are responsible for preparing timetables laying out their abatement programmes and submitting them to the appropriate authorities. These sub-timetables are an important means for meeting regional ambient standards.”

The concept of responsibility for attaining ambient standards is not universally-shared by Polish lawyers, and even less so by administrators. The latter seem to be quite comfortable in the current setting where “everybody should do his or her best to meet the relevant ambient standards”. As the responsibility is diffuse (shared by polluters, regulators, project designers, citizens, etc.), no sanction can be taken against anybody if such standards are not attained. At the same time, this legal vacuum is an important reason why administrators express little interest in developing instruments to meet ambient standards at least cost (they are not personally responsible for meeting these standards at all).

Linked to this is the issue of decentralisation in environmental decision-making. The 1990 documents envisaged substantial freedom in implementing policy objectives, provided that the fundamental principle of responsibility for attaining ambient standards was introduced.

“Within the framework of (certain specified) constraints the (regional) agency empowered to issue emissions permits has full flexibility to plan and implement pollution control policies. In particular, the agency, based on its informational, technical, and staff capabilities, may choose one or more of the following four methods. First, it can demonstrate attainment of national ambient standards by way of compliance by individual sources with their respective emissions standards as established by national authorities. (Currently there exist emission standards for a limited number of source types.) Second, it can by administrative action implement stricter emissions or discharge limits for individual sources which are not exchangeable for charges or fines. Third, it can regulate use of environment by means of a system of charges and fines similar to the one in use currently. Fourth, it can institute permit markets in which individual users trade all or part of their rights to emit or discharge to other users of the same water- or air-basin, subject to the approval of the pertinent permitting authority.”

Seen from the perspective of the late 1990s, policy challenges and dilemmas identified in Government documents of the 1990s are still valid, and crucial for the fate of TPPs. The reluctance to set ambient quality targets persists, and cost-effectiveness is violated in many programmes. Policy objectives set forth in 1990 have never been fully-accepted by many professionals, including lawyers, who prefer to minimise departures from the conventional command-and-control system, supplemented with modest pollution charges. As a result, whenever they appear in draft legal acts, TPPs are introduced as an option without changing the main source-based (and technology-based) approach.

The government officials promoting TPPs in 1989-1991 did appreciate the need for “packaging” this idea, in order to minimise resistance and to maximise political support. The national policy documents were accompanied by educational publications emerging from the Ministry of Environment. In addition to a series of media interviews and popular articles, two major reports were translated into Polish and widely distributed, in order to enhance understanding of environmental policies in modern market economies. The publication *Economic Instruments for Environmental Protection* (OECD, 1989) gave a wide overview of policy experience in the OECD countries, while the *Project 88* (1989) report shed light on “harnessing market forces” in the US. Thus, foundations for new policy initiatives were laid down. What was called for was empirical proof that savings and entrepreneurship can be unleashed if environmental policy is implemented with economic instruments, rather than with inflexible regulations.

Since 1990, the Economics Department has therefore tried to identify a good demonstration site for an emissions trading experiment. Even though the Ministry of Environment was committed to such an experiment, the task was not an easy one, since under the Polish law, it is not the Minister, but regional environmental administrators, who issue pollution permits. If these administrators had been responsible for attaining certain ambient standards, they would have had a strong incentive to seek instruments which reduce pollution in a least-cost way. However, this is not the case, so they feel much more comfortable focusing on technology-based requirements. As a rule, they would also prefer to continue under non-attainment conditions, rather than to take the risk of engaging in an emissions trading project whose merits were perceived to be controversial, and where the chances of political failure were high, in the absence of a satisfactory legal framework.

The plans of Economics Department met with interest from a number of environmental policy specialists, mainly from the US. In 1990, the Economics Department reached an agreement with Environmental Defense Fund (EDF) to jointly develop a pilot project demonstrating the usefulness and viability of TPPs. Officials from both groups started to study possible locations for such a project. Soon, it became obvious that the physical and technological characteristics of a site were of lesser importance than the understanding and the willingness to co-operate of a regional environmental administrator. As a result, several potential locations were eliminated.

The Chorzów project

In 1991 (and at the request of the Economics Department), the Minister of Environment approved the outline of the pilot project in Chorzów. This Ministerial approval was an imperfect substitute for a clear legal provision for emissions trading. In this case, it was the environmental administrator of the Katowice region who made the decision to implement the experiment. Once the location was determined, Economics Ministry officials made a number of trips to Chorzów to: i) identify potential participants in the experiment; ii) assess benefits from emissions trading; iii) present and discuss the results and recommendations at various public fora in Chorzów; and iv) help the local-decision makers to co-operate with each other, even in the absence of a fully satisfactory legal framework. All of these tasks were accomplished (Dudek, Kulczynski and Zylicz, 1992), which had the effect of turning the experiment into an interesting demonstration project.

It should be noted, however, that no “emissions trades” have actually taken place in Chorzów, since Polish law does not recognise “emission reduction credits”. In 1991, a new Environmental Protection Act was drafted by the Government. It included Article 45, which stated that “the terms of a pollution permit can be transferred (either fully or in part) to another plant subject to the approval of the authority who issued the original permit”. The Act was eventually abandoned, so no “emission reduction credits” could formally become a commodity. Consequently, the whole experiment can be labelled as a “tradable permit project” only in a metaphorical sense.

In an extensive appraisal of the Chorzów project, Beblo (1994) stated that its main outcome was to let its participants (*i.e.* local polluters and administrators) learn how to negotiate in the common interest, and to recognise values hidden in various assets. For the first time in implementing environmental policy in Poland, a conscious effort had been made to optimise the allocation of permits, subject to the constraint defined by the desired degree of improvement of ambient quality. In contrast, typical implementation debates elsewhere have focused on technical considerations and (despite the rhetoric) treated ambient quality as a residual outcome.

The City of Chorzów used to be one of the worst polluted municipalities in Poland. Two plants located in the city centre were responsible for a major part of this pollution. These are: the Kosciuszko Steel Mill (hereafter, the “steel mill”) and the Power Plant “Chorzów” (hereafter the “power plant”). Table 1 characterises their atmospheric emissions in 1990.

While both plants contributed similar amounts of particulates and sulphur dioxide, the steel mill was a major polluter with respect to carbon monoxide and hydrocarbons. The steel mill had a government-approved restructuring programme which envisaged significant emission reductions especially

Table 1. Emissions of Selected Pollutants in 1990 (Tons)

	Steel mill	Power plant
articulate matter	4 280	5 067
arbon monoxide	16 951	200
ulphur dioxide	3 020	3 071
itrogen dioxide	1 716	561
liphatic hydrocarbons	492	71
romatic hydrocarbons	282	8

ource: Katowice Regional Environmental Authority files.

after 1995. The cost of achieving pollution reduction through the programme was rather low, since most of the planned investment expenditures were to be borne for production purposes. On the other hand, the power plant – one of the oldest electricity stations in Europe operating now on an obsolete equipment from the 1950s – was expected to be shut down in about 5 years, for technological reasons. Clearly, it would be inefficient to equip it with any modern end-of-pipe installation to reduce the emissions of sulphur dioxide by a significant amount.

Of the two plants, the steel mill was a typical low-cost one with respect to abatement, and the power plant was a typical high-cost one. At the same time, the former was in some financial distress, while the latter was relatively better-off. It thus seemed obvious that a successful solution to the Chorzów problem would require that:

- the steel mill abate more than the power plant; and
- the power plant participate in the steel mill's abatement costs.

However, this result was more easily hypothesised than achieved, since Polish law did not recognise the concept of a tradable pollution permit.

Following advice from the Economics Department, the regional administrator used the earliest opportunity to issue new permits for both polluters (whose operations contributed to the violation of ambient standards), in order to induce a cost-effective allocation of abatement effort. His choice was constrained not only by the absence of the legal notion of an “emission reduction credit”, but also by the fact that background concentrations of some pollutants would not let him issue any non-zero emission permit. Nevertheless, it did prove possible to reach a consensus between the administrator and the polluters, as well as among the polluters themselves, to:

- issue permits approximating the cost-effective allocation of abatement effort between the steel mill and the power plant;
- make the power plant pay for a more liberal permit than would otherwise be applicable; and
- assist the steel mill in radically speeding up its restructuring programme.

All payments related to the project involved transfers to and from the regional environmental fund. (The environmental administrator fully controls expenditures from the fund, and – to some extent – can manipulate its revenues.)

The results of the project can be best judged from Tables 2 and 3. In 1993, emissions from the steel mill were reduced even further, and close-to-zero emissions were recorded in 1994 [100 kg (!) of particulate matter, 3.4 t of carbon monoxide, 3.6 t of sulphur dioxide, and 51 t of nitrogen dioxide]. Emissions from the power plant increased somewhat, but the net ambient effect was unambiguously and remarkably beneficial (Beblo, 1993). Besides, part of the apparent increase of pollution from the power plant was probably “paper growth”, which resulted from improved monitoring. However, the growth of nitrogen oxide emissions was caused – quite paradoxically – by switching to cleaner fuel of higher calorific value (resulting in higher combustion temperature) (Beblo, 1994).

The “greening” of the steel mill was caused mainly by the shut-down of its open hearth furnace, coke plant, and all other raw material units. All of that would have happened sooner or later anyway, but the time factor was critical in this heavily-contaminated neighbourhood. It is estimated that the project

Table 2. Emissions of Selected Pollutants in 1992

	Steel mill		Power plant	
	Tons	1992/1990	Tons	1992/1990
articulate matter	517	12%	4 028	80%
arbon monoxide	47	0.3%	1 141	570%
ulphur dioxide	972	32%	3 226	105%
itrogen dioxide	769	45%	1 782	318%
liphatic hydrocarbons	0	0%	71	100%
romatic hydrocarbons	0	0%	9	112%

ource: Katowice Regional Environmental Authority files.

Table 3. Emissions of Selected Pollutants from Both Plants in 1990-1992

	1990 (tons)	1992 (tons)	1992/1990 (%)
articulate matter	9 347	4 545	49
arbon monoxide	17 151	1 188	7
ulphur dioxide	6 091	4 198	69
itrogen dioxide	2 277	2 551	112
liphatic hydrocarbons	563	71	13
romatic hydrocarbons	290	9	3

ource: Tables 1 and 2.

accelerated the restructuring process of the steel mill by 2.5 years. The compromise this required was to tolerate somewhat higher emissions from the power plant. However, regional authorities have had very limited practical control over these emissions, since this is a co-generation plant providing heat not only for Chorzów, but also for several other districts. The most likely conventional scenario was therefore to tolerate emissions from the power plant (which initiated some abatement activities) for the rest of its life, and to wait to see how the steel mill gets on with its restructuring programme.

Another part of the technical solution was the shutting down of virtually all combustion plants within the steel mill, whose demand for electricity and heat was then met by the power plant. Despite its technological obsolescence, the power plant could provide end-use energy at lower environmental cost than the steel mill could. Its emissions did grow as a result of the increased output. Nevertheless, in net terms, this growth was more than compensated by lower emissions from the steel mill.

On the legal side, the project was made possible by partially manipulating “background concentration” levels, in order to link emissions to ambient concentrations. Nevertheless, all permit decisions – which included a number of other smaller plants not discussed in this paper – were non-conventional, and could be considered controversial. In order to avoid confusion and to minimise the risk of having permit decisions challenged by environmental groups, a major education effort was undertaken (Zylicz, 1994b). At the national level, the Ministry of Environment published and widely distributed Polish translations of the 1989 OECD report, as well as using *Project 88* to explain the rationale for using economic instruments in environmental protection. At the local level, efforts were made to let environmental NGOs and the media understand the project, and to appreciate its benefits. These steps proved to be effective, since no permit decision has been challenged. Instead, many other municipalities approached the regional administrator in Katowice requesting that similar arrangements be applied in their air-sheds. However, because of the lack of a satisfactory legal framework and because of the amount of administrative work (to substitute for the legal framework), the authorities were reluctant to replicate the experiment elsewhere.

Three conclusions can be drawn from the Chorzów project:

- the value of the TPP approach to address complex restructuring and abatement problems in non-attainment areas was demonstrated;
- a remarkable decrease of emissions was achieved mainly by “transferring” permits between just two major local polluters; and
- industry and administrators revealed a highly co-operative behaviour when assisted by competent external experts.

These observations confirm the viability of simple TPP schemes in an economy in transition, in a heavily polluted region, and without any significant external financial aid.

The Opole project

In late 1991, there was a change of government, which resulted in slowing down the pace of policy reforms. For a number of years thereafter, the development of emissions trading was adversely affected by the political process. The draft Environmental Protection Act of 1991 was abandoned in 1992. In 1993, an Air Protection Act was drafted as an independent initiative undertaken by the electric power sector. The expectation was that with unclear perspectives for a new comprehensive environmental protection law, several sectoral or “one-medium” regulations had a better chance of being enacted.

Power plant operators wished to provide for permit tradability, and therefore introduced appropriate clauses (as Articles 25 and 26). The electricity sector had thus become the first important industrial interest group to appreciate the benefits of emissions trading. The prime motive for the attention given to this instrument was the prospect of having to meet stricter emission limits. Many power plants realised that it would be very costly to meet those limits individually. At the same time they could demonstrate lower cost options (leading to the same overall emission reduction) available to some other power plants. Hence, it would be beneficial for the entire sector to co-operate and to meet the limits jointly, rather than individually.

The electricity sector tried to lobby for its air protection bill for a couple of months. However, because of the lack of political will, this Act has never been debated in Parliament.

A new opportunity arose with a PHARE Programme offer to finance a few full-scale research projects addressing problems of practical importance for the Ministry of Environment. A study was commissioned to elaborate the practical steps required to implement tradable permits in a region on a routine basis. In 1994, the Atmoterm, a Polish firm which formed a consortium with several other consultants, was granted a contract under the PHARE Programme to prepare an actual implementation in the Opole region in southern Poland [see Atmoterm (1996) for a description of key elements of this project]. Tradable permits also emerged as an important instrument in another PHARE project commissioned by the Ministry of Environment a few months later. This project, whose focus was on methodologies for assessing the costs of meeting international environmental obligations, concluded that these costs largely depend on the policy instruments which are applied (Peszko, 1996). At the same time, it demonstrated the cost-effectiveness of applying tradable permits in order to let domestic emission sources comply with requirements.

The Opole project was originally conceived as an implementation exercise. Nevertheless, it became apparent that with the political establishment being only marginally interested in policy reforms, there would be no chance for an appropriate amendment of the legislation. Consequently, the project was redesigned and, instead of actual implementation, a series of computer simulations served as a test of the legal mechanism that needed to be developed. The main features of the Opole project are:

- polluters are subject to non-tradable source-oriented emission standards, and tradable permits;
- the tradable permits are basically grandfathered, even though there is also an auction to arrive at an initial allocation;
- the total amount of permits is derived from a regional attainment plan, and it shrinks over time;
- trading is based on exchange rates determined for six ambient impact zones;

- banking is allowed, but emission reduction credits deposited are discounted with 20 per cent annual rate.

With the Opole project completed (on paper) in 1996, there seemed to be no further need to demonstrate the usefulness of the instrument in the Polish context. The political decision of 1990 had never been formally reversed, and combined with the Chorzów project, should have convinced the sceptics. The concept is also well-understood by professionals and the media alike. Important design issues were analysed in both practical and theoretical terms (Stavins and Zylicz, 1995). Nevertheless, attempts to implement it proved that there were yet many additional specific problems that need to be addressed.

The Jadwisin workshop

One issue that has recurred on many occasions in policy debates in Poland is the relation between a tradable permit to pollute and source-specific requirements. The requirements may take the form(s) of:

- an emission standard for a given type of installation (*e.g.* under the Ordinance of the Polish Minister of Environment of 1990);
- an emission limit for a given source derived from some ambient quality considerations (a typical solution in Poland); or
- mandating use of a specific technology (*e.g.* BAT, BATNEEC, or RACT; *i.e.* Best Available Technology, Best Available Technology Not Entailing Excessive Costs, or Reasonably Available Control Technology, respectively).

The main question is whether purchasing an additional permit can free a source from the obligation to comply with any of such requirements. If it cannot, then what incentive is there to purchase the permits in the first place?

While some systems occasionally exist which do not subject polluters to any environmental requirements other than those indicated in a tradable permit itself, as a rule, the permit is only a part of an environmental regulation “package” which affects the pollution source. In other words, the permit (whether tradable or not) held by an enterprise may be a *necessary* condition to operate, but not a *sufficient* one. For all practical purposes, future tradable permit systems in Poland should be envisaged as co-existing with other regulatory regimes.

Additional concerns result from the recent adoption of the IPPC (Integrated Pollution Prevention and Control) Directive by the European Union. This Directive mandates using BAT as a reference for abatement requirements. Does BAT allow for transferring a permit from one source to another? This question is raised by Polish lawyers. BAT is a technology which can actually be applied, and which minimises emissions per unit of performance in a given sector. If a firm can bring its emissions down to a certain level then, by definition, BAT cannot be less stringent than what the firm has just achieved. Therefore, whenever BAT is required, the least-polluting firm sets a standard for everyone else, and eliminates the notions of both “surplus” emission reduction, and emissions trading.

Even though it is theoretically possible, this extreme understanding of BAT can never become a political option. It would eliminate incentives to innovate, and would doom economies to stagnate. Instead, BAT is understood pragmatically as a level of emission per output that eliminates “excessive” pollution at the source. It is (and will continue to be) revised not immediately after a technical improvement has occurred, but rather periodically at reasonable time intervals, once the new technology builds a successful operating record.

To elucidate the relationship between tradability of permits and emerging European legislative issues, the Polish Ministry of Environment, together with Harvard Institute for International Development (HIID) organised a policy workshop in Jadwisin (near Warsaw) in June 1996. The Workshop provided an opportunity to assess Polish plans to introduce emissions trading from the point of view of both US practical experience, and the European legal framework.

The presentation made by the Environment Directorate of the European Commission (see Klassen, 1996) made it clear that the IPPC Directive imposes some extra requirements on pollution sources, but it does not preclude emissions trading. Nevertheless, the latter cannot over-rule any source- and/or

site-specific conditions that policy-makers in Europe are likely to retain. The British regulatory framework for the power sector was cited as an example. Powergen, one of the two enterprises producing electricity in England, is subject to two types of requirements: "Type A" – derived from BAT; and "Type B" – derived from non-technological considerations (such as the Second Sulphur Protocol). Under the former, all Powergen plants are allowed to emit 918 000 tons *per annum*, and quotas for specific plants are not transferable. However, these quotas were judged to be insufficient for meeting the Second Sulphur Protocol target. To comply with that target, the emissions from Powergen should be brought down to 450 000 tons per annum. Thus, on average, each plant should emit roughly 50 per cent less than their respective present BAT levels. This is an ambitious goal, but not an impossible one. To make it easier, plants can trade their initial "Type B" allowances as long as they comply with their respective "Type A" emission limits. Such a framework is consistent with the IPPC Directive.

As explained to Workshop participants (see McLean, 1996), the US Acid Rain trading programme has a similar structure. Polluters are subject to double requirements. Each source has to comply with a non-transferable emission limit, as set in its individual pollution permit. On top of that, each participating source was allocated an emission allowance – more restrictive than the permit – which can be freely-traded. It should be noted that an emission permit in this framework is not identical with an emission allowance. A source may buy a lot of (tradable) allowances, but may not be allowed to actually emit, unless the emission complies with the terms of its (non-tradable) permit. To formally stress this difference, the legal system uses two distinct concepts – a permit and an allowance. The architects of the programme have avoided using the earlier concept of an "emission reduction credit" – implicitly referring to a permit, which served as the baseline for reductions – in order to emphasise the two requirements: one non-transferable; the other transferable.

In the Opole simulation project, sources were subjected to dual constraints too. First, a source may be required to comply with some technological and/or ambient requirements. Second, it is expected to comply with annual (or bi-annual) reduction targets mandated by the provincial governor and resulting from general policy considerations. Additionally, as in the US Acid Rain Programme, everybody can buy an emission allowance, but not everybody can emit the corresponding amount of pollution.

From the point of view of the polluter, the distinction between a permit and an allowance may seem like nothing but an added complexity. However, there are also advantages of such a scheme. First of all, it frees environmental authorities from having to assess the overall impacts of individual transactions, since there are other constraints that prevent excessive pollution at any site. Additionally, since not only prospective polluters can buy emission allowances, governments, environmental groups, and individual citizens may enter the market and thus – by retiring allowances they have just purchased – express their preferences with respect to the amount of pollution. Such activities vitalise the demand side of the market, help to sustain price levels, and ultimately reward those who contribute to the supply (*i.e.* those who abate more than required).

Clarification of the constraints that should (or could) accompany emission trading and its compatibility with European laws might have been the most important outcome of the Jadwisin Workshop. Nevertheless, the list of topics covered in discussions was much longer. In particular, it became clear that the US experience with tradable permits is extremely diversified. The two systems presented at the Workshop – Acid Rain and RECLAIM (see Leyden, 1996) – exemplify two different scales, and two different legal regimes.

The Acid Rain programme creates a country-wide market for allowances. No matter how far from each other emission sources are, they can trade these allowances without constraint. There is also an option to bank unused allowances for future use (without discounting). Authorising unconstrained intracontinental and intertemporal trades was justified by the goal of the programme (*i.e.* to decrease acid deposition over the period 1990-2010). Since sulphuric emissions migrate over large distances, and because their depositions cumulate in lakes and soils, neither the exact location of the emissions, nor their timing, was deemed crucial. However, this does not mean that the programme risks creating pollution "hot spots" caused by an excessive accumulation of allowances at one particular site. The reason for this is that "allowances" are not the same thing as "permits". Anybody can buy *allowances*, but not everybody can actu-

ally use them, since sources also have to comply with the terms of their *permits*, which are both site-specific and non-transferable (see also Chapter II).

The most important feature of RECLAIM is its local scope. The programme is supposed to improve ambient air quality in the South Coast area in California, so its geographical coverage is quite limited. Because of the severity of the pollution problem in Los Angeles area, the size of the market is further reduced by a division into two impact zones, according to environmental sensitivity and population exposure. These factors, coupled with a large scale of emission reductions required, imply abatement costs 5-7 times higher than in other programmes. Another difference of the RECLAIM programme is that (given its local focus) the same permit combines short-term local with long-term regional requirements. This rather unusual design was justified by two factors. First, Los Angeles polluters are subject to extremely severe abatement requirements, so that none of the sources is likely to earn (and sell) a large volume of emission reduction credits. Second, due to the high level of concentration of pollutants in the air, the exact location of a pollution source is of secondary importance.

Polluters from the Los Angeles basin simultaneously participate in the Acid Rain and RECLAIM Programmes. In particular, they can buy Acid Rain allowances, but it is rather unlikely that they will ever use them to increase emissions above their initial allocations. This example illustrates three principles. First, firms may participate in more than one permit market operating independently. Second, purchasing a permit or an allowance is not identical with acquiring the right to pollute. Third, for the same emission reduction, a firm may get more than one credit. The latter principle corresponds to the fact that reducing emissions yields multiple benefits (*e.g.* local and continental), and those who abate should face incentives reflecting all of these benefits.

One common feature that RECLAIM shares with the Acid Rain programme is a rigorous reduction schedule. In both cases, overall emission reduction targets are translated into annual reduction steps that polluters must comply with (not individually, but as a group). This is in sharp contrast with conventional approaches, which give industries or municipalities a reasonable amount of time to achieve the target, and without interim check-points. The usual outcome observed in such regimes is that a significant number of sources ignore the regulation, and reach the deadline without achieving the emission target. The more widespread is this pattern, the less enforcement action is likely to be effective. As a rule, the government amends the regulation and postpones the deadline. Both in Poland and in the US, one can find examples of this type of policy failure. Therefore, it is worth noting that the Opole project replicates the good practice of translating an overall long-term goal into annual steps that are much easier to enforce.

Workshop participants had an opportunity to discuss various technical details of permit market design. All US speakers emphasised one key rule: "keep it simple!" Vitality of the market, political support, and effective co-operation with businesses depend on stability, transparency, and simplicity of the trading mechanism (Sleszynski, 1996). To illustrate how bureaucracy can be kept to a minimum, participants were shown the Acid Rain programme permit application. It consists of 3 pages with just several boxes to be filled out. US experts stressed that one should often sacrifice theoretical accuracy for the sake of simplicity, in order to make the system work more smoothly on behalf of the environment. An example of such a compromise is reducing the number of impact zones or exchange indices. Looking at the problem from this perspective, some participants considered the Opole project (with its 6 impact categories) to be too complicated.

Representatives from the Air Protection Department in the Polish Ministry of Environment raised a number of questions regarding daily operations of permit markets. One important aspect is confidentiality of emission and transaction data. Under US law, some trading details cannot be disclosed, but the total emission of a given pollutant discharged from the plant is publicly-known. Another problem is how to include small sources in the trading system. In general, because of transaction and administrative costs, there are limits to extending the market to include small pollution sources. Policy-makers and authorities should investigate whether alternative instruments, such as input and pollution taxes, performance standards, fuel specifications, or subsidies for upgrading equipment can better achieve the overall goal for these sources of reduced emissions at a lower cost.

Enforcement is also an important aspect of emissions trading. Under the US Clean Air Act, participants in the Acid Rain programme are exposed to three types of sanctions. Failure to comply with the sulphur allowance requirement results in a fine of \$2 000 per ton. Violating the terms of a site-specific permit also triggers a fine of \$25 000. At the same time, any excessive emission is automatically deducted from future allowances of the firm. All of these sanctions are mandatory, and industries do not challenge them.

Another practical issue raised at the Workshop was how to win the support of the Parliament for a law that – like the US Clean Air Act or the (prospective) Polish Environmental Protection Act – must be full of technical concepts and details that are incomprehensible for laymen. While it is true that the law may not be understandable for an average Member of Parliament, environmentalists need to cooperate with politicians who are likely to be “champions” of an innovative regulation. Winning the support of the Polish Parliament for tradable pollution permits will be a process that will require considerable communication and education efforts.

Not a turning point yet

A new Environmental Protection Act (commissioned by the Ministry of Environment) was drafted in 1996. As with previous reforms, this draft included a provision for the tradability of permits (this time in Article 88). As previously, the new provision gave an opportunity for regional administrators to authorise permit transfers, subject to some constraints. The new element was that the article referred to an ordinance that was to set specific rules for emissions trading. Such an ordinance was, in fact, prepared under the Opole project, and it could have also been incorporated into the main Act. However, the drafters retained their original scepticism, and apparently preferred to minimise the presence of an instrument they considered to be controversial.

At the same time, the draft 1996 Act devoted an entire chapter to pollution offsets. Economists may be surprised with the attention given to this special case of emissions trading, but it can be explained by the fact that offsets were also the earliest form of the trading instrument used in the US. They were also included in the German air protection legislation in the 1970s, where they proved to be of little practical importance, as only a dozen of trades took place under the so-called “modernisation clause”. The Polish Chapter on pollution offsets illustrated the fact that it takes time before certain experts get accustomed to new ideas. One may ask how many years it will take before a general scheme of tradable permits can be accepted as a routine mechanism.

When the 1996 draft was abandoned in 1997 (mainly due to the political dynamics of an election year), it became likely that introducing tradable permits would again be put aside for many months. There is not much else that can be done within the environmental economics profession to promote the case. Emissions trading has been a permanent item on the policy reform agenda for several years. The most important obstacles come from elsewhere. Amending environmental laws competes in government, Parliament, and other *fora*, with other issues seen by politicians as higher priorities. This is compounded by the fact that the European Union's environmental policies – a natural reference for policy-makers in CEE countries – have also largely ignored the tradable permit option.

In August 1998, yet another Environmental Protection Act was drafted as a European Union-funded (PHARE) project. This draft differs from earlier versions in many respects. Its volume is larger and it includes a number of new provisions considered to be novel in the Polish legal system. For instance, public participation in environmental management and access to environmental information have been developed rather thoroughly. Simultaneously, sections of the Act dealing with economic instruments are very much the same as they were in the 1996 version.

The 1998 proposal also devotes somewhat more attention to TPPs. These are introduced in Articles 88-90, and attached again to the chapter dealing with offsets. Article 91 authorises the Minister of Environment to issue an ordinance laying out specific rules for emissions trading. A draft of such an ordinance – based on the Opole project – is attached to the bill. The new documents cannot be considered a breakthrough in the legislative process, but at least they maintain the momentum toward TPPs.

Will TPPs be implemented in Poland within the foreseeable future? An optimistic scenario envisages growing pressures towards making Poland's environmental policies more cost-effective. The outcome of this process must be a legal framework that recognises the transferability of permits. A pessimistic scenario extrapolates the political logic of the last few years. It envisages more studies, more drafting committees, more Environmental Protection Act revisions, and more cost-saving opportunities missed.

According to a popular dictum, "there is no constituency for efficiency". Even though cost-saving opportunities in environmental protection are irreversibly lost when abatement is achieved by traditional instruments, no authorities seem to be bothered by this fact. Administrators are satisfied whenever they can point to some polluters being forced to reduce pollution. Questions concerning whether these reductions significantly contributed to ambient quality, or whether they incorporated least-cost options, are considered to be of secondary importance.

Those who advocate TPPs are relatively few, and fairly dispersed. Power plant managers have become the first and (for the time being) the only industrial stakeholder group fully aware of potential benefits to be gained from this instrument. Before the electricity sector started to appreciate those benefits, there were very few promoters of the idea in Poland. Following the Chorzów project, several heavily-contaminated municipalities revealed interest in addressing their environmental problems in a more systematic and cost-effective manner. Soon, however, they turned to looking for money available on concessional terms, for projects with more political appeal. This proved to be more immediately effective than lobbying for a new instrument. Thus, TPP promoters failed to develop a broader constituency for the necessary policy reforms.

Potential constituencies should be sought among at least three principal social and political groups: environmentalists, businessmen, and politicians. Most of the educational efforts mentioned earlier were aimed at the first group, and they turned out to be effective in some instances. The result is that no important environmental NGOs in Poland now have significant reservations about TPPs. To the contrary, environmental activists have often embarrassed subsequent Ministers of Environment at press conferences by asking about delays in implementing emissions trading. However, scepticism among environmental lawyers has persisted, which can be explained mainly by the reluctance to depart from regulations that have been practised for decades. Another important professional group opposing TPPs is environmental inspectors. These favour negotiated compliance schedules as an alternative to TPPs, whenever flexibility is required. This preference is quite understandable, since permit markets would substantially reduce inspectors' power and prestige. Therefore, the question is not how to "convert" inspectors to TPPs (they have vested interests in keeping the legal *status quo*), but rather how to create a countervailing power.

Lobbying politicians should not be neglected, but the outcome is far from predictable. The political scene is fragmented, and politicians have notoriously short planning horizons. Under these circumstances, environmentalists should not expect too much from their efforts to influence this particular group.

The business community is perhaps the most serious potential political ally capable of appreciating the potential benefits from constraining the power of the environmental bureaucracy. At the same time, environmentalists in Poland have seemed to avoid businessmen in their lobbying activities. This could be just a matter of an erroneous strategy. More circumspect analysts, however, may link this reluctance to the ambiguous reputation that the emerging business community enjoys. Examples of honest entrepreneurship, good citizenship, and responsiveness to charitable activities mix with evidence of bribery, fraud, and black market operations. By no means is it certain that the most outspoken business leaders will support a shift towards less discretionism, and more "invisible hand" regulations, *in lieu* of inspectors' decisions. For whatever reason, Polish environmentalists have not actively sought political support from business community yet.

Environmental policy efforts in retrospect

Brief interviews with several key decision makers and experts were carried out in the summer of 1998, in order to solicit their opinions on the Polish environmental reform process, especially as it

relates to emissions trading (see Annex I for the list of interviewees). Each interviewee was asked the same two questions:

- How do you view the Chorzów project from the 1998 perspective? and
- What are the main obstacles to introducing TPPs in Poland?

Given their diverse backgrounds and interests, only a few interviewees focused on the first question, while most did on the second one. The small sample size did not allow the drawing of any statistical inferences. Nonetheless, the interviews did reveal a number of interesting aspects of the environmental policy reform process in Poland.

All but one interviewee saw the Chorzów project as a useful exercise. It is credited with playing the role it was supposed to play (*i.e.* to demonstrate that emissions trading can accelerate environmental improvement while saving money). Additionally, one expert reiterated his earlier assessment (Beblo, 1994) of the project as a lesson in how to reach compromise among various stakeholders involved in the abatement investment process.

An executive from the Chorzów Power Plant was the only one dissatisfied with the project. As other power plant managers, he understands and appreciates the concept of emissions trading. Nevertheless, he stressed that the Chorzów project has never been fully implemented and that no real trades have ever taken place. In his view, the power plant had not been treated fairly. Its emissions had increased as a result of taking over some of the steel mill functions, but the revised pollution permit was not sufficient to match the new situation. The power plant had been “overcharged” by the Katowice regional environmental fund, while the steel mill had received overly-generous treatment.

Responses to the second question were much more extensive. All interviewees agreed that TPPs should (and could) have been introduced to environmental management in Poland already, but they differed in explaining the failure to have done so.

An often-voiced opinion was that no further preparatory steps are now required. Whatever was necessary has already been accomplished: the adequacy of TPPs in transition economies was analysed by academic teams; the Chorzów project demonstrated that things can work in practice; and the Opole project provided the required legislative framework. Hence, it is just the lack of a political will and bureaucratic inertia that postpone the successful conclusion of the policy reform process that had begun in 1989-1991.

While most respondents were critical about subsequent Governments' dedications to the TPP cause, one business executive blamed the Parliament, rather than the Executive Branch, for postponing the necessary legal amendments. *Ad hoc* committees or even individual members of Parliament often revise government proposals, reducing their comprehensiveness. While the specific examples he gave were related to economic instruments, none of these was directly relevant for TPPs (it could not have been otherwise, since none of the draft Environmental Protection Act have ever been formally submitted to Parliament by the Government).

The lack of interest in TPPs that has consistently been revealed, both by successive Governments and Parliaments, indicates that the demand for emission reduction credits must be low. The opinion was expressed that the only way to change this situation would be to strictly enforce existing emission limits. As long as polluters feel that they can negotiate with inspectors on how much pollution will be allowable in their particular case, they will prefer to do so, instead of buying emission reduction credits from someone whose environmental performance was better.

One interviewee indicated that there was an evolution in bureaucratic attitudes towards TPPs in the Ministry of Environment. Initially, policy-makers were reluctant to accept an instrument they did not understand very well, but knew that (for whatever reason) it was virtually not applied in Europe. Another important reason cited was that they were afraid that emissions trading might interfere with the pollution charge system already established in Poland. Later on, once the understanding of the mechanism grew, the emphasis was switched to potential incompatibility with the European legislation in general, and the IPPC Directive in particular.

Most interviewees did not question the adequacy of the legal framework designed within the Opole project. Only one expert admitted that this framework was overburdened by its complicated design, including exchange rates. This might have also contributed to the lack of interest revealed by administrators. Another interviewee suggested that the specific regulations developed by the Opole project team were not quite satisfactory. The proposed system of approving and tracking permit transfers is sufficient, but not all the legal issues (*e.g.* the responsibility of parties, ownership titles, etc.) have been successfully resolved. He suggested that a good administrative law expert should have worked on these issues more systematically.

One of the lawyers interviewed also confirmed that the Ministry of Environment officials revealed little interest in TPPs. He contrasted this with “negotiated compliance schedules”, an alternative instrument of adding flexibility to environmental management in non-attainment areas. He felt it was quite predictable that environmental inspectors would insist on developing “negotiated compliance schedules” in every possible detail, so that this instrument could be applied smoothly. In negotiating compliance, inspectors can exercise their power most effectively. They discuss the technical details of abatement installations, press polluters to accelerate their investment efforts, monitor progress, assess fines, and they can shut down facilities if a deadline is missed. This system obviously invites corruption, but even if all the inspectors were 100 per cent incorruptible, “negotiated compliance schedules” would be a source of satisfaction, power, and prestige for the officers who develop and implement them. TPPs can never be a source of the same kind of power for any professional group, so it is not surprising that they have very weak constituencies among administrators and legislators.

Summary, conclusions, and policy recommendations

The question of the availability of TPPs to economies in transition has been researched since the 1980s. Assessments carried out by both Polish and international experts leave no doubt that in many applications, emissions trading is a preferred alternative, not only to conventional “command-and-control” regimes, but also to emission charges that exist in Poland and elsewhere throughout the CEE region. Based on such judgements, various attempts have been made to implement TPPs in Poland. These range from a demonstration project, to education campaigns, and to legislative proposals. However, apart from a handful of environmental economists (who advocate on behalf of cost-effectiveness) and power plant managers (who realise that their sector can substantially gain from [at least internal] emissions trading), there are virtually no groups pressing for the routine application of TPPs.

Apparently, the political economy of the transition period does not favour introducing market regulations in environmental management. TPP legislative proposals consistently lose out to alternative proposals based either on inertia or on interests of well-identified professional groups. So far, advocates of TPPs in Poland have failed to create a countervailing power to raise the demand for this new policy instrument.

An implementation strategy should recognise the most likely factors behind the failure to introduce TPPs in Poland over the last 9 years. These can be categorised as being either domestic or external. The former include environmental administrators’ preference for technology-based, rather than ambient-quality-based approaches; the lack of an appreciation of cost-effectiveness; as well as unsolved legal and procedural controversies regarding permit transfers. The latter boil down to the absence of TPPs in the European Union legislation, and its orientation towards technology, rather than toward ambient quality solutions.

While external factors are beyond the control of the Polish environmentalists, the following steps may help to remove the domestic obstacles to the implementation of TPPs.

- Emphasise human exposure, nature protection, and environmental quality as primary policy objectives; abatement technology is but a means to achieve these objectives.
- Emphasise the need to use resources cost-effectively.
- Mobilise NGOs and identify “champions” among politicians, administrators, businessmen and legislators to promote these ideas.

- Invest in legal expertise to eliminate controversies associated with planned procedures of permit trading.

No additional feasibility studies seem to be necessary, since TPPs have already been researched rather comprehensively. New demonstration projects are not likely to push the case either. What is needed is professionally-designed political action to let the new instrument compete on a fair basis with alternatives that are preferred by already-entrenched interest groups.

Annex 1

LIST OF INDIVIDUALS INTERVIEWED IN 1998

- Wojciech Beblo, environmental consultant, CITEC, Katowice; Regional Environmental Administrator in Katowice in 1990-1994.
- Krystian Golabowski, Technical Director, the Chorzów Power Plant, Chorzów.
- Wojciech Jaworski, senior officer, Polish Electricity Grid Company, Warsaw; Director, Air Protection Department of the Ministry of Environment in 1991-1997.
- Jerzy Jendroska, environmental lawyer, co-author of the Environmental Protection Act drafts, State and Law Institute, Polish Academy of Sciences, Wrocław.
- Józef Klose, President and CEO, Kosciuszko Steel Mill, Chorzów.
- Maciej Nowicki, President, EcoFund, Warsaw; Minister of Environment in 1991.
- Marek Sobiecki, Acting Director, Environmental Policy Department, Ministry of Environment, Warsaw.
- Stanisław Wajda, PHARE Programme legal advisor, Ministry of Environment, Warsaw.

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Chapter VIII

OBSTACLES TO THE IMPLEMENTATION OF TRADABLE PERMITS: THE CASE OF SWITZERLAND

by

Claude Jeanrenaud

Institut de recherches économiques et régionales
Université de Neuchâtel, Neuchâtel, Switzerland

A VOC trading market for the Basle Region

The Basle region was the first in Europe to have incorporated the conditions authorising emission trading in its environmental legislation. The project in question was initially designed for both nitrogen oxides (NO_x) and volatile organic compounds (VOCs). All implementation-related obstacles were overcome and the experiment was launched early 1993. However, the Swiss federal authorities reduced NO_x emission ceilings late 1992. As that meant that cantonal and federal standards were identical, the potential for trading in NO_x emissions had already vanished before emission trading could start.

Chemicals and pharmaceuticals are the backbone of the regional economy, where several multinationals are based. These firms' production centres are the main potential participants in the emission trading market. To date, the experiment has not yielded the hoped-for results. Very little trading has taken place over the past five years, and there has not been any outside trading giving rise to a payment between two firms. After presenting the design and characteristics of the market established, an effort is made below to understand the reasons why firms do not take greater advantage of the opportunity made available to them to achieve environmental goals at lower cost.

At the outset, it should be noted that the project was prepared at a time when environmental protection was given very high priority by the Basle authorities, by the public-at-large, and by the employees of the firms concerned. The firms were prepared to spend large sums to clean up their stationary sources, and the environment enjoyed a higher position than almost all other stakeholders (shareholders and employees). No one brought up the threat which a very ambitious environmental policy posed to corporate competitiveness and jobs. The general context and the strong position of environmental protection movements explain why emission trading was then subjected to such restrictive conditions.

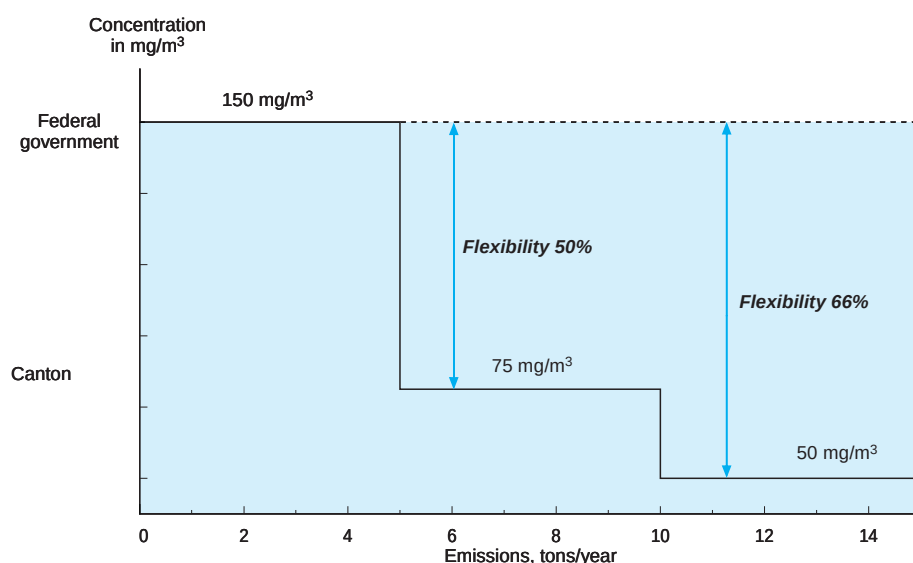
The context

The Federal Government's pollution control strategy (1986 Strategy to Combat Air Pollution) stipulated that emissions of NO_x and VOCs had to be reduced to at least 1960 levels by the year 1995. This goal necessitated a 69 per cent reduction in NO_x emissions and a 57 per cent reduction in VOCs (1993 Report on the State of the Environment). To meet these goals, the Federal Council adopted a Clean Air Act, which went into force in early 1986 (OPair 1986). This instrument contains prescriptions concerning emission ceilings for atmospheric pollutants.

The Cantons are responsible for applying federal environmental legislation. If air quality standards are not attained on the territory of a Canton, it must draw up a plan consisting of measures to reduce the pollutants in question. This may involve a tightening of federal emission ceilings or the introduction of transport-related measures, for example (speed limits).

When they noted that federal air quality standards for the ozone and NO_x were not being attained on their territory, the two Cantons of Basle decided to impose more stringent emission limits on major emitters of VOCs and NO_x . In 1990, VOC emission ceilings were lowered by 50 per cent for all firms emitting more than five tons per year, and by 66 per cent for all those emitting more than 10 tons a year. Thus, in these two Cantons, a dual ceiling was in force: that of the Federal Government and the more stringent one applied by the Canton (Figure 1).

Figure 1. **Federal and Cantonal Emission Standards (VOCs)**



In 1992, the new Clean Air Act (OPair 92) introduced more stringent emission standards for NO_x , practically eliminating any difference between the Basle standard and the federal standard. As this difference constituted the only manoeuvring room for trading, the very existence of the permit trading market for NO_x emissions was called into question. Accordingly, the Basle experiment with emission trading has been limited to VOCs.

In 1995, the VOC emission structure was as follows:

Table 1. **VOC Emission Structure in the Basle Region in 1995**

	Tons/year
Industry	12 500
Domestic	2 500
Transport	2 200
Total	17 200

Source: Lufthygieneamt beider Basle.

By setting up an emission trading market, the Basle authorities wished to offer compensation to firms subject to more stringent standards. The facilities concerned are of course subject to strict standards, but firms have more flexibility when it comes to compliance. The aim was to encourage firms with facilities whose emissions could be reduced at a reasonable cost to make additional abatement efforts, and to reduce emissions to below the cantonal limits.

Market design

General conditions

Since 1993, large VOC emitters in the Basle region have had access to two instruments for emission trading: bubbles and tradable credits. The system resembles the earlier US model (emission trading policy), while featuring a few differences as far as instrument design is concerned. It is not an allowance trading programme, but rather a credit system without initial allocations.

Trading must focus on quantity. Yet, as the legislation currently stands, all standards are expressed in terms of concentrations. Thus, it was necessary to translate the authorised emission concentrations into quantities (kilos or tons). When calculating quantities, due consideration was given to average concentrations, to hourly outputs, and to the average annual duration of plant operation.

Since the federal standard could not be exceeded, manoeuvring room was limited to the difference between the concentration authorised by the Swiss Clean Air Act (OPair) and the lower one set by the Canton. For sources emitting between 5 and 10 tons per year, the margin of flexibility was 50 per cent, as against 66 per cent for the (few) plants emitting more than 10 tons. Plants emitting less than 5 tons per year were not granted any flexibility, and were thus necessarily excluded from the market.

Those who set up the new system did not wish the operation to benefit only industry, via reduced spending on abatement efforts. They decided to make the system also benefit nature and, hence, the general public. Thus, part of the emission reduction (20 per cent) is final; it cannot be offset by an increase in emissions from another source. In practice, this translates into a trading ratio of 0.8. When the project was in the planning stage, its designers were counting on a trading ratio of 0.7. After strong opposition from industry, the legislators finally opted for a ratio that was slightly more favourable for firms.

Tradable credits

A firm which reduces its emissions from one of its stationary sources to below the cantonal limit can ask the Cantonal Environmental Agency for a tradable credit, which it can then sell to another firm, or "bank" for future use.

In order to take due consideration of the uncertainties linked to measuring emissions and maintaining administrative costs within reasonable limits, emission reductions below the cantonal limit must exceed 10 per cent. Beyond this threshold, the firm can receive permits equivalent to 80 per cent of the emission reduction. It will be recalled that the 80 per cent rule was introduced with a view to ensuring that industry and environment shared in the benefits of the trading operations. This sharing was no doubt the prerequisite for persuading ecologists to agree to an easing of current regulations. This dual condition constitutes an obstacle to trading and prevents firms from taking full advantage of differences in marginal abatement costs in order to enhance the efficiency of air protection measures. A gap between marginal costs will remain once the market has reached equilibrium. At the beginning of the experiment, it was planned that the Canton would set up a credit trading exchange. Finally, it may be noted that permits issued by one of the two Cantons are accepted in the other.

Table 2. **Allocation of VOC Emission Credits: An Example**

	Tons/year
Emission level according federal standards	40
Emission level according cantonal standards	20
Safety margin (10%)	2
Baseline emission level from which reductions are measured	18
Actual emissions	8
Quantity used for credit allocation	10
20% reduction	2
Emission credits allocated	8

However, credit allocation is not automatic. Companies envisaging abatement operations must first obtain an *option* from the regulator, which checks to make sure that the emission reduction comes from a real abatement effort, and not from special circumstances (for example, a plant shutdown or slowdown on economic grounds). Once the credit is allocated, its use is still subject to approval. If there is a danger that “hot spots” can arise, the regulator vetoes the permit sale.

Permits can be “banked” for future use, but they lose 20 per cent of their value each year. The reason for such a high depreciation rate is not clear. It appears that the legislators feared that firms would accumulate large quantities of emission rights over long periods.

The system is still strongly imbued with a “command-and-control” culture. Moreover, laws in the two Cantons provide that the regulatory authority will not issue credits for small quantities (“Bagatellenmengen”, Article 9, para. 3), in order to reduce administrative costs.

The main characteristics of the emission trading system set up in the Basle region are as follows:

- for the majority of sources, there has been no change with regard to the previous regulatory regime, as firms remain subject to emission ceilings (OPair 92 and VVESA BS/BL);
- the Basle standards can be exceeded inasmuch as any overshooting is offset by compensation. In this case, compensation is sanctioned by the sale of a credit;
- there is no need to use permits to start up a new plant; all the operators have to do is to avoid exceeding the Basle ceilings;
- there is no cap on total emissions.

“Bubbles”

The idea is that an imaginary bubble is placed over a set of sources, and only the total quantity of pollutants emitted under the bubble is taken into consideration. Thus, firms are free, within certain limits, to offset excess emissions from one plant by a reduction made on another source, as long as overall quantity is not exceeded.

Most frequently, the bubble is limited to sources belonging to a given plant, but it can also encompass sources belonging to several plants. Under cantonal law, total emissions must be at least 15 per cent less than the sum total of emissions authorised for each individual source. The reduction of 15 per cent corresponds to the safety margin, and makes it possible to share the benefits of the trade between industry and environment. The 15 per cent limit was calculated on the basis of a model to ensure that facilities are treated identically with the two instruments. As with permits, federal emission standards may not be exceeded. The setting-up of a bubble must be approved by the cantonal regulatory authority (the two Cantons share the same environmental agency).

Introducing a bubble considerably simplifies trading and administrative procedures. If the bubble covers sources belonging to a single firm, trading is implicit and does not give rise to any payments (internal trading). Permits are no longer needed for emission trading, and the regulatory authorities are not required to approve each transaction. All that is needed is the authorities' approval when the bubble is set up.

The two instruments may be combined. If the various sources under the bubble emit an overall quantity of VOCs less than that authorised in the light of the 15 per cent deduction, the firm in question is entitled to permits. In the example below, the firm obtained tradable permits allowing it to emit 29 tons.

New sources

When the project was in the preparatory stage, consideration was given to a system of offsets for VOCs and NO_x between new and existing sources in a given region. The proposed solution was comparable to the strategy adopted in the US in the 1970s for non-attainment areas. In order to avoid an increase in overall emissions, the new plants were to be approved, on condition that an equivalent reduction was obtained at existing sources. This proposal gave rise to strong criticism by industry, and was finally dropped.

Functioning of the market

To date, the regulatory authority has approved the creation of four bubbles and a fifth request is being examined. Each bubble covers only sources from a given production unit belonging to one of the major chemical groups. Thus, only internal trading takes place. For the Novartis production unit in Basle, quantities of authorised VOC emissions have decreased by 26 tons (Table 3). For the four bubbles set up to date, authorised quantities have been lowered by 70 tons per year (Nyfeler, 1996). There is no overall estimate of the decrease in abatement costs following internal trading. However, for Novartis, the flexibility made possible by the bubble has led to verifiable cost-savings of several millions. Internal offsets have made it possible to avoid installing equipment worth SF 7 million (Eigenmann, 1996).

Table 3. **Combination of the Two Instruments: Bubbles and Tradable Credits for VOC Emissions***

Emission level	Tons/year
According federal standards(OPair 92)*	373
According cantonal standards	172
15% reduction	26
Emissions authorised under bubble	146
1995 emissions	
Actual emissions under the bubble	110
Gap	-36
20% reduction	7
Credits allocation	29
<i>Source:</i> Eigenmann (1996). Data is for the Novartis production unit in Basle.	

For the large Basle chemical plants, VOC abatement costs are soaring. Ever since major abatement efforts began in 1986, the cost of reducing pollution has increased threefold, rising from SF 10 000 to SF 30 000 per ton.¹ Offsets between sources have made it possible to avoid residual abatement, which features a very poor cost-effectiveness ratio.

The tradable credit market has been virtually non-existent to date. There has not been any demand for credits, and thus, no external trading. Moreover, only a few tradable credits covering an overall quantity of 61 tons have been allocated.

Obstacles to trading

The reason most frequently put forward to explain the absence of a market is the federal authorities' refusal to agree to ease the standards of the Federal Clean Air Decree (OPair). This is undoubtedly one of the causes of the low level of interest aroused by the new instruments, but it is not the only one. The market design, the fact that the two Cantons introduced emission ceilings for large emitters only (> 5 tons/year), the time chosen for the introduction of tradable permits, and the lack of interest in external trading are elements to be kept in mind if we are to understand the reasons for the relative failure of the Basle experiment.

Insufficient flexibility

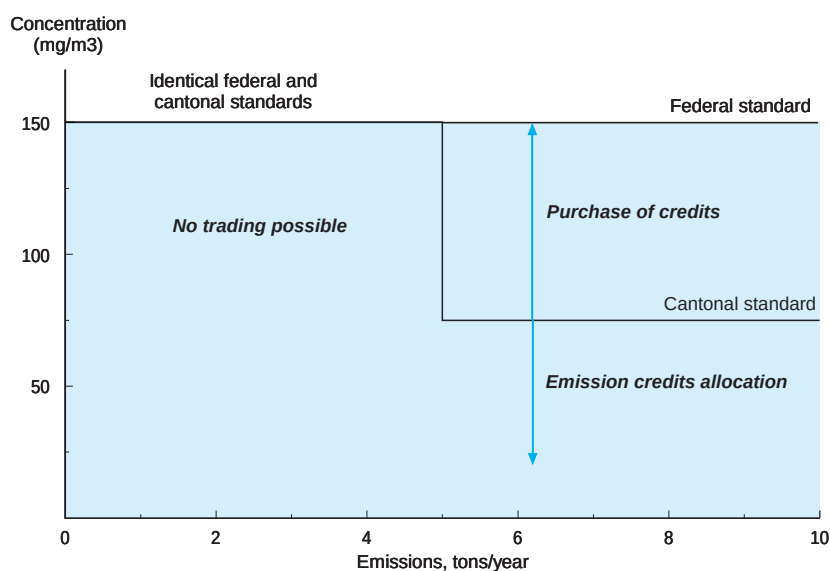
The market is thought to lack manoeuvring room to operate normally which in turn explains the limited number of transactions that have taken place to date. This hypothesis is often put forward. The Federal Government cannot accept a situation where the emission standards laid down in the OPair are exceeded.² Consequently, the flexibility margin for offsetting is limited by the gap between the federal

and cantonal standards. Yet this explanation is not entirely convincing, since there is a considerable difference between the two ceilings (a ratio of 1:2 for sources emitting between 5 and 10 tons per year, and even 1:3 for sources emitting more than 10 tons). However, the total lack of flexibility of federal standards does not allow small firms – those with facilities which annually emit less than 5 tons of VOCs – to participate in the market. The other reason for excluding small sources is the decision of the two Cantons concerned not to modify emission standards for sources emitting less than 5 tons per year.

Staehelin-Witt and Spillmann (1992) give another explanation for the low intensity of trading. When a firm carries out an abatement effort, it has little incentive, from technological and economic points of view, to comply solely with the federal standard. Most often, investing in abatement also makes it possible to comply with both federal and cantonal standards. This is thought to be a reason for the lack of demand for permits.

Part of the difficulties encountered are no doubt due to the rigidity of the federal standards. However, the fact that trading has failed to develop as expected is also due to other factors. Otherwise, there is no explaining the presence of internal trading under the bubbles and the total lack of external trading.

Figure 2. Flexibility Margin for Emission Trading



Characteristics of the emission credit system

- Quantity or concentration

Observers of the Basle experiment feel that the system established is not a true market for tradable permits, since firms remain subject to emission concentration standards (Staehelin-Witt and Spillmann, 1992; Eigenmann, 1996). Indeed, the emission standards for the various types of facilities are maintained, as firms have the possibility of emitting higher concentrations of VOCs, on the condition that offsetting takes place. Thus, the system adopted by the two Cantons is a credit system, and it remains very close to a regime of direct regulation.

The arguments put forward by Staehelin-Witt and Spillmann are not totally convincing. For all firms wishing to participate in the market, emission concentrations are converted into quantities, by forming a bubble or by requesting a credit. In this case, trading is denominated in kilos or tons, not in mg/m³. On the other hand, it is true that participation in the market is optional.

- Absence of initial distribution

The fact that allowances are not distributed at the start of each period constitutes one of the characteristics of the Basle model. Therefore, there is no cap on total emissions. Firms which limit themselves to complying with the initial emission standards do not see any change, when compared with direct regulation. They do not need permits if they comply with the emission standards for each of their facilities. However, the situation is different for firms which introduce more stringent abatement measures than those required by the law, and which reduce their VOC emissions below the cantonal limits. Such firms may immediately engage in internal trading (if they have received approval to set up a bubble) and sell the credits obtained insofar as they find a buyer or bank the credits for future use. Direct regulation remains the rule, and the purchase or sale of permits and implicit trading under a bubble remain the exception.

- Treatment of new sources

One of the advantages of a “cap-and-trade” permit system is the fact that the regulatory authority monitors the total quantity of pollutants emitted in the region, with a view to achieving a particular air quality goal. The Basle system does not offer this advantage. Companies which start up a new source are not obliged to purchase credits from other firms in the area if they comply with emission standards. Moreover, this is one of the reasons why (to date), there has not been any demand for credits, and therefore no transactions.

- Exclusion of small emitters

With a limit set at 5 tons per year, only large plants belonging to the chemical or pharmaceutical industries have stationary sources emitting enough VOCs to participate in trading. Yet they have already made major efforts to reduce their emissions, and marginal abatement costs are very high today. Smaller sources (< 0.1 kg/h) have not been affected by the clean air measures, which in turn explains why abatement costs are generally lower there than they are in the chemical sector. If it were possible to group together all stationary sources in a single market, the significant differences between marginal abatement costs would constitute a powerful incentive for trading and the market would be more active. Yet federal and cantonal regulations do not allow trading between sources which emit more than 5 tons and those which emit smaller quantities. This artificial limit represents a barrier to trading and leads to a loss of efficiency (measured on the figure below by the difference between the grey surfaces).

- Stringency of restrictions on trading

The regulation which makes the allocation of credits contingent on an emission reduction of at least 10 per cent below the cantonal limit also reduces the potential for trading. Given the transaction costs, the difference between marginal costs for buyer and seller must clearly be more than 10 per cent in order for a transaction to take place. The regulation fixing the trading ratio at 0.8 has the same effect. Due to these two constraints, there must be very significant differences between the abatement costs of the two parties interested in a trade for it to take place. An example with figures (Table 4) reveals the extent to which the two regulations mentioned above constitute an obstacle to trading. Imagine that marginal abatement costs, which are assumed to be constant, are SF 30 000 per ton of VOCs in Plant A and SF 60 000 in Plant B. Further suppose that transaction costs come to SF 2 500 for each of the parties. Plant A has already made abatement efforts and its VOC emissions – 8 tons per year – meet the authorised limit. On the other hand, Plant B must reduce its emissions by 1 ton per year to comply with legislation. It is not difficult to imagine that A, which has significantly lower abatement costs, could reduce its emissions in the place of B and obtain compensation by selling permits. Although abatement costs are very different, no trading is possible between the two plants. Indeed, the minimum amount required by the seller is more than what the buyer is prepared to pay.

Environmental officers in major firms have not appreciated the decision to erect such substantial barriers to trading. However, they feel that this is not the determining reason for the low intensity of trading.

Figure 3. Absence of Trading Between Large and Small Emission Sources

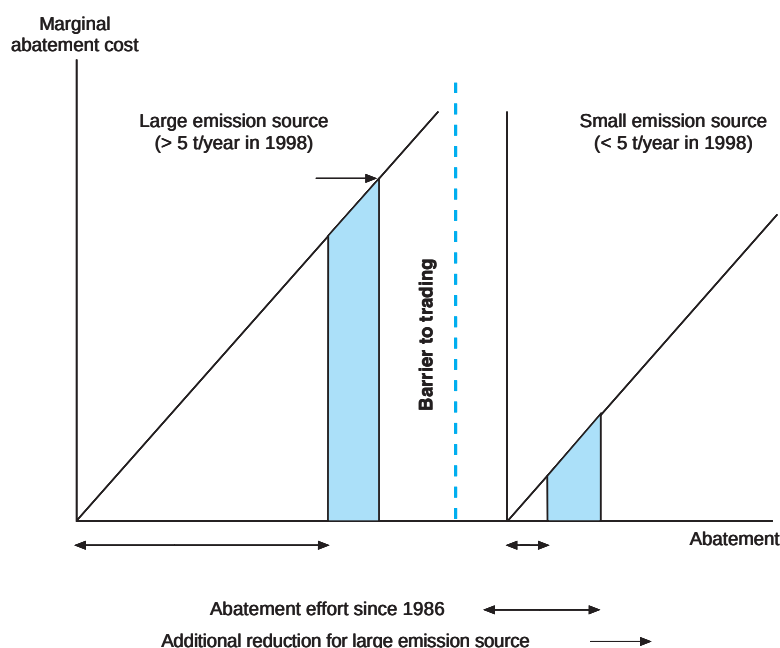


Table 4. Restrictions Limiting the Potential for Trading

	Plant A	Plant B
Actual emissions	8 t	9 t
Emission reduction required		1 t
Emission reduction of emissions to obtain a credit for one ton (0.8 t + 1.25 t)	2.05 t	
Abatement cost in CHF (2.05 t)	61 500	
Transaction cost in CHF	2 500	2 500
Minimum amount in CHF which A requires to participate in trade	64 000	
Maximum amount in CHF that B is prepared to pay to avoid abatement expense		57 500

- Cumbersome procedure

Emission credits are subject to dual approval. When the request is submitted, the Cantonal Environmental Agency sees to it that the emission reduction comes from an abatement effort made by the firm and not special circumstances (a temporary standstill of a facility, a decrease in activity for economic reasons). The sale must also be approved by the regulator, the goal being to avoid an increase in emissions in non-attainment areas. This is one of the causes for the increase in transaction costs. However, the firms surveyed adapt quite well to these few administrative constraints, and do not consider that they represent a significant obstacle to trading.

- Insecurity

Proper transparency and clear operating rules are the prerequisites for developing the market. In the Basle case, the discretionary power of the regulator to decide whether or not a reduction of VOC emissions entitles a firm to a credit constitutes a problem. At the time they decide to make a supplementary abatement effort, firms do not know if they will be able to obtain compensation.

Moreover, the firms surveyed feel it is unfair that the value of the credits banked diminishes every year. As there may be no demand for the moment, a firm which makes a substantial abatement effort may fear that it will not obtain any compensation. Credits which are not used or exchanged within a five-year period lose all their value.

There is a third insecurity factor: the law does not foresee the impact of tighter emission standards on the value of the rights. Thus, firms do not know if the value of the credits will be adjusted to fit more stringent emission standards, or if it will remain the same.

Acceptability

- By firms

Most of the facilities concerned belong to the big chemical or pharmaceutical multinationals. These firms are making efforts to portray themselves as firms respectful of nature. Thus, they cannot adopt, in their environmental policy, behaviour which may shock part of the population, or damage the firm's image. When these firms are asked if they would consider paying (*i.e.* purchasing an emission credit from another firm) rather than taking abatement measures, the answer is clearly negative. On the other hand, it is easier to imagine that these firms might participate in the market as sellers of rights. The most likely scenario is that these firms will bank permits for internal use.

Bubbles are merely viewed as an alternative means of controlling emissions (by quantity rather than concentration). This type of instrument is very well-accepted by firms, which would even like to see it come into widespread use. On the other hand, firms are much less enthusiastic about the sale of permits or (above all) their purchase (which is perceived as buying a "right to pollute"). There are certainly cultural reasons for the poor acceptance of this instrument. Environmental officers in major firms have indicated that, in the 1980s, when all firms were making major abatement efforts, they could have considered purchasing rights enabling them to postpone an abatement operation. Today, the conditions have changed, and they would no longer enter the market as purchasers.

- By the federal regulator

The Office fédéral de l'environnement, des forêts et du paysage (OFEFP) has never been very enthusiastic about the idea of using emission certificates to reach its policy goals. A study carried out for another federal office highlighting the advantages of permits in a clean air policy met a rather cool reception (Stritt and Jeanrenaud, 1992). However, it would be a mistake to conclude that the OFEFP is against this type of instrument in principle. In the context of its policy of reducing emissions of greenhouse gases, tradable certificates are viewed as one of the means of achieving the very ambitious goals deriving from the Kyoto Protocol. In the case of VOCs, the OFEFP's opposition to permit-based experiments, and the refusal to accept an easing of federal emission standards, can be explained by the fact that the Federal Government had opted for a tax. The recent amendment of the Law on the Environment made it possible to levy taxes on VOCs. The degree came into force in early 1998, and the tax will be levied starting January 2000. The tax on VOCs, along with the tax on extra-light heating oil,³ will be the first incentive tool of environmental policy at the federal level. The OFEFP feels that owing to the specific structure of VOC emissions – many small sources and diffuse emissions – the tax is a more appropriate means of action, and one which is easier to administer than tradable permits. The federal regulatory authority no doubt considers that simultaneous recourse to the two instruments is not desirable.⁴ The Basle firms participating in the permit market would also be subjected to the tax.

- By society

No work has been done on the way in which permit trading is perceived and accepted by industry and the population in Switzerland. Nevertheless, interesting information may be found in a study by Bürgenmeier *et al.* (1997) on ecotaxes. In general, the acceptability of incentive taxes by firms seems rather good, as two-thirds of them have expressed a favourable view. This result is surprising if one brings to mind the opposition of industry to the Federal Government project to introduce a tax on CO₂.⁵ The major firms in the Basle area have supported the project to set up an emissions trading market. Talks with

environmental officers reveal that trading carried out within one plant are generally better accepted than external trading. The idea of selling a “right to pollute” has not really been accepted by firms or the public in general.

Timing

The abatement efforts in the chemical industry were made between 1986 and 1994. During this period, VOC emissions from sources belonging to the major firms in the industry diminished by more than 90 per cent. As far as these firms are concerned, the bulk of their investments in abatement efforts have already been made. Thus, the right time to introduce a system of tradable permits was during the mid-1980s, which means that the new instruments came too late. According to the firms surveyed, this was one of the primary causes of the experiment's lack of success.

Emission structure

Those who designed the project in the early 1990s based themselves on the idea that 80 per cent of VOC emissions came from 20 per cent of all sources. Had this hypothesis proved correct, a system of tradable permits reserved for major VOC emitters would have perhaps performed as expected. Yet the emission structure today is quite different, as the lion's share of VOCs comes from small stationary sources and traffic (Staehelin-Witt, 1998). Over the past few years, the large chemical and pharmaceutical plants have made major abatement efforts, reducing their VOC emissions by more than 90 per cent. Today, some 1 000 tons/year come from the chemical industry – including emissions from small firms in the sector – out of a total of some 17 000 tons for all of the sources in the Basle region. VOC emissions from each of the production centres of the major chemical and pharmaceutical firms come to around 100 kg per year. Thus, the new emission structure is less favourable for emission trading than the initial structure.

Reorientation of corporate priorities

In the second half of the 1980s, the general conditions were propitious for the implementation of an ambitious environmental protection policy. The favourable business climate, the support of the local population and the employees of the firms being targeted all supporting the federal and cantonal clean air policy led the major chemical and pharmaceutical firms to earmark very substantial amounts of money for the fight against pollution. The effort which the major companies were asked to make was out of proportion compared to the effort which small and medium-sized firms and owners of motor vehicles were asked to make.

In recent years, however, as a result of globalisation, competitive pressures, mergers and employees' fears for their jobs, the environment no longer enjoys as high a priority as it did when the project was originally designed. Today, firms attach greater importance to their responsibilities with regard to other stakeholders and in particular, shareholders. Employees are more concerned with holding on to jobs than they are with air quality. All these elements explain the tepid reception for a new cap-and-trade model, very similar to the program of tradable permits already in use in the US (BSS, 1995; Staehelin-Witt, 1998). The big firms feel that they have already done quite a bit, that additional reductions would not be effective given the very high marginal abatement costs, and that it is time to shift the burden of abatement efforts to small and medium-sized firms and motor vehicle owners.

Prospects

Noting that the two instruments – bubbles and tradable credits – were used only a little (or not at all); that the system had major drawbacks; that it was no doubt too close to a system of direct regulation; and that the clean air goals were not being attained, the Basle authorities gave thought to the changes to be made to the model, so that it could better meet the needs of firms and of environmental protection. To this end, a consulting firm was asked to do a feasibility study (BSS, 1995).

The model proposed was very close to the one in use in Southern California (SAQMD, 1991a, b and 1993; Trüeb, 1995; Staehelin-Witt, 1997). Permits would no longer be defined on the basis of authorised concentrations but on the quantities actually emitted. At the beginning of the period, the author-

ities would distribute a quantity of permits compatible with the clean air goal, and divide up the rights based on quantities emitted previously ("grandfathering"). Thus, firms opening up new facilities would be obliged to buy permits on the market, or engage in internal offsets. In order to increase the number of market players, it was decided to include all stationary sources emitting more than 2 tons. The obligation to announce trades to the regulatory authority and the possibility for that body to veto such a transaction have been maintained. The Environmental Agencies in the two Cantons would start by determining the total quantity of VOCs deemed to be compatible with attainment of the air quality standards (ozone and NO_x). On that basis, it would set an identical emission reduction rate for all market players. In order to reach this goal over a ten-year period, the quantity of permits allotted to each firm would be gradually reduced. Implementation of the new system would require a change in federal legislation.

As the industry in general and the large firms in particular had already made a major abatement effort, they were against the introduction of the new model. In the major chemical and pharmaceutical firms, any additional emission reductions would result in prohibitive costs, since all economically reasonable abatement measures had already been introduced. Imposing new constraints on these firms could make the region less attractive for the export industry (Staehelin-Witt, 1998). On the other hand, the industry would be favourable to replacing emission standards expressed in terms of concentrations by ceilings denominated in kilos or tons. Making such a change to regulations would be tantamount to ensuring that internal trading came into widespread use, since each establishment would by definition form a bubble. It would be necessary in this respect to amend the Federal Clean Air Decree.

A new tax on VOCs is due to go into effect in early 2000. With regard to small sources and diffuse emissions, this tax is certainly a more appropriate means than a permit system, owing to the significantly lower transaction costs. The bubble is also appreciated as an instrument, and the major industrial establishments will no doubt continue to rely on it. On the other hand, it is not very likely that credit trading between firms will develop significantly in coming years.

Project to introduce a system of emission trading in the Canton of Valais

In a report addressed to the cantonal authorities, the Air Forum⁶ proposed a trading system for pollutants enabling a firm to postpone or even forego an abatement effort, if a firm from the same area reduced its emissions to below the legal norms (Air Forum, 1994b). This would make it possible to attain the air protection goals at lower cost. An exploratory study on the establishment of a market for tradable permits in the Chablais – an area covering 130 sq. km. on the borders of the Cantons of Vaud and the Valais – demonstrated the feasibility of such a project and the reduction in abatement costs resulting from emission trading (Stritt, 1997a, b; Stritt and Jeanrenaud, 1992).

A NO_x trading market in the Chablais

With the support of the Swiss National Fund for Scientific Research,⁷ the University of Neuchâtel set up an exploratory study on the establishment of a market for tradable permits for NO_x in the Chablais. This region is situated in the Rhône plain on the territory of the Cantons of Vaud and Valais. Surrounded by mountains, it exchanges few pollutants with neighbouring regions. From the point of view of its atmospheric properties, the region is well-suited to the introduction of a system of tradable permits (Stritt, 1997a). The purpose of the study was to examine the feasibility of the project, highlight the practical implementation-related problems and estimate the reduction in abatement costs resulting from such trading.

When the study was conducted in 1994, the air quality standards for NO_x were not being met. Average annual concentrations measured in the region ($40 \mu\text{g}/\text{m}^3$) exceeded the threshold tolerated ($30 \mu\text{g}/\text{m}^3$), while the ceiling not to be exceeded more than once yearly ($80 \mu\text{g}/\text{m}^3$) was not respected in city centres.⁸ A major reduction in VOC and NO_x emissions (around 70 per cent) was still needed to attain the ozone emission standard.

The proposed system is similar to the one introduced in Southern California in the early 1990s (SCAQMD, 1991a, b and 1993). The market would be open to the 16 firms in the area emitting more than one ton of NO_x per year. At the start of each period, the regulator would set the quantity of permits

compatible with the clean air goal. Companies starting up new facilities would have to purchase permits on the market. In this way, the Cantonal Environmental Agency would control the total quantity of pollutants emitted by the industrial plants. To calculate the right of each firm to receive permits (initial allocation), the concentrations authorised by the Clean Air Decree (OPair, 1992)⁹ would be converted into tons. In order to avoid an increase in emissions due to a padding effect, the quantities obtained for each establishment would be reduced by a certain percentage. Permits banked for future use would retain their full value (Stritt, 1997). The general conditions for the creation of a tradable permit market (characteristics of the pollutant, air quality goal clearly defined by the federal legislation, existence of significant differences in marginal abatement costs, simple and transparent operating rules) are clearly met on the whole. Given the size of the region and the small number of participants, it is clear that we should not expect a real permit market to develop, but rather trading on the basis of bilateral negotiations.

The efficiency gain depends on the differences between marginal abatement costs for the various sources. A 1992 study highlighted significant differences in abatement costs for cleaning up NO_x and VOCs in Switzerland (Stritt and Jeanrenaud, 1992). In the industrial establishments of the Chablais, marginal costs for reducing NO_x emissions range from SF 11 to over SF 130 000 per ton. From 1996 to 2000, planned abatement measures in the 16 industrial establishments will make it possible to reduce NO_x emissions by 450 tons. A 1:1 trading ratio is assumed, a permit market would make it possible to reduce abatement costs by 60 per cent given the transaction costs. As for permit prices, these would be in the neighbourhood of SF 3 000 per ton (Stritt, 1997).

The study findings aroused the interest of both the firms and the environmental protection agencies of both Cantons. The Government's desire to require strict compliance with the OPair standards, even if offsets cover facilities located near each other, precludes for the time being the creation of a market for tradable permits in the Chablais. Unlike the Basle case, the Cantons of Vaud and Valais have not set emission standards, which would have given the system a certain amount of flexibility. As a result, the idea of introducing a permit market in the Chablais has not been followed up.

Regional emission clubs

In a report addressed to the cantonal authorities of Valais, the Air Forum proposed setting up an emission trading system (Air Forum, 1994b). The aim would be to create "regional emission clubs" offering members the possibility of offsetting overshoot emission standards in certain sources with more stringent abatement measures in other facilities in the same region.

The Cantonal Environmental Agency has taken a favourable view of setting up a system for trading pollutants in Valais. To ensure that the proposed solution is in accordance with the spirit of the Clean Air Act, it gives a restrictive interpretation of the notion of emission trading. Namely:

- exchanges of pollutants should enable firms to "buy" an abatement postponement (without however giving up completely on such measures) by making a deal with a firm investing more in abatement than required by the Act;
- the definition of regions within which emission trading could take place should be coherent and take due account of the geographical configuration of Valais and meteorological compartments;
- companies subjected to prevention-oriented abatement measures (OPair, Article 8) and those subject to more stringent limits (Article 32) due to their location in an area with unsatisfactory air quality (Environmental Agency of the Canton of Valais) could participate in the trading.

For the cantonal regulatory authority, emission clubs constitute first of all a means of enabling heavily polluting industries to reduce their emissions in an economically-efficient way.

The industry supports the idea of setting up emission clubs, and the persons consulted generally give a broader interpretation to trading. This should authorise firms to not only postpone, but also to replace, an abatement operation. Regional clubs – a sort of "bubble" covering all sources in a given region – would make for a more effective clean air policy than the one currently being pursued. Thought was given to a NO_x trade between a major industrial establishment and the incineration plant for household refuse of Upper Valais. The incineration plant would have made a larger investment than the one

required by law and the industrial firm could have foregone an extremely costly abatement operation or possibly obtained a postponement for its implementation. Owing to the very strict interpretation of the OPair by the Government and its refusal to accept offsets between two sources (even those situated near each other), it has not been possible to follow up on the idea of setting up regional emission clubs – an idea which is on the back burner today.

Conclusion

Analysis of the obstacles to the implementation of tradable emission systems in Switzerland, as well as an examination of the difficulties associated with emission trading in the Basle region, both suggest several useful conclusions for the future. These conclusions relate in particular to the way in which the new instrument is being perceived by businesses; to the interest in permit trading in other regions following the Basle experience; to the poor acceptance of external trading; and to the relationship between emission trading and a stricter use of the “subsidiarity principle” in environment policy.

Increased interest in emission trading

Enterprises are demonstrating increased interest in emission trading. Several environmental managers in large firms would like to see existing emission control systems replaced by systems in which environmental objectives for each plant are expressed in terms of quantities. Such a change would in fact generalise the idea of “bubbles”.

Increasing awareness of abatement costs

A system allowing internal trading is perceived by business as being very similar to the present regulatory regime, while still allowing significant reductions in abatement costs. Today, enterprises are much more sensitive to the cost of abatement measures than they were ten years ago, when a large programme of air pollution controls was launched (OPair86). All firms now recognise that measures to control air pollution are necessary for the protection of public health and the natural environment, but they are also aware of the importance of the total cost of investments in that abatement effort. Since the marginal costs of abatement are typically higher in those firms which have already undertaken a significant clean-up effort, the cost of compliance with legislation seems disproportionate. Firms no longer hesitate to express their opposition to the strengthening of environmental standards.

Demonstration effect

Even if the Basle experience has not been as successful as anticipated in terms of the number of exchanges, it has had a demonstration effect, contributing to a better understanding of emission trading systems, not only in the Basle region, but in other areas of the country as well. The idea of creating “emission clubs” in the Valais region would probably never have been suggested if the two Basle Cantons had not taken the first step. In the Basle region, firms continue to believe in the utility of emission trading, and, more particularly, in the efficiency of bubbles. These firms would however like to benefit from more flexibility in the achievement of emission standards imposed by the Federal Government.

Social acceptability

Permit exchanges which give rise to a payment between firms remain poorly accepted. This basic rejection of market transactions in natural assets rests on moral foundations. In this view, clean air (and indirectly, public health) is a public good, which should remain outside the market-place. The idea that a firm might discharge its responsibility to protect the environment simply by making a financial payment is not well accepted by a large part of the population. This is certainly one of the reasons why there has not yet been any demand for emission credits in the Basle market.

Subsidiarity

All initiatives leading toward the introduction of emission trading come from the Cantons. This reflects the fact that the Confederation has always been rather reserved concerning the prospects for permit trading, preferring tax incentives instead, since the latter seem more easily adaptable to the present structure of emissions in Switzerland. In practice, however, the Cantons have too little flexibility to introduce a system of tradable permits, or any other new instrument for that matter (conventions or contracts, for example). The idea of a stricter application of the subsidiarity principle for pollutants with a local impact (*i.e.* where the Federal Government would fix principles and strategies, and where the Cantons would be free to choose the means) therefore merits further examination.

NOTES

1. ECU 6 215 to ECU 18 645 (SF 100 = ECU 62.15).
2. In January 1994, a committee of the Federal Parliament lodged a request calling on the Federal Council to examine the possibility of allowing Cantons or groups of Cantons to practise a more efficient environmental policy through emission trading, by accepting flexibility in the OPair standards. The Federal Council turned this request down. Accepting emission trading between two sources near each other would require an amendment to the Federal Clean Air Decree (Opair). The Federal law would have to be amended for emission trading between more distant sources. Moreover, the Federal Government gave several reasons why it was not in favour of emission trading: excessively high transaction costs; risk of having to grant permits without abatement measures having been taken; need to install instruments for measuring emissions on an ongoing basis; etc. The Federal Bureau of Environment, Forests and the Landscape (OFEFP) feels that the real reason for the lack of success of the Basle experiment was not the rigidity of the Federal standards. In its view, the market is too narrow and the transaction costs too high to operate properly.
3. Will be levied on oil containing more than 0.1 per cent of sulphur.
4. From a technical point of view, one could imagine creating a certificate market for major sources (> 3 tons/year, for example) and subjecting the smallest sources to a tax. Companies liable to the tax would no doubt find having to pay duties on residual emissions unfair whereas companies participating in the certificate market would be exempted. Yet the two instruments can coexist (the task force which examined the options allowing England to fulfil its commitments deriving from the Kyoto Protocol is to propose a combination of taxes and permit-trading – see *The Economist*, June 27th 1998).
5. It is likely that the answers obtained by Bürgenmeier *et al.* were skewed owing to the hypothetical nature of the tax.
6. The Air Forum (Forum de l'air) is a body set up by the authorities of the Canton of Valais which brings together representatives of all milieus concerned by air protection (major industrial firms, SMEs, transport companies, professional associations, the Cantonal Environmental Agency, communes and environmental protection groups). Its main task has been to make proposals with a view to drawing up the Cantonal plan for steps making it possible to attain the air quality goals set out in Federal legislation.
7. Priority programme "Environment" of the Swiss National Fund for Scientific Research, project No. 5001-35281.
8. Today, the average annual concentration of NO_x is less than the OPair emission standards and the short-term threshold is only very rarely exceeded (Environmental Protection Service of the Canton of Valais).
9. The calculation takes due consideration of the hourly volume of effluents, the concentration authorised at emission and the daily and annual operating times of a given facility.

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Chapter IX

THE CASE OF THE UK: SULPHUR PERMIT QUOTA SWITCHING

by

Jennifer Steedman¹

Department of the Environment, Transport and the Regions
London, United Kingdom

Introduction

In 1992, the UK Department of the Environment² commissioned a study which investigated the potential use of emission charges and tradable permits to control sulphur dioxide (SO₂) emissions in the UK, for pre-1983 plant covered by the Large Combustion Plant Directive national emissions totals. The study (DOE, 1992) identified tradable permits as the most promising economic instrument and estimated that the resource saving potential of a permit system would be in the region of £60-80 million per annum. With the cost efficiency argument proven, the government formally proposed the use of emissions trading – or “sulphur permit quota-switching” as it was referred to – as a means of meeting international agreements for SO₂ more cost-effectively. A wide range of views were sought on the scope and design of the permit scheme and, in May 1994, the DOE issued the first proposal, setting out the details of the proposed sulphur permit quota-switching system. Protracted discussions ensued and various revisions were made to the original proposal. In June 1996, however, the UK Government concluded that “there would be no scope for a quota transfer or ‘trading’ scheme” in their Strategy to Reduce National Emissions of Sulphur Dioxide.³

The purpose of this paper is to explain why the plan for sulphur quota-switching was abandoned in the UK in 1996, and hence, to identify lessons for the wider implementation of tradable permits in Europe. While the principle reason for the final abandonment of the scheme was that a rapid decline in coal burn had led to ample emissions quotas, the design of the scheme exposed a number of practical difficulties that would need to be addressed if the scheme were to be resurrected:

- Regulatory:
 - a complex and uncertain regulatory environment;
 - interactions with the existing regulatory regime.
- Contextual:
 - changes in composition of UK energy use away from coal and oil;
 - lack of support from industry and environmental groups.
- Design:
 - allocation of quotas.

It is important to stress at the outset that the regulatory problems encountered derived from the European, rather than the national, regulatory framework. This implies that:

- the UK experience may not be a one-off case;
- all other Member states are potentially susceptible to some of the problems which beset the UK;
- the potential barrier generated by EU legislation to emissions trading warrants further attention.

The regulatory problems may also help to explain why emission taxes are a much more common market based instrument in Europe than in the US.

Structure of Paper

The next section of the paper sets out the main European regulations pertaining to sulphur and explains how these create conditions which are not conducive to the implementation of trading regimes. This is followed by a description of the national regulatory controls on SO₂ and the quite dramatic developments taking place in the energy market at the time sulphur quota-switching was proposed. The details of the sulphur quota-switching proposal are then examined, including a discussion of the problems which arose and the developments which led to the eventual abandonment of the proposal. The paper then summarises the main factors which contributed to the failure of the proposed trading scheme, and the lessons offered by the UK experience.

European regulatory controls on sulphur dioxide

Given the transboundary nature of SO₂, the UK Government works closely with international partners, principally the European Union and the United Nations Economic Commission for Europe (UNECE), to secure reductions in SO₂ emissions. The European regulatory framework pertaining to SO₂ is complex and consists of various international agreements and regulatory controls. The most important for the purposes of this analysis are:

- EC Large Combustion Plant Directive (88/609/EEC);
- UNECE Second Sulphur protocol;
- the Air Framework Directive (AFD) and Integrated Pollution Prevention and Control Directive (IPPC).

EC Large Combustion Plant Directive (LCPD)

The LCPD was agreed in 1988 and sets emission limits for SO₂ and NO_x emissions from new combustion plant and a programme of staged reductions in these emissions from existing (*i.e.* pre-1987) plants. The Directive requires the UK to reduce emissions of SO₂ from existing plants by 20 per cent by 1993, 40 per cent by 1998 and 60 per cent by 2003, compared to 1980 levels.

At the time the quota-switching proposal was being considered, it was known that the Commission was expected to put forward proposals for the revision of new plant emission limits and new national ceilings on emissions from existing plant, by July 1995. The first informal proposals were discussed in March 1996, but targets are still being negotiated.

UNECE Second Sulphur Protocol

In June 1994, the UK signed the second UNECE Sulphur Protocol under the Convention on Long Range Transboundary Air Pollution (CLRTAP). This Protocol represents a more scientific approach to the problem of controlling SO₂ emissions and is based on the concept of critical loads. The Protocol requires each country to abate by differing amounts depending on the quantity of emissions and the impact which these emissions have on the environment across Europe. Under the Protocol, the UK agreed to reduce total SO₂ emissions by 50 per cent by the year 2000, 70 per cent by 2005 and 80 per cent by 2010, compared with 1980 levels. The UNECE targets are significantly deeper than the reductions required under the LCPD and cover all sulphur sources, not just large combustion plants.

It is important to note that, although there are significant differences between the LCPD and the UNECE targets in terms of levels, timing and coverage, both represent top-down approaches to pollution control in that they define aggregate emission reduction targets. The UNECE Protocol is “soft law” that can be implemented by administrative action. The LCPD is “hard law” that has to appear in national legislation.

The Air Framework Directive (AFD) and Integrated Pollution Prevention and Control (IPPC)

The European AFD on emissions from industrial plants was agreed in 1984 and hence preceded both the LCPD and the UNECE second sulphur protocol agreements. The Directive did not specify emission reductions for individual plants or require aggregate reductions in emissions, but rather, was technology-based and required (mainly industrial) operators to use the “Best Available Technology Not Entailing Excessive Costs” (BATNEEC). This concept now plays an integral part of the regulation of SO₂ emissions across Europe, including the UK.

The Directive on Integrated Pollution Prevention and Control (IPPC) replaced the AFD in 1996⁴ and to a large extent was modelled on the UK system of Integrated Pollution control (IPC). IPPC represents an integrated approach to pollution control, whereby releases to air, water and land are regulated together. Industrial installations which come under IPPC can only operate with a permit. The permits specify limit values which are to be based on Best Available Technology (BAT)⁵ and the limit values take account of the technical characteristics of the source, the geographical location and the local environment conditions. New plants are required to apply “new plant” IPPC BAT straight away, while existing plants (*i.e.* pre-1987) have until 2007 before they have to meet IPPC BAT appropriate to their age and condition.

Unlike the LCPD and the Second Sulphur Protocol, IPPC represents a bottom-up approach to pollution control. The focus is on the environmental performance of individual installations and each must be brought up to BAT standards, without reference to an aggregate emission target.

Additional European regulations

In addition:

- there were EC directives relating to the sulphur content of certain liquid fuels which placed mandatory limits on the sulphur content of industrial gas oil and diesel;
- work was underway to develop an EU ambient air quality standard for sulphur dioxide;
- the EU were developing a strategy for tackling acidification and this was expected to be complete by mid 1997.

It is clear, therefore, that the European system for regulating SO₂:

- is complex and uncertain – target limits for existing regulations are subject to frequent revision and, as more is learnt about the sources, characteristics and effects of SO₂, so new Directives are introduced to tackle specific aspects of the problem in a more direct manner;
- incorporates fundamentally different regulatory methods which limit the opportunities for trading systems.

These characteristics raise issues for the implementation of tradable permit regimes and are discussed in more detail below.

A complex and uncertain regulatory environment

The US experience with trading systems represents the best real-life approximation of the theoretical trading system; it demonstrates that, *under the right conditions*, trading systems can achieve specified reductions in pollution in a more cost-effective manner than traditional US-style command and control (CAC) policies; and, through trial and error, has gone some way into establishing what the “right conditions” are (*i.e.* those which are sufficient) for tradable permit regimes to operate effectively.

Under the initial emissions trading programme, for example, it became clear that uncertainty of any kind (*i.e.* security of permits, unstable conditions) inhibited trades. In the original US scheme, for example, participants had no guarantee that, by acquiring an emission credit, they were acquiring anything of value because if the state failed to achieve its air quality target, these credits could be confiscated. Subsequent analysis of the programme showed that this uncertainty over the security of property rights was one of a number of key factors which had significantly reduced the costs saving potential of the scheme.

In subsequent US trading schemes, therefore, deliberate steps were taken to minimise uncertainty over the security of permits by defining allowances as “limited use rights” which could only be terminated

by a further revision of the Clean Air Act and not by an administrative decision on the part of the Environmental Protection Agency (USEPA). Such improvements are evident in the SO₂ trading programme for electricity producers, introduced in combination with traditional control methods to achieve the objectives of the acid rain programme. The framework and objectives of this programme are set out clearly and participants have the necessary assurance that these will not change, at least until 2010.

Given the lessons of the US experience, the ever-evolving nature of the existing European regulatory environment sends a warning signals to anyone planning to introduce a trading scheme in a European state. It is clear that if SO₂ targets are the subject of negotiation and repeated revision, any trading system implemented to achieve the targets would be subject to considerable future uncertainty.

Interaction with existing European regulatory regimes

Another obstacle to implementing trading schemes in Europe reflects the fact that different non-trading regulatory regimes are used to control SO₂. If one has a bottom-up and top-down regulatory regime operating at the same time, then there are three difficulties. First, the two systems can only operate side-by-side when the annual emissions of a plant are below the level implied by emissions at full load factor at the technological standard associated with IPPC for plant of that vintage. If annual total emissions are above BAT, the BAT technology is clearly not being applied. If annual emissions are equal to BAT, the installation is controlling emissions to the level required, and hence, there is nothing left to trade. It is only when annual emissions are below the level required by full load BAT then there is some potential for trade because the BAT technology is being applied, but operators have surplus emissions left to trade.

Second, even if circumstances are such that trading can take place, trading (and hence the costs savings) are clearly limited as trading is clearly only allowed up to the point at which the fixed emission ceiling is reached. After that point, in order to comply with bottom-up regulations, firms have no choice but to cut emissions.

Klassen (1998) has emphasised the need for clarification of the conditions under which countries can use emission trading schemes and recognises the argument that uncertainty on the compatibility of IPPC is a major barrier to the introduction of emissions trading. He argues, however, that:

“Member states can always set more stringent measures than required under IPPC and must even do so if this is needed to meet environmental quality standards set out in EU legislation. These additional measures (going beyond BAT defined in the IPPC) could take the form of emission trading. In line with the BAT requirements of the IPPC Directive, such more stringent measures could imply one of the following: limitation or modification of economic activities of installations concerned; utilisation of techniques considered not to be available under economically and technically viable conditions; use of techniques for which costs outweigh advantages” [Klassen (1998), p. 23].

Klassen concludes, therefore, that “national emission trading schemes can be combined with the IPPC Directive, provided that IPPC requirements are fulfilled.” Sorrel (1998) has subsequently argued⁶ that, “this [argument] would appear to confine trading to marginal adjustments for meeting a very stringent emission target”. It should be noted, however, that while, in theory, the combination of IPPC and trading does seem to significantly reduce the potential role to be played by trading schemes, in practice, there are a number of factors which may mean that a substantial gap still exists for trading.

For example, in the UK, although inspectors give advice on how best to minimise emissions from existing processes, replacement of the process by a more emissions-friendly process is not expected immediately but as part of a long-term investment plan. Further, it may be that the regulator's assessment of existing technology is a number of years behind industries assessment. In practice, these factors are likely to vary between industries and sectors, but they suggest that there may be a significant gap for industry to deliver reductions in emissions in addition to those delivered through regulation. If this is the case, the potential still exists for trading schemes to provide the correct incentives to industry to deliver these additional savings. In any case, any additional requirements on plant in their authorisations in order to meet a national total requirement can always be readjusted between plant by mutual agreement with the regulator.

National regulatory controls on sulphur dioxide

The Directives and agreements described above require that Member states introduce legislation, strategies and administrative procedures for their implementation. The main legislative instruments which give effect to the European requirements in the UK are:

- UK system of Integrated Pollution Control (Part 2, EPA, 1990);
- UK Large Combustion Plant National Plan (Part 1, EPA, 1990).

Integrated Pollution Control (IPC)

Under IPC, authorisations are required from the pollution regulator⁷ (pre-1996, this was Her Majesty's Inspectorate of Pollution, HMIP) before certain prescribed processes can operate (including those which fall under the National Plan). Authorisations are decided on a site-by-site basis and contain a number of legally-binding commitments, one of which is to ensure that BATNEEC is used to prevent the release of prescribed substances (or, where that is not practicable, to reduce releases to a minimum).

It is important to note that the original authorisations required HMIP to carry out a review of IPC authorisations for power stations. The review for the English and Welsh Electricity Supply Industry (ESI) started in 1994, and was completed in March 1996. The review of authorisations for power stations in Scotland was set on a later timescale and was completed at the end of 1997. The review of Northern Ireland power stations has only just been completed. The revised IPC authorisations obliged the generators to make significant reductions in their levels of SO₂ emissions and played an important role in making the quota-switching proposal redundant.

National Plan

The LCPD is given effect in the UK through a national programme of emission reductions. The National Plan sets out separate annual quotas (in kilotonnes of SO₂) for existing large combustion plant in England, Wales, Scotland and Northern Ireland and in the major industry sectors (*i.e.* electricity supply industry, refineries and other industry). Annual quotas have been set for each sector for each year to 2003 and annual quotas for some representative years are presented in Table 1.

Even before the sulphur quotas switching scheme was proposed, the National Plan incorporated some of the basic elements of a trading scheme. The quota targets imposed on the electricity generators, for example, allowed them to effectively operate as constrained emission bubbles, allowing emission limits in various plants to vary provided that the total annual emissions from each ESI were below the annual quota allocation. The English and Welsh ESI were allowed to operate as two individual companies – National Power and Power Gen – and each company was allowed to operate as a separate bubble moving quotas between their plants (but not companies). Intra-firm trading in the ESI has not presented any problems. The situation was more rigid in the refinery and other industry sectors of the National Plan where quotas were allocated on a plant-by-plant-basis.

The National Plan was implemented by the regulators. DOE's intention was that authorisations would contain conditions which enforced the terms of the National Plan, perhaps including allocations for permitted levels of SO₂ emissions in tonnes, reducing on a year-on-year basis in line with the National Plan requirements. In theory, quotas were supposed to be distinct from the BATNEEC limits. In practice, the distinction was only recognised in the electricity supply sectors. In the case of the English and Welsh ESI, for example, a separate BATNEEC limit (Limit A) and quota allocation (Limit B) was set for each power station. Limit A defined the maximum permitted emissions that could be sustained without harming the local environment and hence could not be exceeded at any time. Limit B, however (which, in general, was lower than the BATNEEC limit) could be exceeded at individual plants within the same company, provided that neither the BATNEEC level, or, the company total under the National Plan, were exceeded. The allocation principle was "informed grandfathering". The LCPD quotas did not cover new plants, so new entrant issues did not arise.

Table 1. **SO₂ Emission Quotas for Existing Large Combustion Plant (LCP) in Accordance with the LCPD Directive**

In kilotonnes						
ector	1980	1991	1994	1998	2000	2003
ESI						
National Power		1 595	1 373	982	857	660
PowerGen ¹		1 085	969	669	583	450
England and Wales	2 776	2 680	2 342	1 651	1 440	1 110
Scotland	142	109	102	99	77	57
Northern Ireland	88	92	75	53	46	35
Total ESI	3 006	2 881	2 519	1 803	1 563	1 202
Refineries						
England and Wales	218	86	85	82	80	78
Scotland	50	14	14	13	13	12
Total Refineries	268	100	99	95	93	90
Other Industry						
England and Wales	543	273	233	201	177	140
Scotland	78	39	34	29	25	20
Total Industry	621	312	267	230	202	160
Total	3 895	3 293	2 885	2 128	1 858	1 452

¹ In 1996 Power Gen was divided into two companies – Power Gen and Eastern Electricity – and the former switched some of its quota allocation to the latter.
 ource: DOE (1990).

In the majority of authorisations no distinction was made, and a single annual fixed limit was imposed instead, which typically reflected the BATNEEC emission rate limit. In practice, therefore, the limits were fixed; they could not be reduced on an annual basis as was required by the National Plan requirements. In addition, since these limits were set on a plant-by-plant basis, there was no opportunity for multi-plant companies to operate under a bubble.

Developments in the energy markets

In the late 1980s and early 1990s, the UK embarked on a process of privatising and liberalising its gas and electricity markets. The accompanying developments (*i.e.* increased competition, increases in the supply of gas and hence major reductions in the price of gas) generated a marked shift in the composition of energy use away from coal and oil and towards gas. The so-called “dash for gas” had a dramatic impact on the level of SO₂ emissions generated in the UK, and played a central role in creating conditions which were not conducive to the introduction of the proposed trading scheme.

This change in fuel use away from coal and oil and towards gas was particularly marked in the electricity generation sector; this is demonstrated in Table 2 below.

It is clear from Table 2 that between 1989 and 1996, the share of gas-generated electricity increased from approximately 1 per cent to 22 per cent of total electricity generation. This, in turn, reflected the quite dramatic growth in Gas fired Combined Cycle Gas Turbine capacity, induced and encouraged by low gas prices, low capital costs and minimal siting difficulties. It is clear from Table 2, that the increase in gas displaced coal and oil generated electricity which fell by over 20 per cent and almost 5 per cent, respectively, over the same period.

The quite dramatic shift in the composition of fuel input to electricity generation triggered an equally dramatic decline in sulphur dioxide emissions generated in the UK. Between 1989 and 1996, for example, emissions from power stations declined by 50 per cent.

Table 2. **Thermal Input to Electricity Generation 1989-96**

	% of total				
fuel	1989	1990	1992	1994	1996
coal	64.6	65.3	61.3	50.3	43
oil	9.4	11	10.5	5.5	4.6
gas	0.7	0.7	2	13.4	21.5
nuclear	23.6	21.3	24.1	28.8	28.9
hydro	0.5	0.6	0.6	0.6	0.4
other	1.2	1.1	1.4	1.4	1.7
total	100	100	100	100	100

Source: Sorell (1998).

It is important to note that the full extent and implication of these developments were yet to be recognised in 1994 when the sulphur permit quota proposal was introduced.

Impact on National Plan

Since the National Plan and hence the allocation of quotas was drawn up before the trends in the energy markets revealed themselves, the “dash for gas” effectively left a significant percentage surplus in each sector. Table 3 shows the position in 1993.

Table 3 shows that total emissions from the large combustion plants in 1993 were almost one-quarter below the target level set out in the National Plan. Since, the English and Welsh ESI accounted for the dominant source of UK emissions in 1990 and hence received the largest allocation of quotas, they were responsible for the largest proportion of the surplus (75 per cent).

Although most sectors were operating with a substantial surplus in 1993, some sectors were being constrained or expected to be constrained in the future. For example:

- the ESI in Northern Ireland was operating with only a 2.5 per cent surplus and thus additional measures would be required to keep within its quota;

Table 3. **UK LCP Sulphur Dioxide Emissions and Quota Allocations in 1993**

sector	Capacity (GWth)	Emission (ktonnes)	Quota (ktonnes)	Surplus (ktonnes)	Surplus (per cent)
ESI					
National Power	166.3	1 116.4	1 497	380.6	25.4
PowerGen	44.2	841.8	1 019	177.2	17.4
England and Wales	210.5	1 958.2	2 516	557.8	22.2
Scotland	13.8	52.8	104	51.2	49.3
Northern Ireland	6.4	78	80	2	2.5
Total ESI	230.7	2 089	2 700	609.4	22.6
Other Industry					
England and Wales		154.1	241	86.9	36.1
Scotland		5.9	35	29.1	83
Total Refineries	15.6	160.1	276	115.9	42
Refineries					
England and Wales	7.8	67.7	86	18.3	21.3
Scotland	1.7	11.8	14	2.2	15.9
Total Industry	9.5	79.5	100	20.5	20.5
Total	255.7	2 328.5	3 076	747.5	24.3

Source: Sorell (1998).

- the ESI in Scotland expected to be constrained in the future due to a combination of factors (cessation of gas supplies at its Peterhead plant, increase in electricity exports to England and Northern Ireland).
- industrial operators expected to be constrained in the next few years and multi-site companies complained that they could not enjoy the same flexibility as the ESI by operating as an emissions bubble.

It was clear, therefore, that in light of changes in the energy market, the rigid sectoral and regional targets set out in the National Plan were no longer appropriate. A number of companies outside the ESI made representations requesting a company bubble to enjoy the same advantages as the electricity generators.

The UK quota-switching proposal, problems and subsequent developments

The original proposal

It was in this context that the sulphur quota-switching proposal was introduced. It is important to note that while the UK had more than fully complied with the 1993 LCPD reduction targets for existing plants, the Government believed that a more stringent strategy would be required in the later years of the LCPD to secure the objectives of the Second Sulphur Protocol. In addition to increasing the flexibility of industrial operators, therefore, a significant benefit expected of introducing the scheme was to allow the market to facilitate the distribution of quotas in future years when the total quotas available for allocation would have declined considerably.

In May 1994, the first draft of the quota-switching proposal was prepared which proposed that:

- quota-switching be allowed for all existing plants in the National Plan, with effect from 1 May 1995;
- the quota-switching scheme would operate across and between all three sectors of the National Plan, including between individual plants;
- quota-switching would be based on the existing pattern of quota allocation, that is, those set out in the National Plan (mainly because the “Government is not persuaded that it would be fair or reasonable to make reallocations at relatively short notice in the course of the current scheme”);
- as under the existing system, plants would continue to require an IPC authorisation from HMIP which would specify an annual quota allocation to 2003 as well as an annual BATNEEC limit. Unless switches were agreed, the tighter of the two limits would be used to set the year end limits on emissions. If, for example, the BATNEEC figure was tighter than the National Plan quota, then (although a plant’s emissions were not allowed to exceed the BATNEEC figure) surplus permits could be offered up for switching either outside the company or be transferred to another site within the same company. If the national quota allocation represented a tighter limit than the BATNEEC limit, however, companies would exceed the National Plan quota, provided they could buy in sufficient additional permits up to the BATNEEC level. In either case, the BATNEEC limit would represent a fixed emission ceiling which could not be exceeded in any year;
- a quota would represent an allowance to emit one tonne of SO₂ and each tonne would be allocated specifically for a given year. This, in turn, meant that the banking of permits would not be allowed, since it was believed that this could jeopardise the attainment of the LCPD targets in future years;
- quota-switching within and between the National Plan sectors would be allowed across geographical boundaries.⁸ So, for example, a refinery in England and Wales might be able to switch quotas with other industry plant in Scotland or a power station in Northern Ireland. The overall constraint would be that total SO₂ emissions would equal the sum of existing national totals in the LCPD and that individual plants did not exceed their BAT limits;
- a record of individual plant quotas would be held centrally by the territorial regulators and would be publicly available. Proposals for switches would not require prior approval but notification of switches would be placed in the IPC register held by the regulator and local authorities.

Problems and subsequent proposals

Shortly after the first draft of the quota-switching strategy was issued, problems started to appear which made it impossible to proceed with the original proposal. Revised projections of likely sulphur emissions allowed the UK to conclude negotiations for the UNECE Second Sulphur Protocol and agreed to more stringent targets. It became apparent at this stage that there was a gap between the National Plan Targets and UNECE targets. In order to achieve compliance with the latter, the total number of permits in circulation would have to be reduced. Further, it was recognised that the initial allocation of emissions on authorisations was inappropriate, given developments in the energy markets: a disproportionate share of surplus sulphur was held by the English and Welsh ESI, and hence, the potential existed for these generators to manipulate the permit market.

A revised draft proposal was therefore issued in July 1994. In order to solve the apparent inconsistency between the National Plan and UNECE targets, it was proposed that although the scheme would operate for the full duration of the SO₂ reduction plan in the LCPD (*i.e.* until 2003), quotas would only be allocated for the first 3 years (*i.e.* 1996, 1997 and 1998). This reflected the expectation that in 1999 and beyond, the total supply of permits would have to be reduced if they were to have a role in meeting the requirements of the second sulphur protocol and hence, there would be no point in allocating permits if these would later have to be withdrawn or go surplus.

Although the need for reallocation was recognised, the reason why it has not been attempted in the first place was because reallocation implied large windfalls. It was therefore expected that it would take a long time to reach agreement, time which the government did not have as it was operating under a tight time constraint (the scheme had to be introduced by 1 May 1995 to accommodate any outstanding appeals, and hence, allow the scheme to apply from 1996 onwards). Thus, rather than reallocating quotas, it was proposed that quota-switching would only be allowed within each of the three sectors, and not between them. The quotas contained in the National Plan for the ESI's (*i.e.* National Power, Powergen, Scotland and Northern Ireland) and the refineries, therefore, were to remain in place. For other industry, a reallocation was unavoidable, to ensure that the total sum of emissions in the authorisations did not exceed the sector total under the National Plan. Therefore, it was proposed that quotas in the industry sector would be reallocated on the basis of fuel use.

These alterations did nothing to advance the quota-switching proposal. For example, although the three-year limit on quotas gave the Government a breathing space in which it could think about how to bring it into line with UNECE targets this introduced considerable uncertainty as to what would happen in 1999. Second, although inter-sectoral trading removed the refinery and other industry sector away from the market dominance of National Power and Powergen, it reduced the opportunities to trade for the other small ESIs. In effect, they could no longer purchase quotas from refineries or other industry. In addition, as the Government feared, the reallocation of quotas in the industry sector proved to be highly controversial.

A third, even more diluted, proposal was suggested, namely to exclude the ESI and the refineries from the scheme, and restrict the quota-switching to other industry. However, even this scheme was destined to fail because HMIP's BATNEEC limits, which were largely based on fuel mix at the plant at the time of the IPC application. Quotas would also be reallocated on the basis of fuel mix. The problem with this is that with quotas based on the fuel mix at the time of the IPC application, coal and oil plants had no need for additional quotas, dual-fired sites using either gas or HFO could not purchase quotas, since they would exceed their BATNEEC limit and multi-site plants could not operate as bubbles without exceeding their BATNEEC limits. In these conditions, there was no room for trading.

Subsequent developments

The most important development between the final draft proposal and the abandonment of sulphur quota-switching, was the publication of the HMIP review of IPC authorisations for power stations in England and Wales in March 1996.⁹ The targets to which the generators had agreed were surprisingly large

– a reduction in emissions in the remaining coal and oil fired stations by 70 per cent by 2001 and 85 per cent in 2005, measured from 1991 base levels – but reflected the continuing move from industrial burning of coal and oil to gas and completion of programmes to fit FGD at the English and Welsh stations.

New emission projections which took the new targets into account and made assumptions about the progressive upgrading of other large combustion plant, showed that the UNECE targets for 2000 and 2005 could be met by the fullest operation of the BATNEEC principle, without the need for additional measures. In the Strategy to Reduce National Emissions of Sulphur Dioxide, the Government therefore concluded:

“On this basis, the Government sees no need at present to take any new measures [including the introduction of a switching regime] specifically to meet the UK’s commitments under the Protocol beyond the close monitoring of emissions and the intention to review the position in three years time.”

Since the projections also indicated that the targets in the National Plan will be met by a significant margin with a substantial surplus of quotas forecast in each year to 2003, the Government also concluded:

“Whilst it is necessary to retain the overall framework of the Plan to provide a national programme for meeting the LCPD’s requirements, the sectoral and geographical constraints in the Plan can be relaxed in order to provide greater flexibility”.

It was proposed, therefore, that the national plan be transformed into a series of aggregate annual national emission targets and that, in the future, regulators would be required to allocate quotas to individual plants on the basis of BATNEEC considerations. In effect, therefore, this forced the quota allocations to be consistent with regulation under IPC. Under these circumstances, no plant would be able to utilise more quota than it had been allocated without breaching BATNEEC limits. Hence, there was no scope for a quota-switching scheme. While HMIP did allow flexibility into the revised authorisations for the electricity supply industry which allowed generators to switch emissions between stations, the same flexibility was evident in 1992. Thus, “emissions trading” was back to where it started.

Summary of obstacles

It is clear from the preceding analysis that the quota-switching proposal was withdrawn in the UK for five reasons:

- changes in composition of UK energy use limited the impact of international agreements on individual plants;
- interactions with the existing regulatory regime;
- a complex and uncertain regulatory environment;
- problems with agreeing allocation of quotas (encouraged by existing regulations);
- lack of enthusiastic support by industry and hostility by the environmental movement.

It is clear that the surplus of sulphur removed the need to use a trading system to implement international agreements. But even if this had not happened, a number of other factors presented problems for the scheme.

Changes in the composition of energy use

At the simplest level, the quota-switching proposal failed due to unexpected developments in the energy markets which allowed the UK to meet its targets without any direct environmental policies, including those based on tradable permits. In effect, therefore, quota-switching failed because it was no longer needed. However, this is not the whole story, because it seems likely that the proposal would have failed, even in the absence of the energy changes.

Interactions with the existing regulatory environment

It is clear that the existing technology-based system of regulation limits the scope for trading as trading can only occur in the gap between the quota allocation and the BATNEEC ceiling.

A complex and uncertain regulatory environment

Even if a loose interpretation of BATNEEC had been made in relation to all sectors in the National Plan, and hence switching had occurred in each sector, the scheme would only have operated for a short period of time (*i.e.* the period before the revised allocations were decided when the BATNEEC limit remained relatively relaxed). After the review of IPC authorisations in 1996, the BATNEEC limits for the England and Wales ESI were set at levels which were tighter than those of the UNECE Second Sulphur Protocol, leaving no room for trading. The constantly evolving nature of BATNEEC and indeed the repeated revisions made to other European and International agreements therefore represent a serious barrier to trading as they generate an uncertain environment, one which is not conducive to future trading.

Inappropriate allocation of quotas

Much of the protracted debate which occurred between DOE and industry over the period 1992-1996 focused on the fairness, or, rather, the perceived unfairness, of the initial allocation of quotas. The UK system, therefore, suffered from one of the most common design problems associated with permit systems – that of devising an initial allocation which is both fair and protects against market power.

Lack of enthusiastic support

It is a well-established principle that a trading scheme must be strongly supported by the majority of agents involved. Because the proposed scheme was based on the initial allocation of quotas set out in the National plan, non-ESI industry was against the proposal from the outset. While it is true they were arguing for increased flexibility, they believed that this could be achieved through a redistribution of quotas, or through negotiations with their individual inspectors. Regulation in the UK has historically been technology-based, and industry could see no advantages to be gained from trading, presented the way it was on the quota allocation established in 1990 under the National Plan. There was also a lack of support from environmentalists who opposed trading on the belief that it would damage sensitive areas in the UK. This is in marked contrast to the situation in the US and the introduction of the US Acid Rain Programme which was suggested by industry, and vigorously supported by influential environmental lobbyists.

Lessons

The UK experience has important lessons for the future implementation of tradable permit regimes, both in the UK and elsewhere in Europe.

First, it demonstrates that, in practice, economic efficiency is only one (and not necessarily the most important) of a number of conditions which must be satisfied to make the introduction of tradable permit schemes successful. The regulatory context, existing conditions, design and equity issues are the real factors which will determine whether the potential cost savings can be realised.

Second, if a trading scheme is imposed on top of the existing regulatory regime, then it must be recognised that trading will be limited to the gap between the BATNEEC ceiling and the lowest possible level of emissions. The size and duration of this gap may be significant, and is likely to vary between industries, sectors and countries. However, it should be recognised that this gap is likely to close over time as BATNEEC becomes increasingly tighter and, if trading schemes are expected to operate within the gaps, steps will have to be taken to reduce the uncertainties inherent in the present system of regulation.

NOTES

1. The views expressed in this paper are those of the author, and not necessarily those of DETR.
2. The Department of the Environment (DOE) was merged with the Department of Transport in 1997, and is referred to here as the Department of the Environment, Transport and the Regions (DETR).
3. Department of the Environment (1996).
4. IPPC will not be implemented in the UK until October 1999.
5. The end part of the term “not entailing excessive cost” is absorbed into the definition of availability.
6. Sorell (1998), p. 314.
7. As from April 1, 1996, responsibility for regulating the IPC system passed from HMIP to the Environment Agency in England and Wales, and to the Scottish Environment Protection Agency (SEPA) in Scotland.
8. While a weighting/exchange rate system had been considered for switches between different parts of the UK, it was decided that this would be too complex to administer and would be unnecessary since BATNEEC limits already take local ambient conditions into account; hence, would preventing the creation of any local “hot spots”.
9. The Review of authorisations for power stations in Scotland was set on a later timescale and was not completed until the end of 1997. The review of Northern Ireland power stations was later still, and has just been completed.

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Part III

OTHER IMPLEMENTATION ISSUES

Chapter X

THE COMPATIBILITY OF TRADABLE PERMITS WITH OTHER ENVIRONMENTAL POLICY INSTRUMENTS

by

Stephen Smith

University College, London, United Kingdom

Introduction

The scope for making greater use of tradable permits in environmental policy has been an increasing theme of recent policy discussion among Member countries of the OECD, reflecting growing interest in these instruments and other market mechanisms in the domestic environmental policy-making of some Member countries. The OECD Working Party on Economic and Environmental Policy Integration (WPEEPI) has, for a long time, promoted the greater use of *market mechanisms* to achieve more cost-effective environmental protection. Likewise, in many Member countries, there has been an important shift in national policy towards the use of market mechanisms, including environmental taxes and charges, subsidies, tradable permits, and voluntary agreements. The key argument for the use of market mechanisms is now widely recognised, namely that in comparison with conventional “command-and-control” regulation, they have the potential to reduce the economic costs associated with a given level of environmental protection. In most Member countries, market mechanisms now play a role in environmental policy, and the discussion of environmental policy options routinely considers the merits of market mechanisms alongside other possible approaches to policy.

With the growing recognition amongst policy-makers that market mechanisms can have a valuable role to play in some areas of national environmental policies, attention in policy debate has increasingly come to focus on the relative merits of particular economic instruments in those policies.

- In recent years, the OECD has undertaken major studies on a number of aspects of *environmental taxation* (OECD, 1993, 1996a, 1997). Many Member countries have also undertaken national studies on this subject, and there has also been extensive work amongst academics and the wider policy community.
- There has also been considerable work on the role of *subsidies* in environmental policy. Depending on how such subsidies are implemented, they can have some of the attractions of economic instruments, or, less desirably, can reinforce some of the negative features of command-and-control regimes, notwithstanding the potential negative environmental impacts of sectoral subsidies (OECD, 1998).
- There is a growing debate, too, on the possibility that “negotiated” or “voluntary” *agreements* between governments and industry may achieve many of the desirable effects of economic instruments, in a context that is more acceptable to the private sector, and a number of major studies have recently been published or are in progress [OECD/IEA, 1996b; EEA, 1997; OECD (forthcoming, 1999)].

There is also increasing interest in the role that could possibly be played by tradable permits. Emissions trading and other tradable permit arrangements have become a key element in environmental policy in the United States, and other OECD countries are showing increasing interest in evaluating this

experience, and in comparing the merits of the US tradable permits arrangements with the approach to market mechanisms elsewhere in the OECD countries, where the emphasis has predominantly been on other instruments such as taxes, charges and negotiated agreements.

Characteristics of tradable permits schemes

Schemes for tradable permits could take a number of forms (see also Chapter II):

- The sale, perhaps by auction, of permits entitling the holder to a given level of emissions. The permits can be purchased by those polluters needing them, and may subsequently be resold to other polluters if the initial purchaser wishes to do this (for example, if circumstances change).
- The distribution of permits (either free or subject to some limited charge) to existing polluters (and perhaps others). Where the distribution of permits is made in proportion to existing emissions, such arrangements are known as “grandfathering”. Once distributed, the permits can be bought and sold, thus reallocating between polluters the total level of permitted pollution.
- Emissions trading schemes with properties very similar to tradable permits of the second type indicated above. Polluters are subject to some form of command-and-control regulation, in which emissions limits are determined for individual polluters. However, individual polluters are permitted to exceed the regulatory limits they have been assigned if they can demonstrate an arrangement for corresponding over-compliance with the regulatory limit in some other unit or location. This may be in another plant owned by the same firm, or it may be in a different firm, as a result of an explicit contract between the two firms. “Tradable permits” would not actually be defined, but the outcome of these contracts between firms, agreeing to over-compliance to offset additional pollution elsewhere, would be very similar in effect to that of a scheme involving grandfathered tradable permits.

Each of these schemes has in common two key features:

- i) *guaranteed environmental effectiveness*, since the number of permits issued (or the initial regulatory limits) places a fixed upper limit on total pollution. Unlike other market mechanisms, such as environmental taxes or charges, where the impact on emissions depends on the responsiveness of polluters to a given incentive signal (*e.g.* rate of tax), tradable permit schemes have the ability to achieve a definite pollution or abatement target. This may be desirable, especially in the case of pollution problems where excess pollution would involve very large pollution damage (*e.g.* due to large threshold effects, critical loads, etc). Also, the guaranteed attainment of pollution targets may help to persuade sceptical public opinion of the acceptability of pollution control through this form of market mechanism. On the other hand, it carries the risk for polluting firms that the abatement target may be set too high, at a level which requires inordinately-costly abatement technologies to be used. (Emission taxes, by contrast, give polluting firms the “safety valve” option of simply paying the tax when the only technologies available for abatement are very costly, and thus provide firms with some insurance against unrealistically-stringent environmental policy-making).
- ii) *a cost-minimising distribution of the burden of abatement* between different sources. Different sources of pollution (*e.g.* different factories within an industry) may face different costs of abatement for a number of reasons. For example, a number of different production technologies may be employed by different firms; abatement may be more straightforward (and less costly) with some technologies than with others. Also, abatement costs may differ as between old and newer plants, because the cost of pollution-control investments has to be amortised over the remaining lifespan of the plant, and the annual cost of a given investment is thus higher for a production facility which is nearing the end of its life than for one that is newer. Tradable permit schemes allow firms facing higher costs of abatement to do less abatement, and therefore, to continue with a higher level of pollution, than those firms for which pollution control is cheaper. They can do this either by buying more permits in the initial auction (in cases where permits are distributed in this way), or by buying permits from other firms, which need them less. Either way, the total cost of achieving the given total level of pollution should be lower than if polluters faced uniform abatement requirements – abatement is redistributed through these permit transactions to the firms able to undertake the abatement most cheaply.

The latter advantage of tradable permits is one which, in principle, tradable permits share with the other market mechanisms in environmental policy, such as pollution taxes and charges. Indeed, except where the policy-maker faces great uncertainty about abatement costs there is a very close equivalence between auctioned tradable permits and environmental taxes.

The distinctive advantage of tradable permits lies in their ability to deliver higher levels of environmental effectiveness. Where policy has to be formulated in the context of considerable uncertainty about how polluters will respond (*i.e.* uncertainty about polluters' abatement costs), tradable permits and emissions taxes will lead to different outcomes. In the former case, the fixed emissions limit will be achieved, but at uncertain abatement cost, while in the latter case, an upper limit will be placed on abatement costs per unit of abatement that is required, but the pollution outcome will not be guaranteed. Which instrument is preferable will then depend on which matters more – excessive emissions, or excessive abatement costs. And where pollution damage rises sharply with additional emissions (possibly because of threshold effects or “snowballing”), the improved environmental effectiveness of tradable permits may be a particularly valuable attribute. In these cases, tradable permits would be clearly preferable to other market mechanisms which cannot deliver such improvements.

Obstacles to the use of tradable permits

Why then are tradable permits not used more? The practical experience to date in major environmental policy applications is almost entirely confined to the US, although there are some very similar arrangements in use in other countries (see Part I), including for the allocation of rights to exploit common resources in the fisheries sector in a number of countries (see Chapter V). Moreover, even within US environmental policy, the use of tradable permit arrangements is uneven. In the field of air pollution policy, extensive use is now made of arrangements for emissions trading, but in the control of water pollution, the uses of tradable permits are much more limited (Chapter II).

Reasons which may have inhibited the greater adoption of tradable permit regimes could include all or some of the following:

- Uniformity of abatement technologies across sources. If marginal costs of abatement do not differ between polluters, there will be no gains from trade, and no point in setting up an emissions trading regime. Whether this is true in each particular case is a purely factual matter, but it should be noted that the financial stimulus provided by the introduction of a market mechanism sometimes encourages polluters to seek options for pollution control which were not previously believed to be present.
- Importance of preventing geographical concentrations of emissions from emerging through trading. Trading may be more appropriate for greenhouse gas emissions and other pollution control problems where the location of the emissions is unimportant in determining pollution damage, and may be less appropriate in the case of policies to deal with location-specific damage.
- More generally, it may be difficult to devise suitable permit regimes where there is significant complexity in the relationship between emissions and pollution damage, as a result of nonlinearity and/or interaction effects between different pollutants and/or sources.
- Political resistance to granting any explicit “right” to pollute to polluters, or to allowing polluters to pay to avoid pollution control obligations if they wish to do so.
- Loss of the ability to make future modifications in pollution control for the lifetime of the permits, except through repurchase of permits. Establishing a tradable permit regime thus involves a degree of commitment to the current level of pollution control which may be inappropriate if there are uncertainties about the future, and this may be politically uncomfortable.
- Influence and lobbying from “stakeholders” in the existing pollution-control regime. One of the potential benefits of market mechanisms is that the regulatory authority does not need to be as closely involved in detailed negotiations with individual firms over pollution control – once an overall limit is set, the market takes over this function. The regulatory agency may resent this loss of power, and may have considerable lobbying influence with policy-makers.

- Uncertainty about the legal viability of the “rights” created by tradable pollution permits. How far, for example, is a reallocation of polluting emissions through trading compatible with other existing legislation, and might firms which “purchase” the right to emit more still be exposed to the possibility of litigation from “victims” of the pollution?
- Difficulty of adapting existing “command-and-control” policies to permit trading. If existing policies specify quantitative emissions limits for individual sources, it may be possible to allow trading of emissions through straightforward incremental modification of the existing control regime. However, many command-and-control policies do not take this form, and tradable permits would require a more fundamental change in operation.

It will be noted that some of the possible obstacles to greater diffusion of tradable permit mechanisms in practical policy have to do with aspects of the relationship between tradable permit regimes and *existing* pollution control policies. This issue of compatibility is the subject of this paper, and is explored further in subsequent sections.

Domestic and international tradable permits

Unlike the debate on environmental taxation, in which the international discussion has focused almost entirely on the possibility of implementing environmental taxes within the national policies of Member countries (albeit perhaps subject to some international co-ordination), the debate over tradable permits has both national and international policy dimensions:

- tradable permits may be used within a country in order to achieve an efficient, cost-minimising, pattern of pollution abatement across different pollution sources, whilst guaranteeing a definite limit on the total amount of emissions;
- tradable permits may be used in the context of international agreements to limit emissions, and would allow countries which could only reduce emissions at high cost to transfer their obligations to reduce emissions to other countries facing lower marginal costs of emissions reduction.

Examples of tradable permit systems of the first type include the various arrangements for emissions trading in air pollution policy (Acid Rain Programme, lead trading, etc), and some limited water pollution trading schemes (see Chapter IV).

Schemes for tradable permits of the second type have been widely-discussed in recent years, in the context of international pollution problems such as acid rain and global warming. Provisions for “joint implementation” in international agreements may be seen as embryonic or partial schemes for internationally-tradable permits. The Kyoto Protocol, especially its call for the use of “flexibility mechanisms”, has sparked a vigorous debate about the potential use of international tradable permits systems for achieving greenhouse gas mitigation objectives.

It is worth noting that the two types of permit trading are, in principle, distinct: a scheme for one does not necessarily entail the other. Thus, for example, arrangements for international tradable greenhouse gas permits would not necessarily require that the countries participating should also operate tradable permit regimes to implement national greenhouse gas abatement. International permit trading, based on trading between governments, could be used to achieve the international distribution of pollution control obligations, while these obligations were then implemented at national level using environmental taxes, command-and-control regulation, or other methods. Nevertheless, if international permit trading were to operate in national permit trading regimes, there would be some potential advantages. In particular, the existence of differences between national permit prices would provide a straightforward indicator of the existence of unexploited gains from international trading.

Nevertheless, many of the issues raised by international permit trading are distinct and separate from those involved in the domestic use of tradable permits. Key issues about the wealth effects of the international allocation of permits, and concerning the absence of the international institutions and mechanisms needed for cross-country enforcement, are very different from the key issues in domestic use of tradable permits. For this reason, the focus of this paper is on compatibility between domestic tradable permit regimes and other instruments, and little attention is paid to those compatibility issues which might arise from the use of tradable permits for the international reallocation of pollution control requirements.

Overview of the paper

Following this introduction, the main analysis of the paper is in four main sections.

The next Section reviews the relationship between tradable permits and other environmental regulation in US policy, discussing a number of concrete cases where tradable permits have been employed. The three cases examined are the well-documented, but rather restricted, permit trading arrangement instituted in the Fox River in the 1970s, the use of tradable permits in the form of “lead credits” to promote accelerated adjustment to future regulations restricting the use of lead in petrol, and the development of various quasi-tradable permit arrangements in USA air pollution policy in the form of various provisions for “emissions trading”.

The paper then considers how compatibility issues between tradable permits and other instruments may be an important factor in successful policy innovation. Existing environmental policy measures form the backdrop against which environmental policy innovation takes place. The introduction of new measures, such as environmental taxes or tradable permits, requires some recognition of deficiencies with existing approaches that might be resolved through the introduction of new instruments to the policy portfolio. It also requires that new measures are capable of operating alongside the existing instruments of policy. It is very rare for existing measures to be wholly abandoned in favour of a new approach, and much more common for new measures to be grafted onto existing arrangements, or for existing and new measures to operate in parallel. Where this happens, there are obvious issues about the compatibility and the co-ordination of tradable permits and other instruments. Do different starting points, in terms of the form of existing regulatory policies, provide more, or less, fertile ground for the introduction of tradable permits than others? Are there some types of exiting regulation onto which tradable permit regimes can be grafted with little additional administrative or enforcement requirement, or which might permit an incremental, or experimental, approach to the introduction of tradable permits?

In addition to the above issues of compatibility in the transition to a policy regime employing tradable permits, there are also important *long-run* compatibility questions which need to be assessed. These are therefore discussed in the next section of the paper. It may be general practice for policy innovation to take place through incremental modification and addition, rather than through the replacement of one instrument by another, but is it, in the long-run, desirable that the two instruments should continue to be employed in parallel? One might, for example, imagine that, once tentative experimentation with a new policy instrument had confirmed its effectiveness, it would then be possible to move to a rationalisation of the instrument portfolio, weeding out those instruments which are less effective, in favour of concentration on those which have the greatest advantages. Is this process one which might be anticipated with tradable permits, or is it likely that they would continue to be employed, even in the longer-run, as part of a broader package of measures? Since, in general, one might expect that the use of multiple instruments might involve additional administrative costs compared with the use of a single instrument, such rationalisation might be expected, unless there are clear reasons to retain parallel measures. What, therefore, might be the reasons which would justify continued operation of instruments in parallel, and in what policy contexts are these likely to be particularly significant?

The paper then concludes with some specific compatibility issues arising in the case of the use of tradable permits in conjunction with environmental taxes, direct regulations, and negotiated (or “voluntary”) agreements.

Relationship between tradable permits and other instruments in practice

The following three sub-sections review the principal applications of tradable permit regimes in US environmental policy, with the aim of identifying those aspects of actual practice relevant to the issue of compatibility. Each section begins with a broad overview of the features of the tradable permit regime and its performance, and then draws out a number of aspects of the experience relating to:

- the points of *interaction* between the tradable permit regime and other environmental policy instruments operated in parallel (*i.e.* other instruments affecting the same industry);

- the extent to which the tradable permit regime has proved to be *compatible* in practice with other environmental policy instruments. What difficulties exist for the efficient operation either of tradable permits or other instruments, arising from the interactions between instruments operated in parallel?

Tradable permits and water pollution: the Fox River case

Like most OECD countries, the US has regulated surface water quality primarily through the use of traditional “command-and-control” approaches. The basic regulatory framework was set out in the 1972 Federal Water Pollution Control Act, renamed the Clean Water Act in 1977. Under this legislation, the US Environmental Protection Agency (EPA), or the delegated State authority, specifies the pollutants that may be discharged from a particular source and the quantity and/or concentration limits imposed on such discharges. By 1983, municipal and industrial point sources were required to meet minimum discharge standards based on “best available technology economically achievable”.

In contrast with the case of air pollution, tradable permit mechanisms have been little used in US water pollution policy. Only a limited number of schemes have been initiated, and in some of these (such as the schemes for phosphorous emissions to the Dillon and Cherry Creek Reservoirs in Colorado and for nitrogen and phosphorous effluent in the Tar-Pamlico River in North Carolina), the emissions limits have yet to “bite” to the extent that the provisions for trading become operative. One early tradable permit application to water pollution in the Fox River, Wisconsin has, however, been widely discussed, and the general perception that it failed to achieve the anticipated efficiency gains may be one reason for the slower diffusion of tradable permits into water pollution policy (see also Chapter IV).

The Fox River trading system established an initial allocation of rights in the form of a single limit on the total organic effluent discharges of a paper mill or municipal discharger. The limit applied to the entire facility, rather than placing separate limits on each individual pollution source within it, and consequently it already gave polluters a considerable amount of flexibility in determining how compliance with the overall limit would be achieved. The provision for trading, however, reflected an *ex ante* assessment that there were substantial differences in marginal abatement costs between the various dischargers, such that a reallocation of abatement between different dischargers would potentially generate substantial efficiency savings, of the order of \$4.5-\$7.5 million annually (O’Neil *et al.*, 1981).

The scheme introduced permitted inter-firm effluent trading only on the basis of prior approval from the regulatory authority (WDNR), and only subject to certain significant conditions.

Ex post performance of the trading regime has been disappointing, and the abatement cost savings from reallocation of abatement through trading have not been realised. In the decade following the start of the mechanism in 1981, only one trade took place, between a paper mill that stopped operating its waste water treatment plant and a municipal waste water treatment plant that began taking the mill’s waste water (and was given the wasteload allocation previously held by the mill). This negligible amount of trading activity may be partly explained by the rather constrained scope offered for trading, due to the various restrictions on permitted trades noted above. In addition, in the academic research and policy debate on the Fox River scheme, various further reasons have put suggested for the negligible use made of the opportunity which it offered for emission trading.

- It is conceivable that there could have been some uncertainty about the *legal viability* of the rights being traded, which might have made potential purchasers uncertain of their value. Since the standards set by the WDNR did not conform with the national policy of uniformity established by the Clean Water Act, the standards could be vulnerable to legal challenge (Anderson, Hofmann and Rusin, 1990, p. 36).
- The *financial incentives* for trades were relatively weak, when viewed in the context of other management concerns. Pollution control costs for the paper industry on the Fox River are less than one percent of the cost of the product, and the potential savings from trading are only a small proportion of these costs.

- It was unclear how *rights to future permit allocations* would be affected by participation in emissions trading, and this may have deterred potential sellers from offering surplus emission rights for sale. Sellers may, in particular, have been concerned that by selling their current emission rights they might also be giving up their rights to be allocated permits in the future.
- *Transactions costs* borne by traders, in the form of the administrative requirements of the scheme, may have been large in relation to potential trading gains, thus decreasing the incentives to trade (Apogee Research, 1992). For example, the procedures for approval of trades and making the necessary permit revisions could take at least six months, which would have the effect of shortening the life of the traded rights, effectively reducing their value (Hahn and Hester, 1989).
- *The characteristics of the industry involved* may also have discouraged trades. The industrial dischargers involved (i.e. the pulp and paper plants) are in the same industry. It has been suggested (David and Joeres, 1983; Hahn, 1991) that, due to the oligopolistic structure of this industry, the plants may not behave as competitive firms in the permit market. This may partly explain the dischargers' apparent lack of enthusiasm to participate (Hahn, 1991).

Compatibility between tradable permits and other instruments

The above account shows two key practical interactions between the Fox River tradable permit regime and other policy instruments affecting the Fox River polluters:

- the permits were derived from the existing regulatory emissions limits – they were defined in the same way, and the initial allocation was made on the basis of existing plant-by-plant limits.
- trading arrangements were supervised closely by the existing regulatory authorities, who required that each trade be approved, and who exerted considerable control on trades through this requirement. Arguably, the role of the regulatory authority in approving trades, and the limited grounds on which approval has been given, inhibited the successful development of an active permit market.

Both the existing command-and-control policy and the tradable permits were essentially directed at the same objective: the reduction of levels of dissolved oxygen in the river. The stringency of the pre-existing level of command-and-control regulation on this group of polluters had left little obvious scope for further reductions in pollution through conventional policies. The aim of the tradable permit mechanism was unclear (and purely cost-reducing trades were explicitly prohibited), but one possible rationale that would be consistent with this starting point is that the tradable permits could create an incentive for polluters to identify further possibilities for abatement not apparent to the command-and-control regulators, and therefore not attainable within the existing regulatory regime.

A further compatibility issue is that the “ground rules” about the relationship between trading activity and regulatory limits were not made sufficiently clear at the start of the system. As the preceding discussion suggests, there may have been some potential participants in trading who were discouraged from making trades for fear that this could affect their subsequent treatment by the regulatory authorities.

Tradable permits and adjustment: the case of lead trading

From the early 1970s, the use of lead additives in petrol to raise octane levels and improve engine performance has prompted concerns over adverse health effects. In the US, the Environmental Protection Agency (EPA) required petrol companies to introduce unleaded petrol from 1974, and placed restrictions on the lead content of leaded petrol. These limits were tightened sharply during the 1980s, with the aim of eliminating leaded petrol entirely.

To ease the process of adjustment, especially among smaller suppliers, to these new stricter standards, a system of lead trading (formally known as “inter-refinery averaging”) was instituted from July 1983, to operate during the transition period to the new lower limit of 0.1 grams per gallon (see Chapter II).

Lead trading allowed refiners and importers to trade lead reduction credits in order to meet limits for the lead content of petrol. The lead reduction credits were to be created when suppliers of leaded petrol achieved lead levels lower than those required by the new limits. The system allowed for both internal “trading” (*i.e.* flexibility) for an individual supplier, as well as external trading between suppliers.

Internal trading took two forms. First, the system operated on the basis of quarterly accounting periods, and each refiner was required to show compliance with the required standard (either through lead reduction, or by purchasing lead rights from other suppliers), over the quarter as a whole. Internal flexibility within the quarter was possible, and a firm could use lead rights itself by adding more lead to its petrol at some point during the quarter than would otherwise be allowed. Second, for part of the period, internal flexibility was also possible, by “banking” lead rights created in one quarter for use in future periods. Initially, rights simply expired if they were not used or sold during the quarter in which they were created. However, in all four quarters of 1985, refiners were able to “bank” rights for their own future use through to the end of 1987, or for sale to other refiners until the termination of the lead trading programme at the end of 1986.

External trading permitted lead rights created by one supplier to be used by another. Transactions (though not transaction prices) were reported to the EPA, but did not require prior approval, and were largely unrestricted. The only significant exception was that for a period at the start of the programme, smaller refiners were to be subject to less stringent standards than larger refiners, and were not then permitted to sell lead rights to larger refiners. Also, petrol lead content standards in California were more stringent than EPA standards, so refiners in California were not allowed to trade rights in order to exceed the State's limits.

The general consensus of studies evaluating the experience of the lead trading programme is that the scheme worked much better than most other practical applications of tradable permit mechanisms. There has been some discussion of possible reasons for this. Among these are the facts that the time-scale of the exercise was clearly set out in advance (although, in practice, subsequently adjusted through the introduction of banking provisions); that rights and obligations were clearly defined; and that the commodity traded was clearly defined and homogeneous. A further positive factor may have been that the personnel at different refineries were already accustomed to conducting transactions with each other (for refinery feedstocks and products). Hahn (1991) also observed that the programme was implemented after agreement had been reached about environmental goals, which may have helped to overcome resistance to trading on the part of participants.

Compatibility between tradable permits and other instruments

The relationship between lead trading and the existing regulatory regime is strikingly different from that observed in the Fox River case. In the case of lead trading:

- The lead trading scheme and existing regulations were *directed at different objectives* – lead trading was designed to ease the transition to a long-run outcome governed entirely by command-and-control limits, and it explicitly aimed at cost-effectiveness.
- The lead trading scheme was of *defined and limited duration*, and this was fully understood at the start of the scheme. The implications of trading activity for future regulation were nowhere near as ambiguous as in the Fox River case.
- Unlike the Fox River scheme, *trades were not subject to approval*, and the regulatory authority had only a very limited role in supervising and monitoring trades.

Lead credits were created by savings in lead content against a percentage content target expressed as units of lead per unit volume of petrol, calculated over all petrol produced, and were then specified in terms of kilograms of lead. The monitoring requirements for enforcement of the lead trading arrangements therefore differed to some extent from those required for regulation, and potentially required greater information about output refinery levels (to calculate total lead use). Compliance could not simply be assessed on the basis of analysis of the lead content of sample petrol outputs, even though the regulatory policy is expressed in these terms.

Tradable permits and air pollution: lessons from the US experience

The EPA's emissions trading programme was introduced during the 1970s, with the aim of giving industry increased flexibility in compliance with the requirements of US air pollution policy. This programme operates nation-wide and covers all significant stationary sources of pollution for five principal air pollutants – hydrocarbons, nitrogen oxides, particulate matter, sulphur oxides, and carbon monoxide (see also Chapter II).

The Clean Air Act distinguishes between new, modified, and existing sources. Existing sources are those that existed when emissions were first inventoried in the mid-1970s, those built since then are termed new sources. Modified sources are existing sources which have been altered where this alteration has resulted in significant increases in emissions. These different types of sources have different emissions trading options available to them (“netting”, “offsets”, “banking”, and “bubbles”).

In some areas, there are additional regulatory restrictions on trading activity. For example, trades between sources in a certain geographical relation to each other may not be allowed in order to prevent pollutants from concentrating in a particular area. Administrative requirements may differ from State to State, but all trades must be approved, and this is often a lengthy and costly process.

The performance of emissions trading has been mixed, and the level of trading activity under each of the four programmes varies dramatically. In particular, Hahn (1989) observes that the cost savings that have been realised have been almost entirely from internal trading and fall far short of the potential savings which could be realised if there were more external trading.

In a detailed study of the performance of emissions trading in the Los Angeles basin area, Foster and Hahn (1994) identified a number of key factors affecting performance of the trading arrangements:

- Emissions trading activity has been heavily affected by the frequent amendments to the detailed regulations governing the market, which have generated uncertainty.
- In some cases, the regulations have had the effect of creating a large number of small segmented markets within which trading is thin, and pricing behaviour inefficient due to the small number of participants.
- Firms encounter difficulties in identifying suitable trading partners, and are often limited to trading internally.
- The transaction costs associated with trading were a major source of inefficiency, and frequently exceeded the market value of credits themselves. These costs included the lengthy process of approval for trades (which can take five to twelve months), and financial costs including administrative fees and the cost of preparing documentation.
- The process of approval was also responsible for considerable uncertainty, since only about 20 per cent of trades which reached the approval stage were approved straightforwardly, and about 40 per cent were rejected.

Conditions for successful policy innovation

Policy innovations, such as the introduction of tradable permit mechanisms, face a struggle for survival in the policy-making world. Tried and tested conventional policies have many natural advantages over innovative (but untried) schemes. Where they are already in operation, conventional policies can easily be adapted or redirected to meet new needs, and the process of adaptation generates few uncertainties. Most of the effects (and side-effects) of existing policy instruments are already well-known, and the likely effects of adjustments to existing policies can be observed or inferred from existing experience. Even where what is at stake is the introduction of measures which are wholly-new to a particular country, the choice of a conventional approach allows policy-makers, and the various parties to the policy debate, to assess the likely effects from observation of the performance of many similar measures in operation abroad.

In comparison, the advocates of new and innovative policy instruments have to invite policy-makers to take a major step into unfamiliar territory, guided by little more by conceptual argumentation, or the results of hypothetical calculations and simulations. The advocates of a genuinely new approach cannot

point to its success elsewhere, and cannot easily reassure those who are sceptical about its likely effectiveness or worried about unanticipated side effects. Faced with such an imbalance in the evidential basis for policy, it is perhaps surprising that policy innovation in environmental policy has taken place to the extent that it has, and, in particular, that there has been such a willingness to implement market mechanisms in such a large number of OECD Member countries, and in such a wide variety of practical applications.

One spur for this has been the fertile intellectual and political climate for market-orientated reform in many OECD countries. Confidence in the abilities of the market seems to have been able to override unease about the untried and untested. The diffusion of policy applications of market mechanisms has no doubt also been helped by the vigorous international policy debate, and the exchange of information and experience, at the OECD and elsewhere. A further reason has undoubtedly been a growing realisation of the deficiencies of the existing approach, especially if it is to be applied to some of the major international environmental problems, such as global warming, which may require far-reaching changes in patterns of production and consumption.

But awareness of the deficiencies of the existing approaches, and knowledge of the theoretical superiority of alternative arrangements, may still be insufficient to achieve any major change in the direction of policy, or in the instruments which legislators are prepared to introduce. Scepticism may still be a major obstacle, and its paralysing effects on policy may be magnified by the resistance of these within the public administration and private sector whose influence and expertise (and perhaps even continued employment) depends on the continuation of existing approaches. Command-and-control regulation, if it is to work at all efficiently, requires environmental regulators to be in possession of considerable good-quality information about the production technologies and abatement opportunities of private sector polluters. Through their possession of this information, regulators can exert considerable influence on firms. In turn, through their influence on this information, firms can exert influence on the decisions of environmental regulators. Using market mechanisms to regulate pollution makes it far less important that regulators have such detailed information about the firms with which they interact. Using such mechanisms, therefore, would cut out the need for a substantial (and powerful) part of the existing regulatory bureaucracy.

In the face of uncertainty about the effectiveness of innovative but untried approaches to policy, and resistance from those who stand to lose, it is perhaps most likely that policy innovation will succeed where it can build incrementally on the existing apparatus of policy, taking small steps, (initially at least, working in parallel with existing policy instruments, rather than replacing them). This approach may pose less of an immediate threat to those engaged in current policy, and allows effectiveness to be observed and tested before removing the safety net provided by the continued presence of existing measures.

Of course, there may be countervailing disadvantages to the incremental approach. It may be difficult to disentangle the effects of the new instrument from those of the old. Instruments operated on a small scale may have different effects from those operating at a larger scale – financial incentives may not change the behaviour of firms, if they are so small that they can be overlooked or ignored. Instruments may have effects which conflict, and may subvert each others' effectiveness (detailed command-and-control regulation may give little scope for polluters to reallocate their emissions through trading). And, of course, there is a danger too that parallel operation becomes permanent, even where it involves costly and unnecessary duplication of administrative and compliance costs.

Nevertheless, it is worth noting that in each of the three US tradable permit applications discussed above, the measures adopted played a limited role within an overall policy framework which was predominantly based on existing command-and-control regulation. In none of the three cases was the tradable permit regime introduced as a replacement for existing control; in each case (albeit in different ways), the existing regulatory approach remained largely intact.

- In the case of the Fox River, tradable permits were introduced on a single, limited, stretch of river, where further progress using command-and-control regulation would have been difficult. They took as a starting point for trading the existing regulatory limits, and gave the regulatory authority a major (and arguably excessive) role in approving trades.

- In the case of lead, trading was introduced to ease the transition to new and more stringent regulatory standards. The trading scheme was explicitly time-limited, and was abolished at the end of the transition period.
- In the case of emissions trading, provisions were introduced incrementally, to permit various forms of flexibility in compliance with regulatory requirements. As with the Fox River, existing regulatory limits essentially defined the “currency” for the emissions trading scheme.

The scope for incremental introduction

The preceding section has highlighted the need for some compatibility and coexistence between tradable permits and existing regulatory instruments, in order to overcome the various forms of opposition that innovative policy may encounter. The remainder of this section briefly reviews compatibility issues of a technical rather than political nature, under four headings. What practical conditions, under each of these headings, need to be satisfied for it to be possible to introduce tradable permit mechanisms on an incremental basis, alongside existing approaches?

- Definition of emission rights

A system of tradable permits requires the commodity to be traded be defined in some coherent, measurable way, so that contracts for trades can be written and enforced, and so that the regulatory agencies can understand the implications of trades that take place, and can monitor and enforce the post-trading environmental performance of individual sources.

How straightforward it is to define emissions rights will vary across different types of pollution. If the tradable permits system requires emissions to be measured, the conditions under which these measurements take place need to be defined and standardised.

If existing command-and-control regulation takes the form of firm-by-firm emissions limits, or if the emissions levels of individual polluters are already monitored as part of the apparatus of regulatory enforcement, it may be possible to adapt these existing measurement procedures for the purposes of introducing an emissions trading system.

- Allocation of rights

Under a system of tradable permits, property rights in emissions need to be clearly defined, and some basis needs to be established for the allocation of rights to different sources. This could be through auction, or permits could be allocated to polluters according to some “grandfathering” formula. Whilst there are advantages in auctioning permits (not least the revenues that can be raised), it may be considerably more straightforward to begin by allocating permits to the existing firms in the industry. Grandfathering does not require an auction mechanism to be established, and the free distribution of permits through grandfathering may encounter less opposition from polluters than if permit rights were to be sold to the highest bidders.

One possible approach to grandfathering that would be feasible in a situation where firms are already regulated on the basis of source-by-source emissions limits would be to adopt these existing limits as the tradable commodity. Polluters would, in effect, be allocated tradable permits equal to their current emission limits. Since these limits are already defined, and polluter compliance with the limits already monitored and enforced, the move to tradability would be a small incremental step, requiring the minimum of modification to command-and-control arrangements.

The form taken by existing regulations thus greatly affects the ease with which tradable permits can be introduced. Existing regulations which require particular production or abatement technologies to be employed cannot easily be converted or extended into a tradable permit regime. Emissions-based regulation, by contrast, appears to be a more promising starting-point for the development of a tradable permit system.

- Enforcement

Even where existing regulation takes the form of source-by-firm emissions limits, arrangements for enforcement will not necessarily be based on continuous emissions monitoring. In practice, the use of paper documentation and occasional physical inspection may provide an adequate alternative basis for identifying non-compliant firms, and may be cheaper both for firms and for the authorities. It may, however, be undesirable for enforcement to continue to operate in this way, if firms are able to trade emissions. Once emissions can be traded, residual emissions have a cost to the firm, and it may no longer be adequate to rely on relatively infrequent “proxy” measures of emission performance. Financial incentives for the underreporting and concealment of emissions will be greater, so accurate and continuous measurement of emissions may become indispensable.

- Requirements for polluter responses

The technical conditions needed for the introduction of tradable permits also include various requirements in terms of the organisation and behaviour of the regulated firms. Are they already organised in a way that makes it likely that they be responsive to the price signals emerging from emissions trading? Do they already monitor their emissions performance, and use this information in making decisions on production technologies? Are they already aware of the range of abatement technologies that might be available to them, and to their abatement costs? It is likely that the awareness of firms about abatement options, and their responsiveness to financial incentives will be conditioned by the existing pollution control regimes. If existing policies impose rigid requirements for the use of particular abatement technologies, firms may have little knowledge of their abatement costs, and little appreciation of the other abatement options which may be available.

When might a “mix” of instruments be desirable?

The preceding section has discussed issues concerning the compatibility of tradable permits with existing instruments of environmental policy, focusing on those compatibility issues which may inhibit the innovative adoption of tradable permits. If policy innovation generally takes place through incremental modification and addition, rather than through the replacement of one instrument by another, successful innovation is more likely where existing instruments are compatible with the parallel operation of a tradable permit system, or where the modifications needed would be minor.

However, this need for compatibility during the process of policy innovation does not necessarily imply that tradable permits and other instruments should permanently be operated in parallel. Continued parallel operation might be undesirable for a number of possible reasons. Using two instruments may, for example, be:

- *wasteful* (if employing two instruments doubles the administrative and compliance costs borne by government and regulated firms);
- *inefficient* (if the continued use of regulatory instruments prevents the efficient reallocation of pollution abatement that tradable permits should encourage); or
- *unnecessary* (if only one instrument actually “bites”).

In these circumstances, it might be desirable for the innovative adoption of a new instrument to lead eventually to the replacement of some of the existing apparatus of pollution control. Once the new policy instrument had confirmed its effectiveness, it would be possible to move to a rationalisation of the instrument portfolio, weeding out those instruments which are less effective, in favour of concentration on those which have the greatest advantages.

It is possible that this sequence could (and should) be followed in the case of tradable permit applications. Once sceptical public opinion has been reassured that tradable permits can indeed ensure adequate pollution control, it may be possible to remove remaining command-and-control legislation, and it may also be very desirable to do so if this earlier legislation prevents the full benefits of the tradable permit system being realised.

It is also worth bearing in mind that there may be “stakeholders” in the existing regulatory process who may have an interest in its retention, and who may lobby hard to prevent rationalisation taking place. Much of what “command-and-control” regulators have to do, in the way of negotiating the emission limits or technology requirements with individual firms, is unnecessary when pollution control is based on tradable permits, and much of the power they possess is undermined when market forces (rather than officials) determine the pattern of pollution abatement. It would be unrealistic to think that the people involved in these aspects of the existing system would be happy to see their influence diminished, and that they would not seek reasons to prevent the rationalisation of pollution control policy based on the retention of only the most effective instruments.

However, it is also possible to think of circumstances in which it might genuinely be desirable to retain some aspects of existing environmental policy instruments, operating permanently in parallel with tradable permits. This Section of the paper reviews a number of such possible cases, in which the permanent joint use of two or more policy instruments might be preferable to the selection of the single, most effective, instrument.

The “safety valve” argument

One widely-discussed argument for using tradable permits in combination with another pollution control instrument is the “safety-valve” argument advanced by Roberts and Spence (1976). In this scheme, tradable permits would be supplemented with elements of price-based regulation, in effect taking the form of supplementary emissions taxes for “excess” pollution, and pollution-reduction subsidies for “super-abatement”. The basis of the argument is as follows:

As discussed previously, one of the advantages of tradable permits over some other environmental policy market mechanisms (such as environmental taxes and charges) is that they can help guarantee that polluting emissions will not exceed a certain level (given by the number of permits issued). By contrast, the environmental impact of an emissions tax set at a particular level depends on the responsiveness of polluters to the incentive provided by the tax. Ultimately, this responsiveness should be governed by the marginal costs of pollution abatement faced by individual polluters, in that polluters would be expected to choose to reduce pollution whenever abatement is cheaper than paying the pollution tax. (There may, in addition, be important issues about the internal organisation and decision-making of the firm, which may prevent an efficient pattern of response to the tax.)

Where the regulatory authority knows the pattern of abatement costs for the regulated industry with a reasonable degree of accuracy, environmental taxes and charges may be able to deliver the required level of pollution control as reliably as tradable permits. However, where abatement costs are uncertain, an instrument such as tradable permits that directly regulates the overall quantity of emissions may have a different outcome from that achieved by an instrument that regulates emissions through imposition of a price signal such as a tax per unit of emissions.

It is important to recognise that in this situation, both instruments have potential disadvantages, either in terms of excessively-costly pollution abatement which cannot be justified by the pollution damage avoided, or of inadequate pollution control, leading to excessive pollution damage that could have been avoided cost-effectively through greater abatement. Emission charges or taxes place an upper limit on the abatement costs per unit of pollution that will be incurred, but have an uncertain outcome in terms of pollution quantities. Tradable permits, on the other hand, achieve a guaranteed effect on emissions, but at the price of correspondingly greater uncertainty about the abatement costs that will have to be incurred.

If a choice has to be made between the two instruments, then the one that is preferable will depend on the characteristics of both abatement costs and pollution damage in the particular application. Cases where abatement costs are sharply rising with additional pollution control will tend to favour “price-based” regulation, such as an emissions tax, which will prevent firms finding themselves facing the risk of undertaking unexpectedly-costly abatement measures. Cases where pollution damage rises sharply with additional pollution will, by contrast, tend to favour the employment of “quantity-based” regulation, such as tradable permits.

However, it may be, that in these conditions of uncertainty about polluters' abatement costs, a better outcome can be achieved by using both instruments together, rather than by relying on tradable permits or an emissions tax alone. The Roberts and Spence scheme combines tradable permits with provisions for repurchase of permits if their price falls below a certain floor, or additional supply of permits if the price of the initial quantity offered exceeds a certain upper threshold. The effect of this is to limit the feasible range of permit prices that would be experienced; and hence, to limit the exposure of polluters either to excessively high abatement costs, in excess of the environmental benefits, or the exposure of the environment to an inflexible pollution target, when it becomes clear that further abatement could actually be achieved very cheaply. Within a "central" range, therefore, the scheme guarantees that a particular pollution abatement quantity will be achieved, but it allows for some relaxation of this guarantee where circumstances are such that achieving this quantity would either impose excessive burdens on polluters, or where sticking to the initial abatement target would forego opportunities for further cheap (and cost-effective) abatement.

This argument would then justify the long-run operation of tradable permits in parallel with emission-tax and/or abatement-subsidy instruments. [It also could support incorporating such safety valve features into command-and-control regulation, which (like tradable permits) guarantees quantity, but at uncertain abatement cost.] It is, however, not applicable to the issue of the combined operation of tradable permits and command-and-control regulation, since these are both essentially quantity-based forms of regulation, with similar risks of extreme outcomes in terms of abatement cost.

Enforcement cost

Parallel operation of two different policy instruments directed at the same problem will not always incur unnecessary costs of administration, enforcement, and compliance. Certainly, it is possible that there will be wasteful duplication, and the use of two instruments will mean that these dead-weight costs of policy are unnecessarily magnified. However, this will not always be so, and will depend on how the various costs of administration and enforcement relate to the "stringency" of policy.

The costs of administering and enforcing any policy instrument are expected to be higher, the more demanding the standard of environmental protection which the policy aims to achieve. Compliance is likely with little effort, and at little administrative and enforcement cost, if the policy tries to achieve only a very small change in polluter behaviour. Compliance will be much harder to achieve, and the costs of administration and enforcement correspondingly higher, if the policy aims to require major modifications in polluter behaviour, involving large abatement costs. In this situation, some polluters may be inclined to try to evade the requirements of policy, and to conceal their non-compliance, since the potential penalties from being caught and fined may be much less than the saving in abatement costs through non-compliance.

It is conceivable that the costs of administration and enforcement could rise more than in proportion to the "stringency" of policy. Instruments which are "under pressure" from polluters who would stand to make large savings from non-compliance, may be much more costly than the same instruments aiming at a lower level of abatement. If this non-linearity in administration and enforcement costs is sufficiently severe, it may be less costly to achieve a given abatement level by spreading the burden of policy across two instruments, than it is by loading the policy entirely onto a single instrument.

A critical consideration here will be the informational independence of the two instruments: is their enforcement dependent on the *same* monitoring information? If so, polluters may be able to avoid compliance with both instruments through (costly or risky) efforts to "fiddle the figures" in relation to this required information. Only if the operation of the two instruments is dependent on *different* information (*e.g.* on compliance with required technologies, and on emissions levels) is it likely that there would be benefits to spreading a policy intervention across multiple instruments.

To limit geographical "hot spots"

Joint use of more than one instrument may be appropriate where unrestricted use of a market-based approach might lead to undesirable geographical redistribution through trading, creating "hot spots" with

very high levels of pollution damage. This may be undesirable from the perspective of equity, and uneven pollution may also generate even greater unevenness in environmental damage if critical loads are exceeded in the “hot spot” area, or other threshold effects occur. Combined use of market mechanisms with some form of regulatory control to limit geographical concentration of pollution may allow at least some of the efficiency gains from market mechanisms to be achieved, without excessive “hot spot” problems.

The “scatter-gun” argument

Use of more than one instrument may also be desirable where none of the available instruments can be precisely targeted to the exact environmental externality. An example drawn from urban traffic policy may illustrate the point.

For various political or technical reasons, the ideal instrument to deal with urban road congestion – the introduction of exact time-of-day congestion pricing – may not be feasible. None of the available alternatives, however, provides a very close substitute in discouraging urban congestion, if used on its own. Discouraging urban congestion by increasing petrol taxes, for example, hits rural road use too, and urban road use at less-congested times of day, as much as it affects peak-hour urban traffic. Using petrol taxes alone to discourage urban congestion would be inefficient, but they may have a role to play as part of a package using multiple instruments. If the ideal instrument cannot be used, the next-best approach may be to combine a petrol tax with taxes on complements to congestion (*e.g.* parking) and subsidy of substitutes, such as public transport. Taken together, these may be a reasonable second-best approach.

Where the responsiveness of polluters varies

A further argument for operating multiple instruments in parallel may be that the effects of different instruments vary across polluters. Some polluters may be relatively-insensitive to financial incentive mechanisms (if, for example, their internal organisation separates accounting/finance functions from technology decisions). Such organisational separation may be efficient from the point of view of the enterprise as a whole, since it may lead to an efficient division of labour with respect to the most important decisions facing the firm. However, it does have the drawback that such a firm may be insensitive to the financial incentives created by environmental market mechanisms (especially incentive effects from environmental taxes and charges that are small relative to other business costs). Whilst other firms may respond efficiently to such incentives, the pollution decisions of a firm organised in this way may be more efficiently controlled through non-financial mechanisms.

Policies appropriate to firms of different sizes

An important group of circumstances in which it may be appropriate for environmental policy instruments of different types to be operated in parallel is where the optimal instrument choice varies for firms of different sizes. For example, if there are large fixed costs of emissions measurement and compliance monitoring at the level of the individual firm, then it may be most efficient to regulate small and large firms using different instruments. For large firms, the potential efficiency gains from allowing flexibility in abatement responses may be large relative to the costs of firm-level emissions measurement. It may therefore be cost-effective to employ market mechanisms in regulating these firms. For smaller firms, however, the efficiency savings from regulatory flexibility may be too small to warrant the cost of firm-level emissions monitoring, and it may be more efficient to employ methods of regulation which have lower administrative and enforcement costs. For example, these firms might be regulated by mandating the use of a particular abatement technology. The installation of this technology could be verified by a one-off inspection, and the need for regular or continuous emissions measurement avoided.

International practice includes a number of examples where firms of different sizes are regulated using instruments of different types. The scope of application of the Swedish nitrogen oxides charge was, for example, confined to a relatively small number of large emissions sources, for which emissions measurement costs (estimated at around 300 000 kronor per plant) could conceivably be outweighed by abatement cost savings. Small firms are not subject to the charge, which limits the total administrative

cost of the system. Another example is the case of the water pollution charging system in the Netherlands, where large emissions sources are charged on the basis of emissions measurements, while small emissions sources pay a flat charge, unrelated to actual emissions performance. The incentive charging effect is thus restricted to larger sources, where the potential efficiency savings from the use of a market mechanism might be enough to warrant the costs of measurement.

In such cases, the co-existence of direct regulation (for smaller firms) with tradable permits (for larger firms) may be the most cost-effective instrument choice. However, this co-existence may be uncomfortable, in that it may impose different regulatory burdens on large and small firms. In particular, auctioned tradable permits impose a cost on firms (of the permits that have to be purchased for pollution) that has no counterpart in direct regulation through non-revenue-raising instruments. As with the Swedish nitrogen oxides emissions charge, it may be necessary to find a way of “levelling the playing field” between the large firms (subject to the revenue-raising instrument), and the smaller firms (which are regulated in ways that do not impose a corresponding burden). With emissions taxes, this can be done by recycling the revenues collected to the participating firms, perhaps in proportion to output or initial emissions, so that the average net payment for firms subject to the charge is zero. The same approach could be adopted with the revenues from a permit auction. Alternatively, permits might simply be distributed free (grandfathered), which would have an equivalent effect.

Distributional or competitiveness problems

The extent to which environmental policy can operate solely through the use of environmental taxes, charges or tradable permits may be limited in practice by the (actual or perceived) distributional or competitiveness effects of these instruments. This is most likely to be a problem with revenue-raising instruments (such as taxes, charges and tradable permits), because the pattern of tax, charge or permit payments on residual units of pollution may be a significant additional cost for some polluters, over and above the abatement costs involved in reducing emissions. The use of revenue-raising market mechanisms in the energy sector, for example, is likely to lead to large additional costs for both businesses and households, reflecting the importance of energy as a business input and in household consumption, as well as the low price-elasticities of energy demand. Concerns about the possible effects of carbon tax payments on the competitive position of energy-intensive sectors of industry, and on the welfare of poorer households, have been prominent in the policy debate in a number of OECD countries. In some cases, these concerns have prevented such measures being adopted. But in other cases, they have limited the politically-feasible carbon tax to a low level, requiring a major part of total abatement required to be pursued using other instruments alongside the carbon tax.

Dangers of “packages”

Although the seven previous headings suggest that there are a range of circumstances in which parallel use of tradable permits and other instruments might be justified and desirable, it is worth reiterating some of the observations with which this section began – that the normal presumption would be that long-run retention of multiple instruments to deal with a single pollution problem would be inefficient.

In particular, when more than one instrument is employed, there is a danger of excessive costs (except in the particular circumstances discussed earlier) for at least three reasons. First, each instrument will be likely to have some separate administrative requirements, increasing the administrative costs of the public agencies involved in pollution control. Second, the compliance costs for polluters are likely to be increased due to the greater complexity of the regulatory arrangements. Third, public decision-making may be more complex, since there will be a need for co-ordination of decisions on different instruments, possibly administered by different parts of government.

Compatibility with other market mechanisms

The preceding discussion of the compatibility between tradable permits and other instruments has largely (though not exclusively) considered circumstances where it is envisaged that tradable permits

might evolve from, or be employed in parallel with, various forms of direct “command-and-control” regulation. However, there are also possibilities for tradable permits to be operated in parallel with other market mechanisms, such as taxes or voluntary agreements. Some of the specific compatibility issues that might arise in these applications are discussed below.

Compatibility of tradable permits with environmental taxes

Environmental policies in many OECD countries make increasing use of “green” taxes, both for incentive purposes (*e.g.* to discourage production or consumption of environmentally-damaging commodities, or to encourage substitution to “greener” alternatives), and to raise revenues for environmental purposes. Some countries have introduced comprehensive “green” tax reforms, in which significant additional revenues are raised through a package of ecotaxes, sufficient to permit reductions in some existing distortionary elements of the tax system such as taxes on labour and/or capital. Introduction of tradable permits in a context where substantial green taxes are already levied may raise several issues.

As noted earlier, there may be circumstances in which it is desirable for a tradable permits regime to be combined with some form of pollution tax and subsidy, in order to place limits on the range of possible fluctuation of permit prices. This may reduce the extent to which a permit regime can guarantee a particular pollution level, but the gains in terms of reduced risk of “extreme” abatement costs may more than offset this defect.

Generally, the revenue effects of auctioned tradable permits and emissions taxes would be similar. Grandfathering of tradable permits, on the other hand, foregoes permit auction revenues, and, for a given environmental effect, would also reduce the revenues that could be obtained from an environmental tax. As discussed earlier, there may well be reasons to limit the use of revenue-raising instruments, in which case grandfathered permits would be a useful supplement to an environmental tax that is constrained to an inefficiently-low level by competitiveness or distribution concerns.

Nevertheless, despite the similarity in revenues between auctioned permits and environmental taxes, there may be important institutional differences, especially in how different types of revenue are treated in public budgets.

One source of considerable uncertainty in the value of permits may be future tax levels. Increasing the level of a pollution tax would give rise to a direct (and equivalent) reduction in the value of pollution permits held. This will be incident on the current holders of such permits, who will experience the effect as a capital loss, creating a disincentive to hold permits. Efficient operation of some aspects of a permit regime (*e.g.* banking, and precautionary holding of permits by some polluters) may be inhibited.

Compatibility with voluntary approaches

There has been considerable interest recently in the potential for “voluntary” or “negotiated” agreements to contribute to environmental policy. Such agreements could include both explicit or implicit agreements – in other words, they could include both formal and/or legally-enforceable contracts between government agencies and the private sector, under which polluters agree to control pollution in specified ways. Alternatively, they could involve an understanding between government and the private sector which falls short of an explicit or enforceable contract, but as a result of which polluters nevertheless agree to take certain measures.

Voluntary agreements may provide the opportunity for attaining some of the benefits of the use of market mechanisms in environmental policy in a context which could, in some circumstances be more acceptable to the private sector, and may therefore provide a possible way forward if more conventional market mechanisms face insurmountable political opposition. The sense in which they may be able to achieve some of the benefits of market mechanisms is typically that they provide a context in which polluters negotiate or agree amongst themselves the distribution of abatement obligations.

This process of negotiation between firms may take place prior to the negotiation of the agreement between firms and government, in which case the agreement could specify the agreed firm-by-firm obligations. Alternatively, it may take place after the agreement has been established; the agreement may

then specify an overall level of pollution abatement to be achieved by the participating industry, and the participating firms may then be left to agree amongst themselves the contributions that each will make to the total required abatement. Regardless of the stage at which this inter-firm negotiation takes place, however, polluters may be able to agree collectively on a more cost-effective distribution of abatement than if abatement requirements were imposed on individual polluters, through the unilateral decisions of the regulatory authority.

Questions of the compatibility of tradable permits and voluntary agreements arise in two different contexts:

- First, there are issues concerning the co-existence of voluntary agreements with public policy measures for environmental regulation in the form of tradable permits. How does the existence or introduction of one affect the likelihood of successful operation of the other?
- Second, there are issues about the possible integration of a tradable permits mechanism as one of the elements within a voluntary agreement. Here, the issue is whether a tradable permit mechanism might facilitate the efficient allocation of abatement responsibilities between firms participating in the voluntary agreement. The tradable permits would operate here as a private sector device to facilitate efficient bargaining and agreement, rather than as a public policy measure.

Voluntary agreements and tradable permits in parallel

In considering whether voluntary agreements are likely to be compatible with tradable permit measures, these issues vary depending on which was in place first – the voluntary agreement or the tradable permit regime.

The first case concerns the possible impact of existing tradable permit regimes on the likelihood that industry would be willing to introduce some form of voluntary agreement or commitment on pollution abatement. Here, it is quite likely that the presence of an existing tradable permit mechanism might facilitate the operation of a subsequent voluntary agreement, by providing a straightforward basis for defining individual firm commitments under the agreement, and for monitoring the compliance of participants. The voluntary agreement might take the form of an agreement to extinguish, on a voluntary basis, some proportion of the initial tradable permit allocation to each firm (in effect “donating” a proportion of each firm’s permits to the industry agreement); alternatively, it could require polluters to hold permits to cover their actual emissions, plus some percentage surcharge. The efficiency properties of these two arrangements would be similar, but their distributional effects, in terms of the burden imposed on individual firms, would obviously differ.

The second case concerns the implications of subsequent introduction of a tradable permit system into an industry where a voluntary agreement on pollution abatement is already in operation. Since such agreements are often made as part of an explicit or implicit deal with government to forego certain threatened environmental policy measures, any subsequent innovation in government policy might be liable to disturb the basis on which the voluntary agreement had been reached (*i.e.* the “implicit contract” between government and industry). However, to the extent that tradable permits impose lower burdens on firms as a group than equivalent regulation, and, if distributed free (*e.g.* through grandfathering) also impose lower burdens than equivalent environmental taxes or charges, it is possible that introduction of tradable permits might be able to tighten “official” environmental policy without the same risk that the voluntary agreement would thereby be undermined.

Private sector tradable permits?

Perhaps the more intriguing set of issues concern the possibility of tradable permit mechanisms being employed as a private sector initiative, to cut through some of the complexities of determining a cost-effective distribution of pollution abatement using negotiations between individual firms.

In the absence of any such arrangement, it would be perhaps even more likely that an industry association negotiating over individual firms’ abatement commitments to the agreed industry total would resort to uniform rules (such as rules of the form “each abate $\times$ per cent”, or “each reduce emissions by 5 tonnes”) than that government agencies would impose rules of this form. In the case of both government regulation and private sector agreement, such uniform rules would fail to achieve the cost-

minimising pattern of abatement, and would thereby impose excessive abatement costs on the industry as a whole. However, in both cases, there may be a major obstacle to moving towards a better (cost-minimising) distribution of abatement obligations, because the necessary information on the abatement costs of individual firms may be difficult to obtain. Indeed, it is probable that firms may be less concerned to conceal this information from a government regulator, whose only interest is in using it for the purposes of environmental regulation, than to conceal it from fellow firms, which may be able to exploit it in the process of competition.

An industry agreement which allowed emissions trading between participating firms would be able to begin with a relatively straightforward allocation of initial abatement commitments, perhaps according to a simple uniform percentage rule, without sacrificing the potential for a cost-minimising distribution of abatement between the participating firms. Offset deals could be negotiated between individual firms, allowing them to exploit the gains from shifting abatement to lower abatement cost firms. Alternatively, the industry association could distribute permits, and allow these to be traded between participating firms. Indeed, it would be possible to devise a scheme for wholly-anonymous trading, so that the identities of buying and selling firms were only known to an agent or broker, so that the commercially-sensitive information (that might otherwise be revealed by the offer to buy or sell permits) would not be revealed to competitors within the industry.

In assessing the feasibility of grafting tradable permits onto voluntary agreements in this way, it is necessary to bear in mind the circumstances under which voluntary agreements may arise, and the conditions under which a voluntary agreement can be successfully negotiated. One aspect of this, as noted above, are the circumstances which create a collective interest in successful conclusion of an agreement – the motivation for the industry or group of firms to undertake costly abatement measures which are not mandated by legislation. Generally this has to do with the selection of the lesser of two evils – the industry “volunteers” to undertake pollution control measures in order to head off the imposition of equivalent (or more stringent) requirements through legislation. Much of the interest in voluntary agreements in European environmental policy seems to have been sparked by proposals for environmental taxes on energy, and by proposals for onerous legislation on packaging.

However, a second, and equally important, set of conditions for the successful achievement of a voluntary agreement or commitment to reduce industry pollution has to do with the requirements for agreement amongst the firms in the industry. Individual firms have to perceive benefits from undertaking costly abatement, if the industry as a whole is to be able to deliver pollution control through such an agreement. Unlike the regulatory agency, whose requirements for emission reductions can be backed by the force of law, an industry association or other group of firms can only reduce pollution if the participating firms agree to do so. Free-riding by individual firms is obviously a problem. So, too, is asymmetry between firms in the costs and benefits of the collective action. Any system of collective voluntary agreement will need to distribute the costs and benefits of the voluntary agreement sufficiently equally that individual firms are not led to defect from the agreement.

In considering whether a voluntary agreement might then be augmented by some form of tradable permit mechanism for the distribution of abatement between firms, key issues are going to be:

- the expected total gain from allowing trading;
- the expected distribution of these gains between the participating firms; and
- the process of industry decision-making.

Allowing emissions trading between firms participating in the voluntary agreement will confer benefits on those firms which participate in trading. Those that sell some of their emission rights will benefit in terms of the revenue from the sale, while those that purchase the right to additional emissions will gain to the extent that this costs them less than the pollution abatement they would otherwise have to make. One possible problem concerns the attitude of firms that do not participate in trading either as a buyer or seller. Such firms may not be indifferent to the decision to institute a tradable permit regime, since they compete in the product market with firms which *do* make gains from trading. They may therefore perceive the cost savings that their competitors make through emissions trading as undesirable, since they may shift competitive advantage in favour of these competing firms.

As an example, consider the case where the main effect of emissions trading is to give an opportunity to some of the most advanced firms in the industry to exploit their low marginal costs, both of abatement and production. If the majority of firms in the industry are not in this position, and if the industry decisions are taken on the basis of a simple majority vote, it is quite possible that the industry would prefer a voluntary agreement based on equal-abatement per unit of output, rather than the introduction of emissions trading.

As another example, consider the case where the main effect of including emissions trading within a voluntary agreement is to allow firms with old technology to avoid particularly-burdensome abatement requirements. In doing so, it may slow the pace of market adjustment and restructuring, by keeping these older firms in business longer than if they had faced the same pollution abatement requirements as newer firms in the industry. The relative weight of old and new firms in the industry's decisions will then affect the attitude that is taken to the possibility of including provision for trading within the voluntary agreement.

Conclusions

Various aspects of the “compatibility” between tradable permits and other instruments may be an important factor in successful policy innovation. The introduction of tradable permits will often take place in the form of additions and modifications to existing systems of regulation. The chances of successful policy innovation are therefore likely to be greatly enhanced if some aspects of existing “command-and-control” regulation can be used as the starting point for a tradable permit regime. If the existing system of regulation takes the form of emissions limits for individual polluters, it may be a small step to allow firms to trade emissions within these limits. The tradable commodity is defined by the emission limits, and the enforcement operations needed after trades take place can build on existing mechanisms for the enforcement of firms' compliance with the regulatory limits on emissions. In contrast, if the existing system of regulation takes a different form (such as requiring that polluters employ particular production or abatement technologies), the introduction of tradable permits would be a much larger step beyond existing arrangements, requiring firm-by-firm emissions limits to be defined, allocated, and enforced.

If the introduction of tradable permits takes place gradually, through the incremental modification of existing regulatory regimes, it is likely that tradable permits will operate side-by-side with other, “conventional” instruments (at least for some period). Gradual introduction in this way may be necessary in order to maintain confidence in the system of environmental regulation. However, in the longer-run, once the effectiveness of the tradable permit system has been established, the question needs to be asked whether it is efficient to continue to employ multiple policy instruments in parallel. Unnecessary use of more than one instrument is liable to be wasteful, incurring excessive administrative costs, and some of the gains of flexibility with market mechanisms might not be fully achieved if they are permanently combined with other regulatory instruments. Are there any circumstances where permanent parallel operation of tradable permits and other instruments might be desirable?

This paper has suggested some situations where it might indeed be inefficient to rely on a single instrument, and where tradable permits and other instruments might be compatible, in a long-run sense, within the overall policy portfolio. One possibility is that one instrument may be used to offset some limitation of the other (*e.g.* a regulatory limit might be used to prevent excessive geographical concentration of emissions through trading, or to ensure that “insensitive” firms undertake at least some pollution abatement). Another possibility is that sole reliance on a single instrument may increase the costs of enforcement (by raising the gains to firms from misrepresentation and other forms of deliberate non-compliance). A further important group of such cases involve situations where the costs per emissions source of emissions measurement make the use of tradable permits worthwhile for large emissions sources only. For smaller sources, the efficiency savings from the use of a flexible market mechanism may be insufficient to warrant incurring these measurement costs. The parallel use of direct regulation for smaller firms with tradable permits for larger firms may then be the most cost-effective instrument choice.

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Chapter XI

TRADABLE PERMIT SYSTEMS AND INDUSTRIAL COMPETITIVENESS: A REVIEW OF ISSUES AND EVIDENCE

by

Nick Johnstone

Environmental Economics Programme
International Institute for Environment and Development
London, UK

Introduction

Tradable permits are no longer quite as novel as they once seemed. Many OECD countries either have some in place, or are discussing their introduction in the near future. At the international level, they are being discussed as a means to address the problem of climate change. In general, their relative “efficiency” as a means of internalising environmental externalities is no longer questioned (although problems of implementation are considerable in some cases). However, some of the more specific economic effects of tradable permits still need to be analysed.

In this paper, an attempt is made to draw out some of the “competitiveness” implications of a tradable permit system. Any paper which tries to evaluate the effects of alternative policy instruments on competitiveness starts with the thorny problem of defining competitiveness. According to a recent OECD (1996) document, the keys to increasing competitiveness are: the ability to minimise costs, to capture market share and to increase the technical efficiency of production.¹ These are the indicators which are used in the discussion which follows.

Despite being concerned with the competitiveness effects of environmental policies, the paper does not draw on this literature directly. It is already well-developed and does not require yet another contribution. Instead, this paper focuses on the literature concerned explicitly with tradable permit systems. Since the United States has been most active in the use of tradable permit systems, the study draws primarily upon American evidence. Six programmes are used to cast light on the potential competitiveness effects: the EPA’s sulphur dioxide allowance trading programme; Los Angeles’ programme for oxides of nitrogen and sulphur; the lead trading programme for petroleum refineries; the chloro-fluorocarbon trading programme; the Fox River programme for trading in permits for biological oxygen demand; and the Dillon Reservoir phosphorous programme. This gives a relatively comprehensive overview of the main characteristics of the systems and the contexts in which they can be applied.

While none of the literature concerned with these cases addresses competitiveness issues directly, much of the evidence is relevant for issues of competitiveness. Unfortunately, much of it is quite tenuous, since experience with tradable permits is still not sufficiently broad or of sufficient duration to allow for systematic empirical analyses. While this is true of rather unambiguous indicators, such as financial costs of production and market shares, it is particularly problematic for examining the effects of tradable permit systems on overall productivity since – despite being the most important long-run determinant of competitiveness – it is also the most difficult to evaluate. As such, the study often relies upon indirect forms of evidence and insights from theoretical and simulation studies. Nonetheless, it is hoped that this gives a reasonable indication of the potential for competitiveness effects (positive and negative) to arise.

The paper argues that, since tradable permit systems are efficient environmental policy instruments, their competitiveness effects are likely to be positive. However, in distributional terms, the “devil seems to be in the details, with the effects for individual firms and sectors being largely a consequence of the precise form in which the regime is implemented. Thus, there is often a rather protracted period of negotiations which precede the introduction of tradable permit systems, with firms lobbying for systems which are beneficial to their interests. This flexibility is one of the strengths of tradable permit systems, since it allows public authorities to respond to such concerns. However, it is also potentially one of their weaknesses, since the aggregate and dynamic benefits of tradable permit systems are dependent upon their precise form of implementation. As such, while it is important that the public authorities respond to the concerns of firms, sectors, and regions about loss of competitiveness, they should be careful not to undermine the overall efficiency of the system in an effort to protect the interests of particular groups.

Following this brief introduction, the more general effects of environmental policies on competitiveness is briefly reviewed. The evidence of the potential cost savings from tradable permit systems is then examined, followed by a discussion of some of the constraints on the realisation of these savings. The main factors which determine the distribution of costs and savings among firms and sectors are then assessed, followed by a number of factors likely to be important determinants of dynamic competitiveness. The final section of the paper summarises the discussion.

Environmental policy choice and competitiveness

The empirical literature on the international competitiveness effects of environmental policy is large and expanding (see J. Adams 1997 and Jaffee *et al.* 1995 for recent reviews). In general, such studies have not found a statistically-significant negative relationship between the stringency of environmental regulations and international trade and investment patterns (Tobey 1990, Grossman and Krueger 1992, and Kalt 1988). However, there are exceptions, with some sectors and some particular trading relationships revealing limited (and ambiguous) evidence of a negative relationship (see Han and Braden, 1996; van Beers and van den Bergh, 1996; and Xing and Kolstad, 1996 for examples). Moreover, studies conducted at the national level have generally found more evidence of competitiveness effects, insofar as they are reflected in firm profitability or sectoral productivity (Gray and Shadbegian, 1995; Jorgenson and Wilcoxon, 1990; and OTA, 1994).

While these studies examine the effects of the relative stringency of environmental regulations on competitiveness (usually using abatement costs declared by firms in industrial surveys as the explanatory variable), there are few studies which actually look at the competitiveness effects of different types of regulations. This is significant, since the economic effects of a regulation may be more a function of its nature in an institutional sense, than its stringency in terms of emissions abated. Intuitively, one would suppose that the effects of market-based instruments such as permits and taxes are likely to be less than more direct forms of regulation due to their relatively greater efficiency (*i.e.* a given reduction in emissions can be achieved at lower economic cost). However, as discussed below, how this is reflected at the level of the firm depends upon the precise nature of the instrument. For instance, since most direct regulations (*i.e.* mandated emission reductions or abatement technology) effectively zero-charge all residual pollutants, they may have lower short-run firm-level costs than market-based instruments (*i.e.* permits, charges or taxes), which charge for all pollutants.

However, there are two theoretical studies which have looked at these issues in more detail. Firstly, Markusen (1997) compares the effects on plant location decisions of regulations which tend to impact on capital costs and those which tend to impact on marginal costs. He finds that the economic effects of regulations which impact on fixed costs tend to be reflected in terms of firm entry and exit, while the effects of regulations which impact on variable costs tend to be reflected principally in terms of reduced levels of production per firm. This is significant since many direct regulations mandate specific technologies with considerable fixed costs. Conversely, depending upon the optimal strategy adopted by firms, MBIs may affect variable costs to a much greater extent. For instance, prior to the Acid Rain programme, electricity generators had to introduce capital-intensive scrubbers, but afterward, they could also buy permits, mix different fuels, or buy lower-sulphur coal, all of which affect variable costs.

Sartzetakis and Constantatos (1995) use a two-country model to compare the trade effects of introducing a policy based on mandated emission reductions, relative to one based on a tradable permit system. Not surprisingly, they find that the market share of firms in the country which mandates a tradable permit system rise relative to the market share of firms in the country which mandates emission reductions. This difference becomes greater with the diversity in the abatement technologies used. Indeed, it is possible that the increase in market share is sufficient for the absolute level of emissions to rise in the country with the tradable permit system.

Cost savings from tradable permit systems

In order to determine the competitiveness effects of a tradable permit system, it is first necessary to evaluate the cost effects associated with their introduction. A relatively more efficient environmental policy – one which achieves its environmental objective at lower economic cost – may not necessarily have less adverse effects in terms of the competitiveness of manufacturing firms. As discussed in the following section, depending on the precise form of implementation, many firms (or even the manufacturing sector in general) may face higher costs or lose market share, following the introduction of the tradable permit system. However, depending upon how they are designed, the flexibility of tradable permit systems does at least allow for the possibility of environmental objectives to be realised without jeopardising firms' competitiveness. As such, it is important to look at the aggregate cost savings arising from a tradable permit system.

Determining the relevant baseline is a key, when evaluating cost savings, since tradable permit systems have never been introduced as the initial form of regulation for a pollutant and/or sector. Rather, they have almost always either supplanted or co-existed with direct forms regulation, which have usually either taken the form of percentage emission reduction requirements or their technology-based equivalents. Thus, rather than trying to evaluate the cost effects of tradable permit systems relative to a base case in which no regulations are in place, it is usually more pertinent to evaluate their cost effects relative to a base case in which other regulations are already in place.

A number of authors have tried to compare the cost effects of tradable permit systems and direct regulations by estimating the ratio of abatement costs under the two systems. The most widely cited figures are those from Tietenberg (1985), who collates the results of a number of simulation studies. These and results from other studies are summarised in Table 1 below. The ratios range from 1.07 for sulphates in Los Angeles, to 22.0 for particulates in the Lower Delaware Valley. However, it should be emphasised that while most of the estimated costs of direct regulations are derived from actual costs, the costs for tradable permit systems are usually based on simulations of their implementation, assuming away many of the problems associated with implementation.

More recently, a number of studies have attempted to compare costs for direct regulations and tradable permit systems for existing programmes. In particular, a large number of studies have attempted to evaluate the savings from the US SO₂ allowance trading programme. Four of these are reviewed in Bohi and Burtaw (1997), and the results are summarised in Table 2. While there is considerable uncertainty about the precise magnitude of the savings, it is clear that savings are significant. Moreover, the estimates of savings have risen with time.

Studies of other programmes also reveal considerable savings. For instance, the South Coast Air Quality Management District estimated that implementation of the technology-based 1991 Air Quality Management Programme for the Los Angeles area would have cost \$138.7 million annually for the period 1994-1999 had it been implemented, but the permit-based RECLAIM programme only cost \$80.8 million (figures cited in Polesesky, 1995). An independent study estimated similar annual cost savings of \$57.2 million relative to the previous regime (Johnson and Pikelney, 1996). The EPA estimated that the annual costs savings from the refinery lead trading programme was approximately \$250 million (cited in Stavins, 1998).²

What factors lie behind these results? The most important factor determining the extent of cost savings is the degree of abatement cost heterogeneity prior to the introduction of the permit-based system,

Table 1. **Estimated Ratio of Costs of Direct Regulations and Tradable Permits**

study	Pollutants	Area	Ratio
tkinson and Lewis (1974) ¹	Particulates	St. Louis	6.00
oach <i>et al.</i> (1981) ¹	Sulphur Dioxide	Utah	4.25
ahn and Noll (1982) ¹	Sulphates	Los Angeles	1.07
rupnick (1983) ¹	Nitrogen Dioxide	Baltimore	5.96
eskin <i>et al.</i> (1983) ¹	Nitrogen Dioxide	Chicago	14.40
cGartland (1984) ¹	Particulates	Baltimore	4.18
poffard (1984) ¹	Sulphur Dioxide	Lower Delaware Valley	1.79
poffard (1984) ¹	Particulates	Lower Delaware Valley	22.00
aloney and Yandle (1984) ¹	Hydrocarbons	All US Dupont Plants	4.15
almer <i>et al.</i> (1980) ¹	CFCs	United States	1.96
tkinson (1983)	Sulphur Dioxide	Cleveland	1.50
ohnson and Pekelney (1996) ²	NO _x and SO _x	Los Angeles	1.35-8.33
yon (1982)	Phosphorous	Lake Michigan	1.24
yon (1982)	BOD	Wilamette River	1.16

- . Cited in Tietenberg (1985).
- . Results depend upon year.

Table 2. **Annual Costs for Implementation of Phase II of SO₂ Tradable Permits**

\$US billion

study	Direct Regulation	Full Trading
CF Incorporated (1989) ¹		2.7-4.0
an Horn Consulting (1993) ¹	5.1	2.2
overnment Accounting Office (1994) ¹	4.3	1.4
urtaw <i>et al.</i> (1997) ¹	2.5	1.0
arlson <i>et al.</i> (1998) ²		1.4

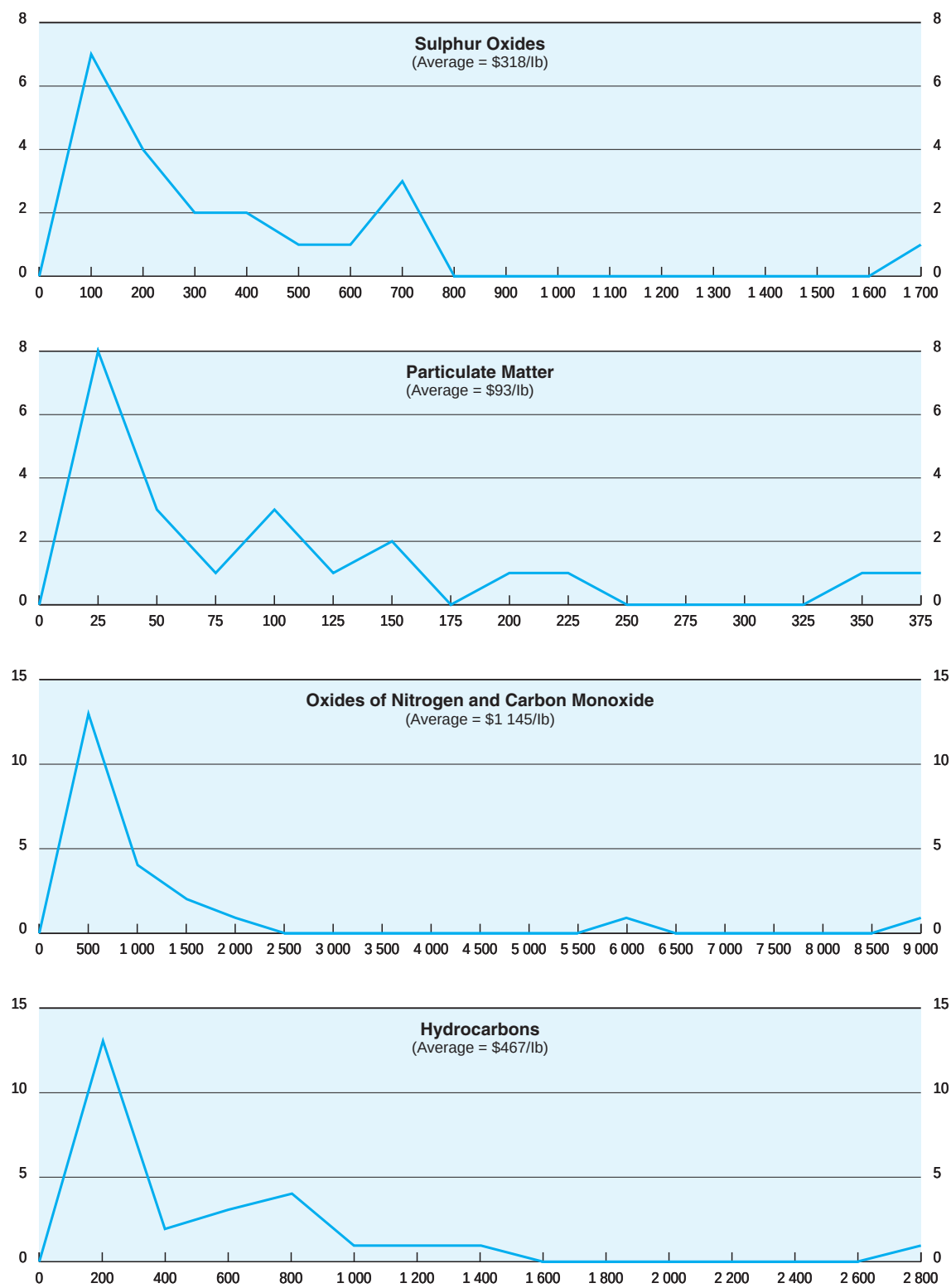
- . Cited in Bohi and Burtaw (1997).
- . Cited in Burtaw (1998).

since this will determine potential efficiency gains. If the existing methods of regulation are very inefficient (*i.e.* marginal abatement costs differ widely for emissions with equivalent damages), efficiency gains are likely to be very high.

For instance, prior to the Acid Rain programme, there was also thought considerable variation in SO₂ emission abatement costs from the electricity supply industry. This was largely a function of the age of plants, their utilisation rates, and their proximity to low-sulphur coal deposits [Stavins (1998) and Schmalensee *et al.* (1998)]. There were also thought to be large variation in abatement costs for lead refineries prior to the introduction of the trading regime, with older, more costly plants (in terms of abatement) located in the East [Hahn and Hester (1989) and OECD (1992)]. And finally, in a study of the RECLAIM programme, costs of NO_x reduction for the 53 electric utility boilers likely to be affected by the programme ranged from \$3 000/ton to \$130 000/ton (OECD, 1994).³

More generally, using abatement cost estimates from Hartman *et al.* (1994) based on US Census Bureau data, it is possible to draw a more general picture of the degree of heterogeneity (and thus inefficiency) which existed in US pollution control regimes in the mid-1980s. Figure 1 shows the frequency distribution of marginal abatement costs in 1983-1985 for four classes of air pollutants (particulates, SO_x/NO_x, CO, hydrocarbons) from the Hettige *et al.* (1995) study. On this basis, it is clear that there was considerable inefficiency in the US air pollution control regime in the mid-1980s. Given that there has been only limited substitution of direct regulations by more efficient (*i.e.* MAC-equalising) regimes, it is quite likely that heterogeneity of a comparable order still exists. Moreover, if there is considerable variation in MACs within individual sectors these figures will underestimate the potential benefits.⁴

Figure 1. Frequency Distribution of Marginal Abatement Costs by Sector



Constraints on the realisation of cost savings

There are clearly significant efficiency gains to be realised from the introduction of a tradable permit system instead of direct regulations, although individual firms may not realise these benefits themselves. Moreover, in practice, there may be significant constraints on the realisation of such benefits. In particular, there are three factors which may restrict the extent of savings from the introduction of a tradable permit system: regulatory constraints on its application; transaction costs in implementation; and the existence of power in the market for permits. All of these prevent abatement cost equalisation and/or affect the equilibrium permit price.

Regulatory constraints

As noted above, tradable permit systems have usually complemented, rather than replaced, traditional direct regulations. In many cases, this has restricted the potential cost savings from the trading programmes. For instance, under the lead trading programme, refiners in California could not use that programme to exceed the state standards (Hahn and Hester, 1989). Since California standards were higher than those standards upon which the EPA allocation was based, this effectively meant that a number of high abatement cost refineries were not extensively involved in trading, reducing the overall efficiency of the system. More specifically, a number of California refineries which might have benefited from purchasing permits (along with their trading partners benefiting from the sale) could not do so. Similarly, in the CFC trading programme producers were not able to increase their emissions by more than 10 per cent of their allocation (15 per cent after 1998) (Tietenberg, 1998). If abatement costs differ widely, this would have severely constrained the potential for efficiency savings to be realised, since there would not be full equalisation of abatement costs.

In addition, many of the systems impose a variety of restrictions based on the location of the emitting source, due to the spatial characteristics of damages from the pollutants. These may take the case of specified trading ratios between regions with different pollution concentrations (as in Los Angeles' RECLAIM programme) or explicit constraints on certain trades between regions (as in the Fox River BOD programme). It is difficult to gauge the effects that such restrictions have on the overall efficiency of the programme, since they are designed to deal with the differentiation in the effects of emission pollutants depending upon their location of emission, and thus require estimates of environmental benefits as well as control costs.

If they are administratively straightforward to implement, do not constrain firms trading opportunities excessively, and approximate the differentiated environmental benefits of abatement, then welfare is likely to improve relative to a system which does not impose such constraints. However, this may be the exception. One of the main reasons behind the failure of the Fox River BOD programme was the imposition of spatial trading restrictions in a market which was already very thin (see also Chapters 6 and 10). Thus, it is significant that the most recent (and perhaps most successful) case – the SO₂ programme – ignored spatial issues altogether, despite the spatial differentiation in damages depending upon the location of emission.

Unquestionably, the most important regulatory constraints on the realisation of efficiency gains from tradable permit systems relate to the treatment of new firms. Under the US regulatory system, there has always been a strong bias against new firms, particularly those that wish to establish themselves in non-attainment zones. While the introduction of tradable permit systems could have been an opportunity to remove this bias, in many cases, they have actually reinforced it by keeping regulatory requirements in place and distributing permits freely amongst existing emitters. Thus, new firms must not only meet more stringent regulatory requirements than existing firms, but they must also purchase permits which are distributed freely to existing firms.⁵

For instance, under Los Angeles' Air Quality Management Programme, new firms in non-attainment zones must invest in the more stringent "Best Available Control Technology" and in "extreme" non-attainment zones in technologies which generate the "Lowest Achievable Emission Rate". These remain in place under RECLAIM (Polesetsky, 1995). Moreover, since the definition of "new" is often retrospective, in some cases, this can affect a large number of firms and plants which are already operational. For

instance, in the case of the lead trading programme, all “new” sources were those established after the mid-1970s (Hahn and Hester, 1989). This has important implications for the overall efficiency of the programme, since a large proportion of efficiency-improving trades cannot be realised if the “new source bias” of pre-existing regulations is retained.⁶ “New” firms with high marginal abatement costs will not be allowed to buy a sufficient number of permits from “existing” firms with much lower abatement costs, as a way of equalising abatement costs. In effect, the existence of regulatory requirements places a ceiling on permit holdings for “new” firms, which is likely to be at less than the optimal level.

In some cases, regulations which are introduced by agencies other than the environmental agency may have even more important effects on constraining the potential efficiency gains from a TP system.⁷ For instance, there have been concerns that state public utility commissions – through their roles as regulators of the electricity supply industry – might erect barriers to the full implementation of the SO₂ allowance trading (Stavins, 1998). In particular, states with large high-sulphur coal deposits might encourage the use of scrubbers relative to imported low-sulphur coal in order to protect local jobs in the coal mining industry, even though the latter option would be more economically-efficient.

The potential for public utility commissions and other regulatory agencies to restrict the efficient implementation of a tradable permit programme is explored in a theoretical study by Bohi and Burtaw (1992). They cite a number of potential avenues through which the regulator can affect the compliance strategies adopted by firms: the determination of the allowable rate of return on alternative assets; the inclusion of some expenditures (but not others) in the ratebase for pricing regulation; allowances made for depreciation rates on capital equipment and other factors. These factors will determine whether firms invest in scrubbers, buy low-sulphur coal or purchase SO₂ allowances. However, one of the few studies to look at this issue (Bailey, 1996) has not found any evidence that the PUCs have discouraged the efficient implementation of the SO₂ allowance trading programme by distorting incentives for the firm.

Transaction costs

Another important factor determining the magnitude of aggregate cost savings from the implementation of a tradable permit regime in place of direct regulations is the extent of transaction costs in permit trading. If transaction costs are high, many of the potential benefits of trading will be lost, not only because firms will bear expenditures for unproductive activities, but also because the equilibrium level of abatement will be affected [Stavins (1995) and Xepapadeas (1997)].

Stavins (1995) splits transaction costs into three main forms: search costs, bargaining costs and monitoring costs. Search costs are those costs borne by the firm in order to find a buyer (seller) for the permits it wishes to sell (buy). Bargaining costs are incurred in the course of agreeing upon a price for the permit. Monitoring costs arise from the need to ensure that actual emissions are in accordance with permit holdings.⁸ While the first two costs (search and bargaining) are ultimately borne by the firm, they may be paid for indirectly by contracting brokerage services. The third form of cost (monitoring) may either be borne by the firm and/or the implementing agency. For instance, as part of the SO₂ programme firms had to purchase monitoring equipment and the government had to analyse and collate the results.

How high are the transaction costs? Estimates of transaction costs for the EPA's Emissions Trading Programme (a precursor to later trading programmes) were as high as 10-30 per cent of the total costs (see Koutstaal, 1997). However, this programme was deeply flawed in senses which were likely to have increased transaction costs, relative to the more fully-developed systems which followed. Klaassen and Nentjes (1995) cite a figure of 5 per cent for brokerage fees for the SO₂ allowance trading programme. The CFC and lead trading programmes are also thought to have had relatively low transaction costs [Stavins (1995); Hahn and McGartland (1989)]. Conversely, the Fox River BOD programme appears to have had very high transaction costs, partly explaining the limited trading activity (Hahn, 1989).

Transaction costs are likely to be determined in part by the characteristics of the affected sectors. In the case of lead trading, the firms involved in the market (refineries) had a long history of co-operating and even trading. Thus, search and bargaining costs were probably quite low. However, for other pollutants – such as the proposed inclusion of ROGs in the RECLAIM programme – the diffuse nature of sources may increase such costs. Firms will have to seek out other firms with whom they have not

previously had commercial or institutional relationships in order to complete a trade. In a related (and perhaps conflicting) vein, it would seem that market depth is important. Koutstaal (1997) argues that, in most cases, there is likely to be a negative relationship between the size of the market and transaction costs.

The means by which the regime itself is implemented will also affect transaction costs. For instance, it is widely assumed that grandfathered permit regimes have higher transaction costs than auctioned permit regimes. This is because trades are sequential and bilateral, rather than instantaneous and multi-lateral [Atkinson and Tietenberg (1991) and Polesetsky (1995)]. Buyers (sellers) do not need to seek out other firms who happen to wish to sell (buy) permits at a particular point in time. The inclusion of a small primary auction in the SO₂ trading programme was partly motivated by fears of the adverse effects of transaction costs in the secondary market. Thus, the primary market was introduced in order to force efficiency into the secondary market. However, it would appear that it is prices in the secondary market which have been driving prices in the primary market and not *vice versa* (Joskow and Schmalensee, 1996).

Market power

The potential for market power has long been recognised as a disadvantage of tradable permits, relative to other MBIs such as taxes [see Hahn (1984) for a discussion]. In the presence of market power, some firms behave as price-setters, whether as sellers (monopolists) or buyers (monopsonists). This has important efficiency effects, with many of the potential benefits of a tradable permit system being lost [Koutstaal (1997); Misiolek and Elder (1989); Xepapadeas (1997); and Hahn (1984)].

When is market power most likely to occur? Most importantly, it is dependent upon the degree of concentration of permit markets. If no firms are important players in the market (whether as buyers or sellers), there is little likelihood of there being any market power. Indirect evidence of the potential extent of market power for permit markets for different air pollutants is presented in Table 3 which gives (very rough) estimates of the degree of concentration of emission sources in the US. The figures are derived on the basis of World Bank IPPS data (Hettige *et al.*, 1995) on emissions by sector (4-digit ISIC) and UNIDO sectoral data on the number of firms in each sector.⁹

Using this approximation, it would appear that the greatest potential for market power would be in a market for permits for particulate matter, followed by SO₂ and CO. Indeed, since the effects of the former are quite localised any programme for particulate permits is likely to be local in nature. As such, it would probably only deal with a sub-set of these firms, increasing the potential for market power to be exercised. Ideally, the data should be disaggregated spatially, to reflect the spatial “scope” of the pollutants.

This is also true of water pollutants since the environmental problem is localised and thus the permit system is likely to be spatially restricted [Koutstaal (1997); Misiolek and Elder (1989); Lyon (1982); and Hahn (1989) review some cases; see also Chapter 4]. Indeed, in the Fox River BOD case, the potential for

Table 3. **Concentration of Air Pollution Emission Sources in the United States**

Approx. Number of Firms Accounting for 50% of Total Emissions From Manufacturing

pollutant	Number of Firms	Sectors
Total Suspended Solids	5 105	Cement, lime, and plaster; pulp, paper, and paperboard; petroleum refineries; sugar factories and refineries; oils and fats; structural clay products; spinning, weaving, and finishing textiles; iron and steel
Particulate Matter	917	Cement, lime, and plaster; pulp, paper, and paperboard; oils and fats
Volatile Organic Compounds	5 106	Petroleum refineries; distilled spirits; synthetic resins, plastics materials, and manmade fibres; pulp, paper, and paperboard; tires and tubes; spinning, weaving, and finishing textiles; industrial chemicals except fertiliser; sugar factories and refineries; oils and fats; iron and steel
Carbon Monoxide	2 579	Pulp, paper, and paperboard; petroleum refineries; iron and steel
Sulphur Dioxide	1 799	Petroleum refineries; pulp, paper, and paperboard; cement, lime, and plaster; spinning, weaving, and finishing textiles; non-ferrous metals

market power is even greater than first appears, since the market is sub-divided to reflect “clusters” where pollution is most problematic. At two points, only six and seven firms are able to trade amongst each other [see Hahn (1989)]. Thus, in a simulation of the programme, 90 per cent of pollution rights are purchased by 3 firms, and 44 per cent by just one firm. Other results for hypothetical water pollution permit programmes in the US (phosphorous in Lake Michigan) and the UK (BOD in the Forth Estuary) are similar, if not even more striking.

The potential for market power is also a function of the sectoral “scope” of a permit market. If few sectors are included, there may be few firms involved in the market. For instance, in the SO₂ programme only electricity generators were included, and not other sources. As a consequence, permit trades were dominated by very few firms. In 1995 and 1996, over 75 per cent of all inter-utility purchases of permits were bought by one firm, Illinois Power (Ellerman *et al.*, 1996). While this would indicate the presence of considerable market power, this is tempered by the fact that intra-utility trade was 3-4 times the level of inter-utility trade. Although not directly involved in the market in an arms’ length sense, such trades do have an effect on the equilibrium price, and thus, on the ability of a firm to set prices.

And finally, the degree of concentration is also affected by the coverage of different “classes” of emitters. Thus, in the discussions concerning the introduction of the CFC trading programme, it was feared there was significant potential for market power to be exercised, since production of CFCs was so concentrated in the US. In the end, permits were issued to 27 firms. The problem could have been mitigated through the inclusion of downstream users who were, after all, the ultimate emitters of CFCs [Hahn and McGartland (1989) and Tietenberg (1998)].¹⁰

There is, regrettably, little empirical evidence of the extent to which market power is a problem in existing tradable permit systems. However, firms themselves clearly perceive the danger. For instance, in the case of SO₂ permits, utilities in fast-growing States advocated that at least some of the permits be issued through an auction – even though this would appear to have adverse effects for them in financial terms – because of concerns that utilities in other States would exploit the market to their disadvantage [Joskow and Schmalensee (1996)]. This fear was also behind the concerns of the small firms advocating an auctioned permit allocation for the RECLAIM system, even though it would not appear to be in their financial interest (Polesetsky, 1995).

Distribution of costs and savings

The preceding sections have looked at some of the determinants of aggregate costs and savings from a tradable permit system. However, individual sectors and firms are rather less interested in the aggregate economic benefits of a tradable permit system, than they are in how it will impact upon them directly. For this reason, it is important to look at the determinants of the distribution of costs and savings amongst firms and sectors.

As with any policy measure, the relative burden of tradable permits for firms will depend on the extent to which they can pass on the costs of compliance to customers. This is largely a function of the demand elasticities for the firms’ output. In cases where demand is highly inelastic, the firm will be able to pass on most of the costs, whereas the opposite will be true if demand is elastic. Analogously, if supply is relatively elastic, the price will rise by more (and output will fall) by more than when it is more inelastic. And finally, the cost effects of tradable permits will be determined by the extent to which firms are able to substitute away from the use of permits in favour of other inputs. However, all of these factors are true of any environmental policy change and do not warrant further discussion in the context of this study.¹¹ This section concentrates only on those characteristics which are specific to tradable permit systems.

Permit allocation mechanisms

Unquestionably, the most important determinant of whether firms benefit from the introduction of a tradable permit system (and how much they benefit), relative to other environmental policy instruments, is the permit allocation mechanism applied. In particular, there is likely to be a large difference in financial expenditures by firms, depending on whether or not permits have been allocated through a

“grandfather” scheme, or through an auction. In the latter case, firms in aggregate will have to pay for abatement control costs, as well as the costs of permits themselves. In the former case, firms as a whole will only pay abatement costs since permit expenditures are between firms, and not between firms and the government.

In terms of financial expenditures auctioned permits approximate emission taxes, since in both cases, the firm pays for residual emissions, while grandfathered permits approximate “efficient” (*i.e.* equal abatement cost) direct regulations, since firms (in aggregate) do not pay for residual emissions (Table 4). However, relative to most direct forms of regulation in which abatement costs are not equalised, all firms will tend to benefit from the reduced costs associated with more efficient abatement, irrespective whether the permits are auctioned or grandfathered. Moreover, the effects for individual firms will vary. Firms with high emissions (and thus, high allocations) have relatively more to gain from the introduction of a grandfathered tradable permit system than they do from either a tax or auctioned permits. Depending on the relative proportion of permit expenditures in total compliance costs, many firms may be worse off relative to direct regulation under the auction scheme in strict financial terms.

Table 4. **Cost Effects of Alternative Environmental Policy Instruments**

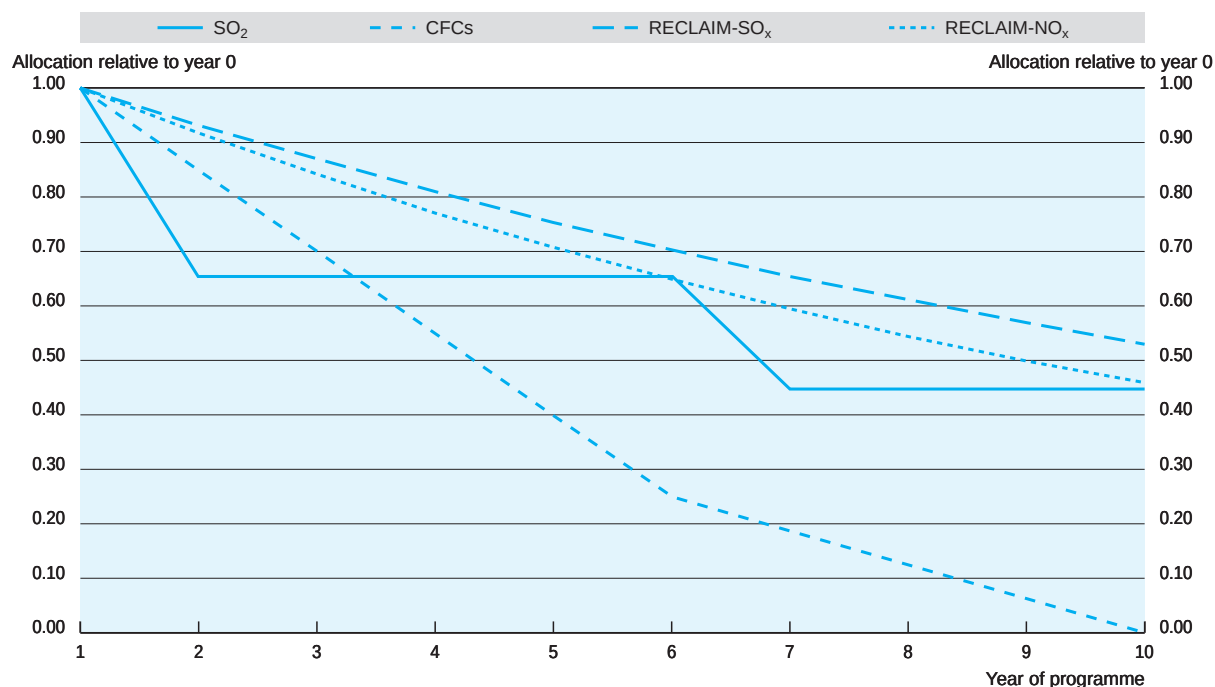
	Grandfathered Permits	Auctioned Permits	Emissions Tax	Technology-Based Regulation
ay for Residual Emissions	No (in aggregate)	Yes	Yes	No
batement Cost Efficiency Gains	Yes	Yes	Yes	No (in most cases)

The key question is whether or not affected industries will gain more from efficiency benefits than they lose in terms of rents. In general, the ratio of permit expenditures to abatement costs will be lower with high levels of abatement. As such, the advantages of replacing direct regulations with a grandfathered tradable permit system, rather than an auctioned tradable permit system fall as the environmental stringency of the programme rises. Figure 2 gives an overview of the approximate emissions reductions (relative to the year prior to the introduction) of the programme from different tradable permit systems.

Some simulation studies have been done on the relative control costs of the two types of permit allocation. For instance, in his study of phosphorous control in Lake Michigan Lyon (1982) found that (overall) firms were better off under a grandfathered system (\$4 880/day) than under a command-and-control system (\$3 930/day), but considerably worse off under an auction-based system (\$9 860/day). Expenditures on permits exceeded efficiency gains by almost six-to-one. In the same study, it was found that permit expenditures for BOD control were 30 per cent greater than the reduction in abatement costs. Tietenberg (1998) reviews a number of other simulation studies which also find that permit expenditures are high enough to outweigh efficiency gains. Similarly, in their study of air quality in St. Louis, Atkinson and Tietenberg (1984) find that 20 of 27 firms will be at least \$25/day worse-off¹² in the event that an auction allocation mechanism is introduced. This compares with only two firms if a grandfathered system is used.¹³ However, the simulation assumes a 12 per cent decrease in emissions relative to the benchmark case, and thus, part of the financial burdens are a consequence of tighter environmental regulations, and not due to a shift in the regulatory regime *per se*.

Thus far, it has been assumed that there is really only one type of grandfathered permit system – *i.e.* one which is based on actual emissions prior to the introduction of the programme. However, few of the tradable permit systems actually introduced involve pure grandfathered allocations. For instance, under the SO₂ allowance trading programme permits were initially allocated on the basis of energy output and a constant emission factor (2.5 lbs sulphur/million BTUs), irrespective of the actual level of emissions from each plant.¹⁴ Relative to an emission-based grandfathered system, this favours the low SO₂

Figure 2. Emission Reductions in Tradable Permit Programmes



emitters. Similarly, in the lead trading programme, permits were allocated on the basis of refinery gasoline production and an assumed lead content (1.1 grams/gallon). The firms which produce relatively low-lead gasoline will benefit. More significantly, those which produce relatively more unleaded gasoline will benefit.

Alternatively, other programmes allocate permits on the basis of assumed compliance with existing (or prospective) direct forms of regulation. For instance under RECLAIM, firms are granted permits according to the firm's production level and plant-specific emission factor, based on the level of emissions they would have if they abided by the technology-based rules in place when the TP programme was instituted [Polesetsky (1995)]. Moreover, this is done on a prospective basis, with the emission factors applied for the determination of future permit allocations calculated on the basis of the proposed AQMP, which was never implemented.¹⁵ Relative to a pure grandfathered system, this approach favours those firms which are in accordance with existing regulations when the system is introduced. This prevents firms from benefiting from non-compliance – a situation which could theoretically arise with a pure grandfathered system. In future years, the system will tend to favour firms which would have been subject to less stringent regulations under the AQMP.

Initial abatement costs

The distribution of marginal air pollution abatement costs across sectors was reviewed earlier, and it was found that in the US, there was considerable heterogeneity. This is largely a reflection of the existing variation in the relative stringency of direct forms of regulation and is also likely to be true of other countries. At the aggregate level, this is significant, since it will determine the cost savings arising from the implementation of economic instruments, including tradable permits. However, it is also significant at the level of the firm, since precisely where a particular firm or sector is located on this distribution prior to the introduction of the TP programme will go a long way toward explaining how much (if at all) it is likely to benefit.

In the case of grandfathered schemes, it is true that all firms will benefit, but the distribution of these benefits may vary widely. Those firms initially furthest from the post-permit equilibrium in terms of marginal abatement costs are likely to gain the most (OECD, 1994). For those firms nearest to the permit-equilibrium level of MAC, there is little to be gained from the system in terms of efficiency benefits. However, for those firms with low initial MAC, they will be able to increase their emissions, selling their grandfathered permits at a price considerably greater than the abatement costs that they will have to incur in return. This would include firms which were not previously subject to regulation, but which have been included in the programme (*i.e.* smaller firms and mobile sources). Similarly, those firms with high initial MAC will be able to buy permits at a price considerably less than their pre-regime abatement costs. This would include firms which were previously subject to very stringent direct forms of regulation which are no longer in place (*i.e.* new sources and particular sectors).

This intuition is reflected in the SO₂ allowance trading programme. It has been estimated that a large majority of the benefits (\$706 million) from the trading programme have been realised by just four firms: Central Illinois Public Service, Illinois Power Company, Duke Power and Wisconsin Electric Power (Cole, 1998). All of these were high-abatement-cost utilities which exploited the possibility of buying allowances above and beyond those which they had been grandfathered at a price lower than the abatement costs that would have otherwise been incurred.

The case with an auctioned permit regime is quite different, since firms pay for both control costs and permits. All low abatement cost firms will be made worse-off relative to technology-based regulations, but the combined cost in terms of abatement costs and permit expenditures will fall as the initial abatement cost deviates from the permit price. However, for high-abatement-cost firms, the effects are ambiguous. Some firms may benefit relative to technology-based rules and the likelihood of them doing so increases with the difference between initial abatement costs and the equilibrium permit price. Thus, as with a grandfathered allocation, those firms initially furthest from the equilibrium permit price are likely to gain the most (or lose the least).

Figure 1 illustrated just how much heterogeneity there is in terms of abatement costs in the US. In order to cast light on which specific sectors are likely to gain the most from the introduction of a tradable permit system, Table 5 lists those firms which are more than two standard deviations from the abatement-weight mean average abatement cost, based on estimates from Hartman *et al.* (1994).

Spatial issues

Like most direct forms of regulation (and unlike most emission taxes), the scope of tradable permit systems is often spatially delimited. This is significant, since firms are affected by the scope of the market for their output relative to the scope of the market for permits. If there is complete congruence between

Table 5. **Sectors with High or Low Abatement Costs**
> 2 Standard Deviations from the Emission-weighted Mean of Average Abatement Costs

articulate Matter	Sulphur Oxides	Nitrogen Oxides and Carbon Monoxide	Hydrocarbons	Lead	Hazardous Emissions
Public Goods Transp. Eqpm. Footwear Apparel Printing Textile Radio Leather Goods Vehicles Metal F Metals Refineries	Public Goods Plastic Petrol. and Coal Radio Metal Vehicles Transp. Eqpm. Rubber Drugs Machinery	Beverages Leather Pottery Vehicles	Tobacco Beverages	Food Footwear Vehicles	Comp. Eqpm. Furniture Printing Pottery

the two, the effects for the firm are strictly a function of the extent to which demand elasticities allow them to pass on any costs incurred. However, if the two have different coverage, the effects can be quite different.

For instance, in the case of RECLAIM, only firms within Los Angeles are affected. As such, the extent to which they are able to reap the savings of the regime relative to the AQMP will depend on the extent to which they compete with firms outside of Los Angeles. If they do so, to a great extent, the firms themselves will be able to realise the savings. If not, a proportion of the savings will accrue to consumers, since the savings will be reflected in prices [OECD (1994) and Johnson and Pkelney (1996)]. The same would also have held true for other cases such as pulp and paper mills in the Fox River BOD programme, who compete with firms nationally (Hahn, 1984). It must be remembered, however, that in areas with particularly acute environmental probes (*i.e.* smog in Los Angeles), even if the programme is relatively less expensive for firms than direct forms of regulation are, rather than bestowing a competitive advantage, the tradable permit system will put them at less of a competitive disadvantage, relative to firms in areas with less severe environmental problems.

In cases where the permit programme is national in scope, the international competitiveness effects of a tradable permit programme are determined by the tradability of the sector's output. For relative non-tradables, there is little danger of losing market share due to competition from abroad. This would be true of the SO₂ allowance trading programme which targeted the electricity supply industry. Conversely, for goods which are highly tradable, even if the programme is national in scope (*i.e.* affects all domestic competing firms), the firm may face considerable competitive pressure. In theory, this would have been true of CFC trading, if there had not been a simultaneous multilateral agreement on global emissions.

While spatial coverage is clearly an important determinant of the competitiveness effects of a TP programme, the more specific treatment of spatial issues within the programme will also affect the distribution of costs and savings between the firms which are covered. Differentiated treatment of firms, depending on where emissions occur is a reflection of the fact that almost all pollutants have different effects depending upon their location of emission.¹⁶ Under direct regulation in the US, spatial concerns are addressed in a rather rudimentary manner through different technology-based rules in attainment and non-attainment zones. These have remained in place with permit trading since a non-differentiated trading system may have adverse consequences if the effects of the pollutants are dependent on the location of emission. However, in some cases, the trading system itself has also been adjusted to reflect spatial differentiation of effects.

For instance, under RECLAIM, Los Angeles' air basin was divided into eastern and western zones. Firms in the western zone – which has much worse air quality – can only purchase permits from other firms within the western zone, while firms in the eastern zone can purchase permits from any firm in the basin. In addition, permit price ratios were differentiated depending upon the location of buyers and sellers [Stavins (1998); Johnson and Pkelney (1996); and Keohane *et al.* (1997)]. Once this constraint becomes binding, western-denominated permits will become increasingly scarce, driving up their price. This benefits firms in the eastern zone, relative to firms in the western zone, because they can purchase permits at lower cost. It may also affect investment flows, with capital flowing from west to east.

The effects of a system with an "ideal" treatment of spatial issues is explored in a simulation of air quality in St. Louis conducted by Atkinson and Tietenberg (1984). They compare the effects of introducing a system based on emission permits with a system based on ambient permits. In the former case, permits are not differentiated by location of emission and therefore, the total level of emissions will have to be lower than under an ambient permit system which distinguishes between air quality within the urban area. Aggregate cost savings to meet the primary EPA standard are approximately 10 per cent higher with the ambient permit system. More significantly, many fewer firms are made worse-off using the ambient permit system than the emission permit system. For instance, as many as twelve of 27 firms are appreciably worse off (> \$25/day) under the emission permit system, but none are worse-off under the ambient permit system. Presumably, the firms which benefit from an efficient treatment of spatial issues would be the same firms which have to reduce emissions (or buy permits) under an emission permit system, despite the fact that their own emissions are not contributing to exceedences of air quality at local receptors.

Conversely, if spatial issues are not addressed directly in the TP programme, firms in areas with high pollution concentrations may benefit disproportionately from its introduction. For instance, despite the importance of spatial issues in the generation of acid rain, the SO₂ allowance trading programme ignored spatial issues altogether. In particular, the majority of damages were taking place in the Northeast due to emissions from a relatively small number of plants in the Midwest [Sorrell (1994) and Cole (1998)]. While it was recognised that this would make the programme less efficient, it was felt that spatial issues could be ignored, since allocations were to drop significantly in aggregate in any case, and since relatively low abatement costs in the Midwest would make them big sellers of permits, even without differentiated incentives [Tietenberg (1998) and Sorrell (1994)]. This implies that (relative to their contribution to damages) firms in the Midwest have probably gained disproportionately from the introduction of the more efficient tradable system.

Permit incidence

While it is quite likely that the most important cost (and competitiveness) effects associated with the introduction of a tradable permit programme will be incurred by firms which are directly involved in the programme itself, in some cases this might not be so. Upstream suppliers and downstream users may face more significant competitiveness effects following the introduction of a tradable permit regime which does not target them directly. Thus, the “point of incidence” for permit systems may be a significant determinant of the distributional effects of tradable permit systems.

This is clearly illustrated by the Fox River BOD programme, which targeted wastewater treatment plants. Since these are local monopolies, and usually owned by municipalities, any competitiveness effects would have been felt by important generators of wastewater, and not by the permitted facilities themselves. Similarly, the SO₂ allowance trading programme is restricted to generators of electricity. The competitiveness implications for the generators themselves are small, since they are subject to price-regulation, and are often local monopolies. However, for these very same reasons, the implications for downstream users may be significant. Table 6 gives data on the relative intensity of electricity use by manufacturing firms in the US in 1991. On this basis, primary metals, petroleum and coal products and textile mill products, paper and allied products and stone, clay and glass products are worst affected by a programme which results in cost increases for electricity. Analogously, they will gain the most if costs fall.

Table 6. **Primary Energy Consumption by Manufacturing Firms**

	1 000 BTU/\$ Value Added
Food and Kindred	1.134
Tobacco	0.123
Textile	3.768
Apparel and Other Textile	0.547
Lumber and Wood	2.265
Furniture and Fixtures	0.826
Paper and Allied Products	3.421
Printing and Publishing	0.501
Chemicals and Allied Products	2.756
Petroleum and Coal Products	4.332
Rubber and Misc Plastics Products	2.175
Leather and Leather Products	0.737
Stone, Clay and Glass Products	3.295
Primary Metal Industries	10.372
Fabricated Metal Products	1.279
Industrial Machinery and Equipment	0.801
Electric and Electronic Equipment	0.947
Transportation Equipment	0.776
Instruments and Related Products	0.485
Misc. Manufacturing Industries	0.576

Sources: USDOC and USEIA.

The users of electricity were not themselves grandfathered permits, and as such, the rents will be accrued by the permit holders, but control costs will be borne by the downstream users. Similar issues arise in the case of the CFC trading programme, where the biggest downstream industrial uses are as a cooling agent in refrigeration and air conditioning, as a solvent to clean electronics equipment, as a propellant in aerosol sprays, and as a foaming agent in insulation and packaging equipment (see Levy, 1997). As such, the competitiveness effects may not be felt by the manufacturers of CFCs, but by the electronics sector, many service sectors and the construction and packaging industries (Hahn and McGartland, 1989).

Upstream sectors may also be adversely affected by the permit trading programme. For instance, much of the negotiations in the SO₂ trading programme revolved around the effects that it would have on States with large high-sulphur coal deposits. States such as West Virginia feared that the switch from a technology-based regulatory regime (based on the use of scrubbers) to a more flexible system which allowed alternative compliance strategies would hit their coal mining communities badly [see Joskow and Schmalensee (1998)]. While the previous regime effectively provided no disincentives for the use of high-sulphur coal, the tradable permit regime does do so, since all emissions are charged. Thus, as discussed earlier, mining communities in the low-sulphur Powder River Basin in Wyoming and Montana benefited from this change.

Revenue recycling

As noted above, unlike grandfathered permits (and direct forms of regulation), auctioned permits (and taxes) require firms to pay for their residual emissions. While true, this overstates the difference between the various policies in financial terms, since some of these same firms may reap some of the benefits associated with increased government revenue arising from the permit sales or emission taxes. Moreover, under a grandfathered system, the public authorities could tax away the rents. This was the case in the CFC trading programme. Even if no explicit effort is made to capture the rent, the windfall gains from grandfathered permits will be subject to existing taxes on profits.

However, there may be more general efficiency gains from the introduction of a revenue-raising policy instrument. Indeed, in discussions of the effects of environmental taxes, it has been argued that under certain circumstances the revenue might generate a “double dividend” whereby the policy would reduce both environmental and economic distortions. [For a review of the conditions under which this is likely to be so, see Goulder (1994). Other discussions include Parry 1995, and Bohm 1997.] There has been a lively debate about the relative likelihood of such a “double dividend”, revolving around the likely relative sizes of the “tax interaction” effect (the extent to which a tax exacerbates distortions on existing taxes) and the “revenue recycling” effect (the extent to which the revenue generated can be used to reduce distortions by lowering tax rates).

There is a general consensus that any policy which generates revenue (*i.e.* an emission tax or an auction-based allocation) is potentially less-distorting than one which does not (*i.e.* a grandfathered allocation or direct regulation), since the benefits of the “revenue effect” are not exploited [Burtaw (1998)]. Thus, in an analysis of SO₂ emissions Goulder *et al.* (1997) find that a tax (or auctioned permit) would have been more efficient than a grandfathered permit, since the revenue can be used to lower other (more distorting) taxes. Assuming that enough permits are issued to generate a permit price of \$250/ton, the efficiency gains would be twice as great for the auction-based system than the grandfathered system. In fact, they find that unless the environmental benefits exceed \$104/ton, a grandfathered permit system would actually reduce welfare.

The firms directly affected by the permit system are more concerned with how the revenue is recycled than they are with overall gains in welfare. In theory, it is possible that the existence of revenue recycling could entirely compensate firms for their expenditures on permits. Thus, there is no reason to expect that “productive” sectors of the economy will face increased costs. Indeed, depending upon how this is done, the auctioned permit system could have equivalent effects to a grandfathered system, with revenues recycled according to analogous allocation rules. Alternatively, the revenue could also be used to meet the programme’s environmental objectives, with the funds being targeted for abatement technology innovation, adoption, and diffusion.¹⁷ This would alleviate adverse competitiveness effects and (if effective) could also result in environmental gains.

However, in most cases, individual firms will tend to do rather better under a grandfathered allocation. Why is this so? It is important to remember that almost all of the discussions about the public finance aspects of revenue-raising environmental policies relate to carbon emissions. In this case, it is plausible to assume that the authority implementing the tax is the same authority which is responsible for the public purse. More specifically in the case of CO₂, it is plausible to assume congruence between the environmental and fiscal regimes. It is also plausible to assume that most firms would pay the tax, to one extent or another, and also that they will realise the benefits of recycling, to one extent or another. Some firms will win (those whose production processes are intensive in the use of the factor or good whose tax rate is lowered), and some will lose (those whose production processes are carbon-intensive).

The situation is more complicated in the case of most existing tradable permit systems, this is because the number of firms directly affected by TP systems tends to be much smaller, even for national systems. As such, the expenditures will be borne by few, but the benefits of recycling will be widely diffused, unless special arrangements are established at the same time. If all SO₂ permits were auctioned, under almost any conceivable means of revenue recycling (*i.e.* lower income taxes, employers' national insurance contributions, corporate taxes or VAT), the affected utilities would gain little unless the benefits were specifically targeted at them. Given this, the firms directly affected by the permit system will still prefer a grandfathered system, even if some of the costs of an auctioned permit system can be mitigated through revenue recycling.

However, there is one form of revenue recycling for which this may not be the case: a zero-revenue auction. In this scheme, firms are distributed permits, which they then have to offer for sale in an auction and simultaneously submit bids for the number of permits desired at different prices. The market-clearing price is determined, and firms receive permits consistent with their bids. The permit price is then used to determine the amount firms pay for permits purchased and the amount they receive for permits allocated [Sorrell (1994) and Franciosi *et al.* (1993)]. The original elaboration is in Hahn and Noll (1982). Net revenue for the government (and for firms in aggregate) is nil.

In theory, such a scheme would reduce fears of adverse competitiveness effects from a pure auction-based system, without granting firms the rents as under a grandfathered system. However, it has never been attempted. Part of the reason for this may be due to the susceptibility that such a system would have for strategic behaviour. For instance, firms with higher *ex ante* emissions than would be cost-effective under a tradable permit system would have an incentive to bid up the price of permits [Bohi *et al.* (1990)]. Thus, due to fears about the potential for strategic behaviour undermining the efficiency of the programme, under CFC trading the rents were "soaked up" with a tax, rather than a zero-revenue auction.

Tradable permits and dynamic competitiveness effects

The preceding sections have been largely concerned with the financial effects of tradable permit systems. On the one hand, in aggregate, firms will tend to benefit from the efficiency gains of tradable permit systems relative to direct forms of regulation, although in cases where permits are auctioned rather than allocated freely these effects may be more than counter-balanced by the cost of payments for residual pollutants. On the other hand, the specific competitive position of individual firms and sectors included in a tradable permit system will be affected by factors such as the permit allocation mechanism, initial abatement costs, the spatial scope of the programme, the incidence of permits in the production cycle, and the means of revenue recycling.

These factors are important, and (as noted above) are the subject of most of the negotiations and controversies when tradable permit systems are proposed. However, over time and for the economy as a whole, the dynamic effects of tradable permit systems may prove to be even more important. In this section, five such factors are addressed: adjustment costs; economic rents and investment rates; shut-down credits and firm exit; innovations in abatement technology; and firm entry and productivity levels.

Permit systems and adjustment costs

By their very nature, tradable permit systems give firms a degree of flexibility in the way that environmental objectives are met. This is true not only in terms of differences in the way that individual firms

respond, but also in terms of the way that a given firm adjusts its compliance strategy through time. This is particularly important when the long-lived nature of capital investments is considered. If capital is sector-specific, the speed of adjustment may be slow and costly. This is particularly true if capital investments and compliance strategies are not strictly separable, such as when abatement is bound up in more general production processes.

Thus, in some cases, direct forms of regulation may slow down the rate of capital turnover, by providing incentives for the retention of capital equipment which adheres to existing regulations. In other cases, however, direct forms of regulation may mandate adjustments in capital which are excessively costly, by mandating the retirement of equipment prior to obsolescence. Both of these will have adverse competitiveness effects, either by slowing down the rate of technological change, or by mandating technological change at excessive cost. Conversely, permit systems (and emission taxes) allow firms to adjust their compliance strategies in a manner which is consistent with their more general plans for capital turnover, and thus comes at lower cost.

However, adjustment costs may occasionally be higher under a tradable permit system than under an emissions tax. The former sets the quantity and lets prices adjust, while the latter sets the price and lets quantities adjust. If firms find that the adjustment costs of compliance strategies are adversely affected by price instability, they will prefer taxes to permits. However, tax rates may also be adjusted if environmental objectives are not met. Moreover, the capacity to bank permits (as in the lead trading programme and the SO₂ trading programme) may help firms to adopt a lowest-cost compliance strategy. For instance, if there is considerable uncertainty about future permit prices, banking may allow firms to adopt a compliance plan with more confidence about the overall cost. Other firms may be concerned about permit availability in a wider sense. This is valuable to risk-averse firms, and its importance is reflected in the fact that approximately one-third of Phase I permits in the Acid Rain Programme have been banked for use in Phase II [Ellerman *et al.* (1996) and Schmalensee *et al.* (1998)]. In cases where banking is not allowed, the development of a futures market for permits (as exists for SO₂ in Chicago) can serve the same function. Thus, in some senses, adjustment costs may be less under a tradable permit system than under an emissions tax, because the former provides mechanisms through which adjustment costs can be minimised.¹⁸

Adjustment costs may also be reduced considerably if there is a relatively long lead-in time. This would arise for two reasons. On the one hand, if there is considerable uncertainty in the costs of compliance prior to the introduction of the tradable permit regime, it may take time for the market to settle down, and for convergence to occur in permit prices. On the other hand, if there is a long lead-in time, this will help firms evaluate alternative compliance strategies, some of which have far-reaching implications. In the SO₂ programme, the permit market came into being a number of years before the programme itself was implemented, and prices became much more stable over time. However, despite this, a number of firms appear to have undertaken compliance strategies (principally buying scrubbers) which are not their lowest-cost adjustment path.

Economic rents and investment rates

It might be presumed that the need to pay for residual emissions under an auctioned permit system would give firms that have been grandfathered permits a competitive advantage, allowing them to increase their market share. However, this is not necessarily the case. In terms of the firm's decisions with respect to pricing, the difference between the two systems may be unimportant in the short-run, since in both cases, the possession of a tradable permit implies the existence of an opportunity cost equal to the value of the permit [Koutstaal (1997)]. Opportunity costs arise (even if permits are granted freely) because profits are foregone if the firm retains the permits. Under perfect competition, a profit-maximising firm will behave as though its marginal costs have increased by the same amount as under an auctioned allocation. Under conditions of imperfect competition (in the market for permits or products) this will not be precisely true, but there will still be an opportunity cost associated with the retention of permits.

This does not, however, mean that differences in the two allocation mechanisms are not relevant for longer-term competitiveness effects. In order to clarify the distinction, let it be assumed that markets are perfectly competitive, and that total control costs (abatement costs and permit expenditures) rise under auctioned permits, relative to direct forms of regulation, but fall under grandfathered permits. Since prices are the same for the two permit systems, this means that profits will rise for firms which are grandfathered permits, and fall for firms which are auctioned permits.

Firms which have been grandfathered permits can use the economic rents (the windfall profits) which have not been taxed away by the government either by increasing dividends to shareholders or by increasing investment rates [Holling and Somerville (1998)]. Assuming that a proportion of the rent goes toward undertaking investment, this will help firms to increase their productivity, making them more competitive in the medium- and long-run. They may also be able to expand into new (and more profitable) product markets. And finally, they may be in a better position to survive economic down-turns. Firms which have to buy their permits in auctions (or pay emission taxes) will not be in a position to do so, and may even find that earnings which had previously been retained under direct forms of regulation (and thus available for investment) are now devoted toward expenditures on permits. This is particularly problematic if abatement is low, relative to total “uncontrolled” emissions.

Despite the fact that the role of permits on profits and investment levels may contain some of the most important competitiveness effects, this issue has not been explored in the literature on existing tradable permits, which tend to concentrate on the short-run effects of permit systems. Those studies which do look at the effects of grandfathered permit systems on profit rates are more concerned with overall welfare effects than competitiveness effects for the permitted sectors [Fullerton and Metcalf (1997) and Goulder *et al.* (1997)].

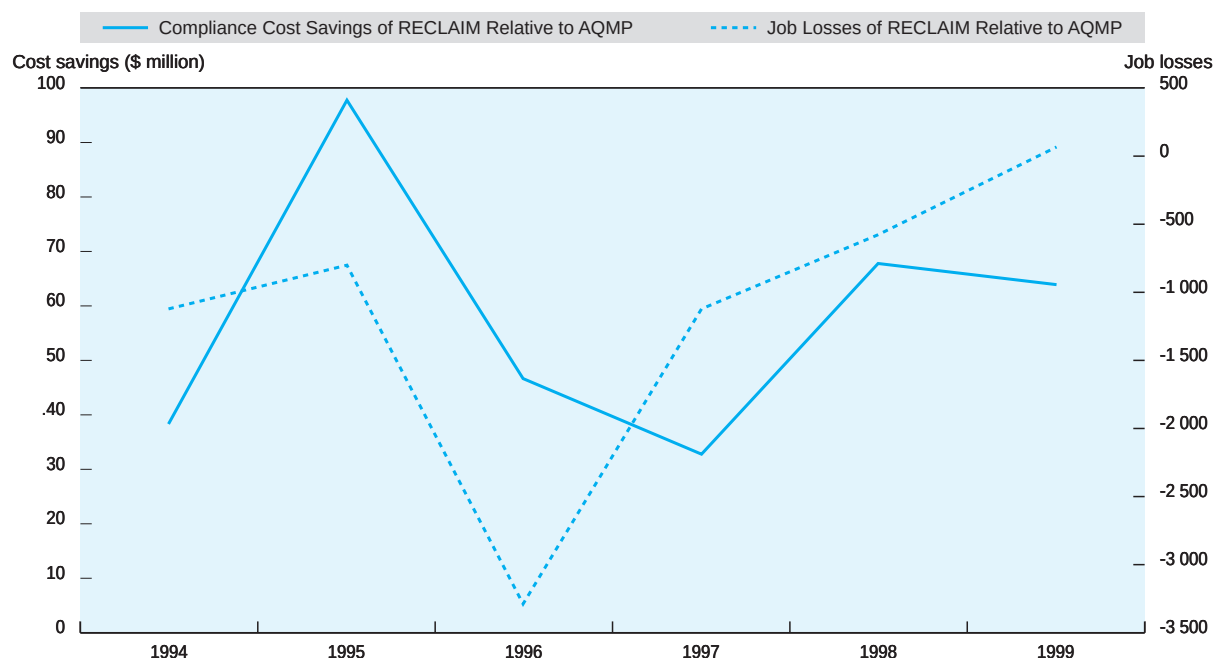
Shut-down credits and firm exit

The opportunity cost of permits may also generate an incentive for firms to relocate or shut down. By doing so, they will avoid payment of compliance costs, but still retain the benefits associated with the value of the permits themselves [Johnson and Pkelney (1996); and Koutstaal (1997)]. Paradoxically, relocation may be a more attractive proposition under a permit system than under direct forms of regulation, even if (as under a grandfathered permit system) the compliance costs of the former are necessarily lower.

This fear that permit systems would provide an incentive for firms to relocate has been largely behind controversies concerning the treatment of “shut-down credits.” In the event that shareholders retain full rights to the permits, irrespective of whether or not the facility for which the permits were granted remains in business, the incentive to relocate will be greater than if this is not the case.¹⁹ Regulators often try to find ways to avoid this by driving a wedge between the opportunity cost of the permit for a plant within the region (or in business), relative to one outside the region (or out of business).

For instance, if permits owned by firms which are leaving the sector or the region are restricted in some way, their value will be much less than for those which remain. While this may discourage some relocation, it may do so in a very inefficient manner. For instance, some plants will continue to produce in one location, even when it is more efficient to produce elsewhere. Similarly, they may continue to produce particular goods in order to retain the value of their permits, even though it is not efficient for them to do so. Alternatively, it may encourage some firms to undertake unproductive activities. For instance, some firms may keep a skeleton plant in the original location in order to retain permit rights, but shift the rest of production elsewhere. And finally, the cost of ensuring that a “modified” plant is not a “new” plant may be prohibitive for the implementing agency [see Tietenberg (1998), Sorrell (1994), and OECD (1994) for discussions].

Some indirect evidence of the importance of opportunity costs on relocation are found in the simulations of the RECLAIM programme conducted by Johnson and Pkelney (1996). Even though compliance costs are considerably lower under the RECLAIM programme relative to the pre-existing direct regulations (an unweighted average of just over 50 per cent between 1994 and 1999), jobs foregone actually exceed those under the AQMP system by 1999 (Figure 3).

Figure 3. **Cost Savings and Job Losses of RECLAIM Relative to AQMP**

Source: Johnson and Pekelney, 1996.

Due to the significant opportunity costs arising from high estimated permit prices, it would appear that some firms may reduce their level of activity or leave the region altogether. This has implications for both regional outputs and employment levels. Thus, it will be politically difficult to justify the creation of rents which are realised by firms when they cease to contribute to productive activity and employment in the region. However, any effort to remove this potential effect may be inefficient, by encouraging unproductive (and polluting) firms to remain in business in the region. There may not be optimal “exit” from the sector in the region, and environmental targets may not be achieved at lowest cost.

Innovations in abatement technology

Another important effect of tradable permit systems in dynamic terms is their capacity to induce technological innovation in abatement. As far back as the mid-1970s, it was pointed out [Kneese and Schultz (1975)] that “over the long haul, perhaps the most important single criterion on which to judge environmental policies is the extent to which they spur new technology towards the efficient conservation of environmental quality”. Analogously, the effect that policy measures have on incentives to innovate in abatement technologies are certainly a more important determinant of their long-run economic costs for firms than short-run differences in compliance costs of the sort discussed above.

On this basis, permits (and other market-based instruments) are likely to have fewer adverse competitiveness effects than many other measures. In particular, while the case for market-based instruments relative to direct regulation has usually been made in static terms related to the efficient allocation of given factors, it is thought that the case is even more convincing when the dynamic effects of technological innovation are included in the analysis. Although the benefits of technological innovation are largely lost to the firm under more direct forms of regulation, under a market-based regime, the firm itself is able to realise some of the economic benefits of technological innovation and adoption [Downing and White (1986); Milliman and Prince (1989); and Nentjes and Wiserman (1987)].²⁰

This much is well-documented in the literature. What is less well-documented is the effect that alternative economic instruments have on incentives to innovate, and what limited work that has been conducted in this area has been on the relative merits of subsidies, on the one hand, and permits and taxes, on the other.²¹ For the most part, the latter have been advocated by economists, since subsidies may lead to firm entry, and thus potentially have perverse environmental consequences. Although the pollution-intensity of production may fall, aggregate pollution emissions may rise [see Wallace (1995) and Baumol and Oates (1988) for a discussion]. This point is clearly relevant for the distinction between emission taxes and auctioned permits, on the one hand, and grandfathered permits, on the other (since firm entry and firm exit will differ).

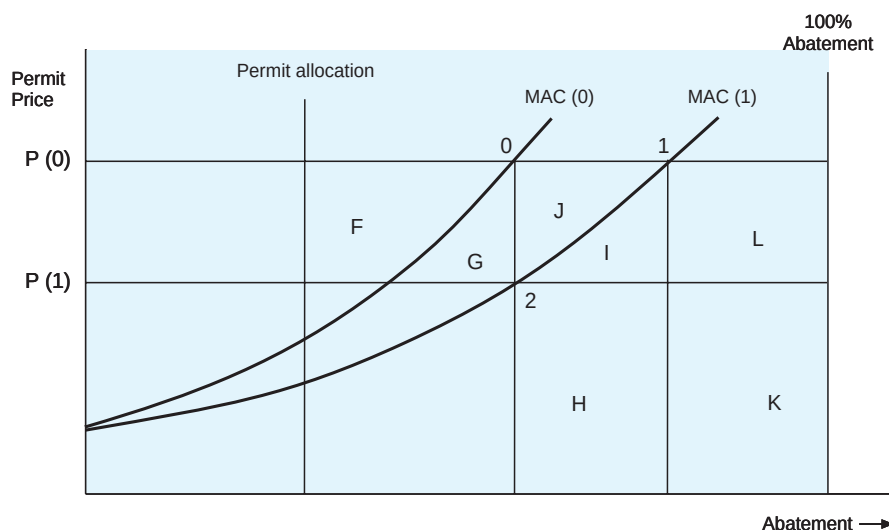
However, there has been some work which has explicitly compared the effects of auctioned and grandfathered permits on abatement innovation. Malueg (1989) argues that auctions are more effective than grandfathering in encouraging abatement innovation. In his study, he shows that if permits are grandfathered, and if the “potential” adopter of an innovation is a buyer of permits, the firm may not actually adopt the innovation. This is because, unlike under an auction (or a tax), the savings from the innovation are only realised up to the point of the allocation actually grandfathered, and not for the entire reduction in emissions.²²

Milliman and Prince (1989) compare the effects of emission subsidies, permits and taxes in terms of incentives both to innovate and to adopt innovations. They find that all of these instruments have the same effects in terms of incentives to innovate (as reflected in terms of net benefits), and all are superior to direct regulation. However, they may have very different effects in terms of incentives for innovators to encourage diffusion. While taxes and subsidies perform well, the effects of the two permit systems are very different, with auctioned tradable permits having much stronger incentives than grandfathered tradable permits. This result arises because under an auctioned allocation the innovating firm will get the benefit of lower permit prices if other firms adopt the innovation. Under a grandfathered allocation, the fall in permit price will benefit buyers, but hurt sellers.

This can be seen in Figure 4, where an innovation leads to a reduction in abatement costs from $MAC(0)$ to $MAC(1)$, shifting from the initial equilibrium to the new Equilibrium 1. Since the innovating firm is not large enough to behave as a price-setter, the permit price will remain the same. However if the innovating firm allows for the diffusion of the innovation it will drive down the permit price in the market, and thus shift down its own marginal abatement cost curve to Equilibrium 2. If the firm has an initial grandfathered permit allocation less than its cost-effective level of emissions, it will not seek to diffuse the innovation. This is because, in doing so, the gains from reduced abatement costs will be exceeded by the loss in payment receipts. Overall, the loss will be equal to the area $F + G + J$ if it allows for the diffusion of the innovation since its abatement costs will have fallen (by area $H + I$), but its permit receipts will have fallen by more (area $F + G + H + I + J$). Conversely, under an auction system, it will gain overall (area $I + L$) if it diffuses the innovation since its abatement costs will fall by the same amount as under a grandfathered system ($H + I$), but permit payments will rise (fall) by area $H-L$. Thus, the effects will depend upon the innovating firm’s initial allocation, but will in any case be less than under an auctioned allocation.

However, if the innovations are patented, the positive effect of royalty payments arising from diffusion will certainly far outweigh any negative permit price effects for permit sellers under a grandfathered allocation. Moreover, much of the literature assumes that innovations are internal to the sector (*i.e.* generated by firms which emit the permitted pollutants). This is not always (or even usually) the case. Thus, the differences between the effects of different allocations under tradable permit systems is certainly less than the difference between TP systems in general and direct forms of regulation. And finally, applying insights from the political economy literature Kemp (1997), argues that technological change is likely to be greater under a tradable permit system in which permits are grandfathered than under a tax-based system or an auctioned tradable permit system. This result arises from a belief in the reciprocal relationship between regulators and firms, whereby the nature and stringency of regulations may be endogenous. In particular, if there is uncertainty about the costs and benefits of environmental measures and the regulator is wary of imposing excessive costs on firms out of a fear that it will have adverse competitiveness effects, a grandfathered tradable permit system will tend to be more stringent, thereby inducing greater incentives for technological innovations.

Figure 4. Incentives to Diffuse Innovations Under Auctioned and Grandfathered Permit Systems



The more general belief in the ability of tradable permits to stimulate abatement innovation has been borne out by some of the tradable permit systems actually introduced. The most telling example is certainly the SO₂ allowance trading programme. Under the Clean Air Act's rules, firms effectively only had one option for reducing emissions (*i.e.* to install scrubbers). The allowance trading programme allowed firms more flexibility in their choice of abatement options such as switching fuels, buying low-sulphur coal and buying permits. Thus, it introduced innovation in the sense that firms adopted technologies that they had not adopted previously.

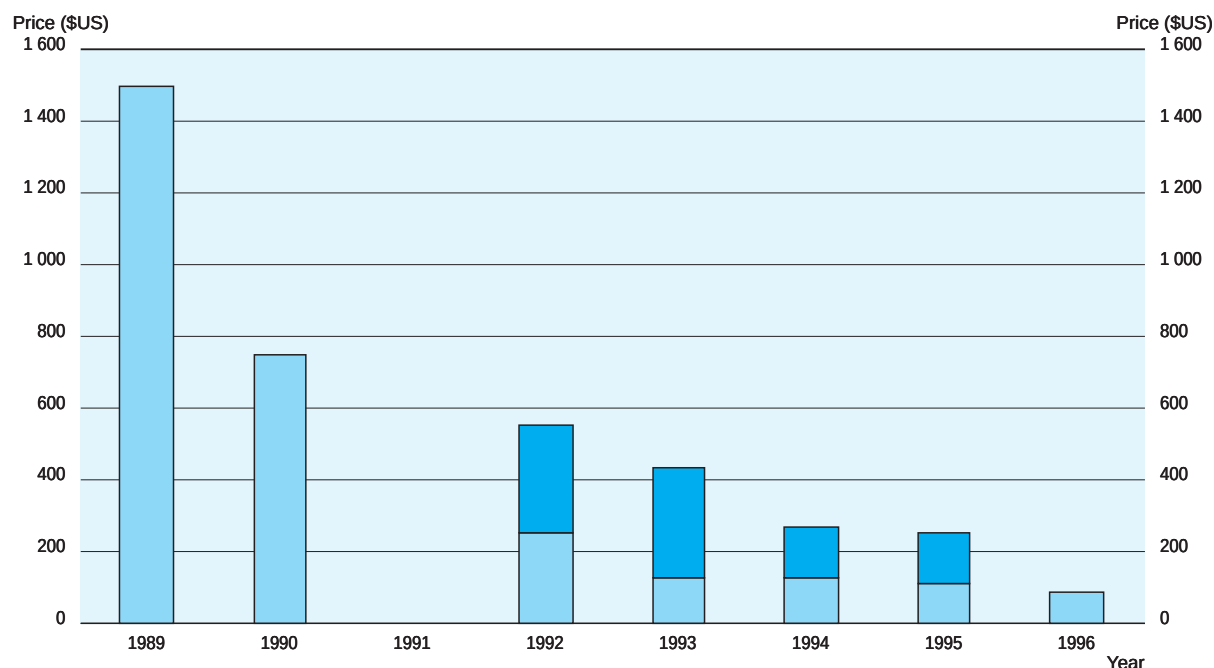
However, it also introduced more fundamental innovations. On the one hand, since the inauguration of the tradable permit system, technological improvements have allowed the price of scrubbers to drop significantly. In 1995, the capital cost of a scrubber sufficient for a 639 MW plant cost less than a scrubber half this size in 1989 [Burtaw (1996) and Bohi and Burtaw (1997)]. On the other hand, there have been improvements in fuel-mixing technologies, allowing firms to shift toward lower-sulphur mixes in a more cost-effective manner. And finally, the costs of transporting low-sulphur coal from the Powder River Basin have fallen, although this is due mainly to institutional, and not technological, factors.

All these developments arose because firms now had to compete in the market against other abatement options. Previously, there had been little incentive to innovate, unless it was felt that in doing so the regulators would respond and change the technology-based rules²³ [see also Ellerman and Montero (1996)]. In light of this, abatement costs plummeted. Figure 5 traces SO₂ allowance prices, and shows how important these effects can be [Bohi and Burtaw (1997)].

In general, therefore, there seems to be little question that market-based instruments provide better incentives for abatement technology innovation and diffusion than direct regulation. They will have positive competitiveness effects since a policy which pushes firms onto a technological trajectory which allows them to meet given environmental objectives at lower cost are likely to be more important than the costs borne by the firm in the first instance.

Barriers to entry and productivity

As noted earlier, power in the market for permits may have implications for the efficiency of tradable permit systems. However, in the long-run, the effects of market power on the underlying competitiveness of the economy may be more significant than this would imply. There is a danger that power in the permit

Figure 5. SO₂ Allowance Prices¹

1. Figures for 1989-1990 are *ex ante* estimates and the shaded region for 1992-1995 gives the range of prices.
Source: Bohi and Burtaw (1997).

market may be used not only to affect relative costs of production for existing firms, but also to exclude firms from the market altogether [Misiloeck and Elder (1989)]. Unlike under taxes or direct regulations, there is a danger that permits may become the vehicle through which existing firms are able to exercise market power, by excluding other firms. This may have the direct “competitiveness” effect of allowing for the exclusion of firms (domestic and foreign) which did not have plants in the permit jurisdiction in the year which is used to determine the baseline permit allocation.

This “exclusionary” behaviour is perhaps more significant in terms of competitiveness effects, due to the fact that new firms are often important instigators of new products and production processes [see Geroski (1991) for a very full treatment of the role of firm entry in the dynamics of the economy]. Thus, market power may reduce competitiveness, by slowing down the rate of technological change, and by slowing down the rate of product development. Since two of the most important factors behind “competitiveness” are the ability to increase levels of productivity and the ability to shift into new markets, the potential for exclusionary behaviour in the market for permits needs to be examined closely.²⁴

What are the conditions necessary for there to be exclusionary behaviour in the market for permits? Most importantly, it is a pre-condition for exclusionary behaviour that firms compete in both the market for permits and outputs, since there is no benefit to be gained from excluding firms from markets in which the firm with market power does not compete. Thus, exclusionary behaviour is more likely for pollutants in which emissions (or at least permitted emissions) are highly concentrated in a few sectors, and most firms in this sector are included in the systems. For instance, the danger of exclusionary behaviour in the market for CFCs was probably much greater than the danger of exclusionary behaviour in the market for carbon monoxide.

Table 7 illustrates the approximate degree of correlation which exists between sectoral concentration and emission-intensity, listing the percentage of total emissions from manufacturing and the approximate Hirshmann-Herfindahl indices of sectoral concentration. The figures were derived using IPPS data on emissions by sector (Hettige *et. al.*, 1995) and USDOC data from the Census of Manufactures on sectoral

concentration (UDOC, 1992).²⁵ Significantly, for all pollutants there is a negative correlation between the concentration of emission-intensity and the economic concentration of the sectors'. Thus, cost manipulation as a means of excluding firms may be less likely than is often feared.

Table 7. Relationship Between Emission Concentration and Sectoral Concentration

sector	PM	CO	SO ₂	NO ₂	VOC	BOD	TSS	Approximate H-H Index
Food and Kindred	5.37%	0.63%	2.25%	4.71%	3.10%	22.20%	0.25%	203
Textile and Apparel	0.94%	0.28%	1.30%	3.69%	3.42%	0.74%	0.05%	75
Food and Paper	11.95%	7.46%	1.37%	5.67%	25.32%	0.89%	0.15%	80
Petrol. and Chemicals	4.40%	10.10%	7.09%	9.02%	7.70%	36.17%	4.26%	198
Rubber and Plastics	6.55%	17.60%	13.12%	22.86%	34.09%	15.30%	1.77%	48
Stone, Clay and Glass	46.02%	3.20%	20.09%	29.60%	3.54%	0.23%	0.12%	115
Primary Metals	24.22%	59.95%	54.19%	23.49%	15.93%	24.36%	88.48%	73
Chemical Products	0.39%	0.74%	0.51%	0.81%	4.56%	0.10%	0.04%	910
Other Manuf.	0.16%	0.03%	0.09%	0.14%	2.34%	0.00%	4.89%	120
Correl. Coefficients	-0.290	-0.269	-0.285	-0.407	-0.327	-0.195	-0.193	

However, cost manipulation by existing firms may not be the most important means of exclusion in any case. It is not marginal costs of production so much as the absolute costs of entry which matter. Thus, the means by which permits are denominated may prove to be important, since it is the cost of initial permit purchases relative to total start-up costs which will determine whether or not new firms can be excluded. For instance, if permits are denominated in terms of flows, and are valid for a number of years, it will be much easier for firms with market power to drive up the cost of permits sufficiently to present an insuperable barrier to new firms [Koutstaal (1997)]. As noted above, in the Fox River BOD programme permits were denominated in terms of annual flows, valid for five years, making initial permit-related capital costs relatively higher. Fears of the potential for barriers to entry were responsible for the decision not to denominate SO₂ allowances for multiple years.

One of the few researchers to look at the effects of new firms on total factor productivity is Geroski (1989 and 1991). Although he finds that relatively few entrants can be characterised as "innovators" (in terms of products or processes), he feels that firm entry still has appreciable effects on innovation, both directly through the innovators themselves, and indirectly as a spur for insiders to innovate. He concludes that approximately 30 per cent of productivity growth in the UK can be attributed to firm entry, and the effect is even greater in the long-run. Looking directly at the relative productivity rates of new entrants and incumbents, Baldwin (1995) finds that as much as 29-30 per cent of growth in total factor productivity in North America can be attributed to new firms and plants. Overall, plant turnover may be responsible for 40-50 per cent of productivity growth. Some of these effects may be quite indirect. For instance, it is felt that new firms (or even the possibility of their entry) may also encourage more efficient use of given productive resources and spur innovation amongst incumbents (Geroski, 1991).

On the basis of results such as these, it is not surprising that many commentators have expressed concern that tradable permit systems which allow for exclusionary behaviour through market power may have adverse effects on the long-run competitiveness of affected sectors. Thus, with specific reference to RECLAIM, Polesetsky (1995) fears that tradable permit systems which generate market power may reduce market innovation; Maloney and Brady (1988) also express concern that it may reduce technological innovation. More generally, Koutstaal (1997) fears that an economy's "dynamism" will be adversely affected. This, perhaps more than anything else, has clear and troubling consequences for competitiveness.

Conclusion: the political economy of permit systems

There is little question that tradable permits have the potential to bring about significant efficiency gains relative to the more direct forms of regulation which they complement or replace. However, the

more specific effects for individual firms and sectors may vary widely. On the one hand this is attributable to fundamental economic and structural characteristics. On the other hand, it is attributable to the precise form that such systems take, affecting total costs of compliance in terms of abatement costs, permit expenditures and fiscal costs. (The first three rows of Table 8 reviews some of the main factors discussed in the study.) Thus, despite the apparent economic efficiency of tradable permit systems, for many firms and sectors there may well be adverse effects arising from their introduction.

Table 8. **The Main Competitiveness Effects of Tradable Permit Systems**

	Likely Direction	Main Determinants
Abatement Costs	Positive relative to direct regulations in aggregate. Likely to be same as emission taxes.	Affected by abatement cost heterogeneity, and initial abatement cost relative to equilibrium permit price. Also affected by imperfections in the market (<i>i.e.</i> transaction costs and power in the market for permits).
Permit Expenditures	Negative for auctioned allocations relative to direct regulations and approximately equal to taxes. Under grandfathering, dependent upon precise form of allocation for individual firms.	Affected by abatement cost heterogeneity, and initial abatement cost relative to equilibrium permit price. Also affected by imperfections in the market (<i>i.e.</i> transaction costs and power in the market for permits).
Fiscal Effects	May mitigate effects for permit expenditures under auctioned allocation, but unlikely to compensate fully.	Dependent upon precise form of fiscal reform in auctioned allocation. In grandfathered allocation also dependent upon tax treatment of “windfall” profits.
Adjustment Costs	Likely to be less than for direct regulations due to flexibility. Permit price instability may exacerbate adjustment costs relative to emission taxes.	Adverse effects of permit price uncertainty reduced by permit banking and futures market.
Industrial Relocation	Potentially negative for grandfathered allocation relative to direct regulations and taxes.	Spatial coverage of the permit system relative to market for output and the treatment of shut-down credits
Investment Rates	Likely to be positive for grandfathered permits relative to taxes and auctioned permits.	Affected by existence (or introduction) of any taxes on windfall profits and firms’ use of rent in grandfathered allocation.
Abatement Innovation	Positive relative to direct regulations and approximately equivalent to taxes. Grandfathered may have less positive effects on diffusion.	Distinction between auctioned and grandfathered allocation less important for patented or externally-sourced innovations.
Sector Productivity	Likely to be positive relative to direct regulations and equal to taxes. Potentially negative if permits allow for exclusionary behaviour.	Affected by sectoral coverage of permit system (concentration), and permit denomination (entry costs). Also affected by retention of regulatory bias against new firms.

This explains the intense negotiations and lobbying which always seem to precede their introduction. Indeed it is one of the great strengths of tradable permit systems that their flexibility in terms of implementation allows for the concerns and vulnerability of firms and sectors to be addressed. However, public authorities must ensure that this does not come at the expense of other firms in the economy. Most evidently, holding the environmental objective constant, increased permit allocations for one firm will come at the expense of allocations for other firms. The effects can, however, be more indirect than this. For instance, the point of incidence of the permit seems to be an important determinant of the distribution of costs and savings from tradable permits. In some cases, supplier industries or end users may be more adversely affected than the permitted firms themselves.

However, in general, the flexibility afforded by tradable permit systems is an important advantage. The allocation mechanism can be used to compensate firms which are adversely affected, leaving marginal incentives to reduce emissions the same. This is quite different from direct regulations and taxes, where exemptions and differentiated treatment of firms introduce distortions. This apparent separability between equity and efficiency in tradable permit systems is, however, dependent on a number of assumptions, and some of the measures which mitigate competitiveness effects for individual firms may, in fact, undermine some of the aggregate benefits of the measure.

In particular, the dynamic consequences of tradable permit systems will also depend on the precise means by which the measure is introduced. Adjustment costs, investment rates, industrial relocation, abatement innovation, and factor productivity are all affected by the characteristics of the measure. For instance, permit allocation mechanisms will affect all of these factors. Other factors such as the depth of the market, whether or not banking is allowed, how permits are denominated, the treatment of shut-down credits, and whether or not there are taxes on windfall profits are also potentially important (see the bottom five rows of Table 8).

Therefore, public authorities have a delicate balancing act to perform. There is little question that some firms may suffer from the introduction of a tradable permit system, particularly if permits are auctioned and permit expenditures represent a large proportion of total control costs. However, the flexibility inherent in tradable permit systems allow for any potential adverse effects to be addressed. Nonetheless, it is important that they not attempt to mitigate the adverse effects for individual firms in a manner which imposes excessive costs on other firms (or consumers and taxpayers), or which undermines the dynamic benefits of tradable permits.

NOTES

1. A fourth factor – the ability to exploit new market opportunities – is less relevant for this discussion, and is only discussed parenthetically.
2. Early estimates of the savings from the Fox River BOD programme were as much as \$7 million per year, but since only one trade took place, this illustrates just how speculative estimates based on *ex ante* simulations can be (Hahn, 1989).
3. Average costs of reactive organic gas abatement in Los Angeles prior to the introduction of the RECLAIM system ranged from \$113/ton to \$86,970/ton (OECD, 1994). However, in the final proposal, ROG's were not included in the programme.
4. Conversely, since the environmental consequences of all these pollutants are spatially-differentiated, an efficient environmental policy regime will have some degree of heterogeneity in marginal abatement costs across regions.
5. The latter issue is discussed below since, in theory, it need not affect the efficiency of the system, although it will affect the distribution of costs and savings between firms.
6. The implications that this may have on long-run productivity (and thus sectoral competitiveness) are discussed below.
7. This is an example of a “common agency” problem, whereby uncoordinated activities by multiple regulators of a single sector may inhibit efficient outcomes.
8. Sorrell (1994) adds assessment costs to this list. This involves the cost of evaluating what compliance strategy to adopt. For instance, under a technology-based system firms have no choice about compliance strategies and thus do not need to assess alternatives. By introducing flexibility, firms need to incur costs in assessing the relative desirability of different strategies. However, this is not specific to tradable permit systems, and will not be included in the discussion.
9. The figures are significant underestimates of the degree of concentration, since it is assumed that all the largest sources are from those sectors which have the highest average emission rate per firm. The more uneven the distribution of emissions within each individual sector, the greater will be the underestimation of the degree of concentration.
10. Conversely, this would have increased the transaction and administrative costs of the programme considerably, which is why the regulators were hesitant to expand the programme.
11. Although in the case of technology-based direct regulations, factor substitution elasticities are effectively nil.
12. This compares with regional cost savings of \$5,844/day, relative to the existing technology-based system.
13. They also look at a case where the same number of permits are allocated free of charge on the basis of “uncontrolled emissions.” In this case, 12 firms will face net payments in excess of \$25/day.
14. Although allocations were adjusted for “clean” units, which had installed scrubbers.
15. As with the SO₂ and lead cases, these factors are reduced proportionately over time, in order to meet more stringent air quality standards.
16. Most clearly, damages from pollutants such as SO₂ differ by location, depending upon diffusion functions and the characteristics of the deposition points. In addition, synergistic pollutants such as VOC's and NO_x (precursors to ozone) are affected by the simultaneous presence of other pollutants in a particular location. Significantly, all of the extant tradable permit systems involve pollutants which have non-uniform spatial effects.
17. See Carraro and Galeotti 1997 for a recent discussion.
18. However, in some cases, the efforts of individual firms to reduce their own uncertainty by banking permits may increase price instability in the market itself.

19. In the SO₂ case, some plants which did not even exist anymore were granted permits when the programme was introduced.
20. Kemp (1997) disputes the general validity of these results. Amongst other factors, he asserts that they neglect innovation by outside suppliers whose incentives are very different from the polluting firms themselves, and model the effects of innovations in pollution control technology in an unrealistic manner.
21. Wallace (1995) reviews some of the literature on differences between the effects of permits and taxes in terms of innovations. However, since these arise largely from contextual issues, such as market structure, they will not be discussed here.
22. However, it should be noted that the opportunity cost of grandfathered permits appear to have been ignored. If opportunity costs are included, the firm will adopt the innovation and sell the excess permits.
23. In fact, if rules are technology-based, the incentive for the innovator would be greater if the firm is certain that it will generate a rule change. This arises since the innovator's rents are protected by the patent (as under alternative policies) and the market is guaranteed by the rule (unlike under alternative policies). However, this seems unlikely and the innovating firm would face considerable risk in undertaking the necessary investments.
24. However, it is important to emphasise that a number of empirical studies have shown that CAC-based rules have tended to have particularly adverse consequences for firm entry, capital turnover and productivity growth (see Maloney and Brady 1988 and Nelson *et al.* 1993 for examples).
25. The figures are approximate, since the emissions data are based on the ISIC system of disaggregation and the concentration data are based on the SIC system of disaggregation. The latter (which uses a finer classification disaggregation) were aggregated into the former, and the Hirschman-Herfindahl indices of concentration have been derived by weighting by the value of shipments under the SIC system.

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