Implementing Domestic Tradeable Permits

RECENT DEVELOPMENTS AND FUTURE CHALLENGES



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Implementing Domestic Tradeable Permits

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Implementing Domestic Tradeable Permits:

Recent Developments and Future Challenges

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FOREWORD

An Experts Workshop on Domestic Tradeable Permits: Lessons and Future Directions was held on 6-7 September 2001 in Paris, as part of OECD work on the use of domestic tradeable permits (TPs) for environmental policy. The workshop objectives were to examine new areas for domestic TP applications currently being considered by OECD countries, and to better benefit from the lessons learned from the past experiences in domestic TP schemes by reviewing *ex post* evaluation methodologies.

This book contains the proceedings of the workshop, which were prepared by the workshop contributors, and were subsequently revised to reflect feedback of workshop participants and delegates of the OECD Working Party on National Environmental Policy. The Secretariat is grateful to the individual authors, workshop participants, and for comments provided by Member countries. The views expressed are those of individual authors. The book is published under the responsibility of the Secretary-General of the OECD

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Part I

SUMMARY OF LESSONS LEARNED

Chapter 1

DOMESTIC TRADEABLE PERMITS: SUMMARY OF LESSONS LEARNED

by

Olivier Godard
Laboratoire d'économétrie, Ecole Polytechnique, Paris, France

1. Transferable permits at a turning point

Growing interest in transferable permits (TPs) has developed from various horizons. They are increasingly seen as an efficient instrument for implementing environmental and natural resources management policies in the broader framework for sustainable development. Earlier analytical work[1] and general guidelines[2] by OECD also outlined such strengths as well as key design and implementation considerations for domestic TPs. Conceptually, some types of TPs appear to be quite appropriate to support sustainable development strategies. They explicitly set overall quantitative limits on key variables for environmental quality or sustainable management of resources and, at the same time, provide the flexibility required to achieve the best individual allocation of rights through decentralised transfers guided by the intensity of demand.

TPs are now considered in wider geographical regions, such as Europe, where they represent quite a new approach. They are also considered in new policy areas beyond the classic fields of local and regional air pollution control and regulation of fisheries. The most prominent extension in process is global climate policies and control of greenhouse gas (GHG) emissions in the context of the Kyoto Protocol (Kitamori, 2001)[3]. A framework for an EU-wide scheme of CO_2 TPs will presumably be in place by 2005. Also, other initiatives for more sector-specific use of the TP concept have been developed, for instance, for municipal waste management (the UK) and renewable energy development[4] (e.g. Denmark, Italy). Meanwhile, reflections have matured on the options for more systematic use of TPs in the transport sector (Raux, 2001)[5]. As a matter of fact, on the basis of the experience gained in the last 25 years, the present stage might be a turning point in environmental policies.

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1. See OECD (1999), *Implementing domestic tradable permits for environmental protection*. Paris, OECD Publ.; OECD (2001), *Domestic transferable permits for environmental management. Design and implementation*. Paris, OECD.

2. OECD (2001), *Strategic guidelines for the design and implementation of domestic transferable permits*, OECD WPEEPI, ENV/EPOC/GEEI(99)13/FINAL.

3. See Chapter 3, K. Kitamori (2001), *Domestic GHG Emission Trading Schemes: Recent Development and Current Status in Selected OECD Countries*.

4. See Chapter 4, R. Baron (2001), *Renewable energy certificates: trading instruments for the promotion of renewable energy.*.

5. See Chapter 5, C. Raux (2001), *The Use of Transferable Permits in Transportation Sector*.

2. What are TPs?

Any TP scheme is based on four basic conditions: (i) a quantitative environmental performance target to be achieved individually or collectively; (ii) a defined spatial and temporal flexibility given to regulated agents in the choice of location of resource extraction or pollution emission; (iii) enforcement capacities to ensure that actual performance of agents matches their allocated permits and objectives; and (iv) a distinction between assignment of environmental obligations and initial allocation of permits/property rights (Salmons, 2001)[6]. This disjunction allows separate treatment of equity and efficiency issues. In converse, where such a disjunction is already in place, there is room for introducing TPs. The condition (ii) above requires a well-defined environmental equivalence to enable transfer or trade of allowances for emissions in different locations or in time periods, for example.

A quantitative limit for TPs can be introduced either as a maximum ceiling for "cap and trade" schemes, or as a minimum performance commitment for "baseline and credit" schemes. These targets can be either in absolute terms or in relative terms. Many alternative baselines are possible for credits, allowing room for inventiveness on the part of policy-makers in designing a TP scheme. TP schemes can be developed along the lines of two social concepts:

1) as a ***private activity***; once the required regulatory framework is set-up, brokers and exchanges could play the main role in facilitating trades and give them an economic meaning, with minimal bureaucracy.

2) as a ***public activity;*** based on extensive public procedures such as public hearings and prior authorisation for all requested trades or transfers. This approach may be appropriate if transfers affect third parties and/or community concerns that may need protection, but at the risk of bureaucratic delays and unpredictability which could net out the economic benefits of trade.

There are elements of both in implementation of TP programmes in real life.

3. Sectoral lessons from "old" and "new" programmes

Past and on-going experiences of TPs have been reviewed several times at different phases of development of this still new instrument. Nevertheless, *ex post* evaluation remains a difficult exercise when necessary information is not identified up front when a new TP scheme is launched and collected during its implementation. Existing *ex post* evaluations do not use a standardised framework, and adopt a wide range of approaches from academic exercises to practical ones that are carried out in specific regulatory contexts. The most important issues for *ex post* evaluations are[7]: (i) the ability or the willingness of the regulated entities to take advantage of the opportunities of transferring or trading allowances or credits; (ii) the dissemination of economic information through the market; (iii) the magnitude of transaction costs, as well as regulatory and administrative costs; (iv) the extent to which market power can potentially be problematic; (v) adaptability and sustainability of a TP programme in face of unexpected difficulties and surprises; (vi) environmental performance in absolute terms (particularly important for schemes based on relative performance targets) and geographical distribution (issues of local pollution "hot spots" and social equity in access to environmental quality); (vii) change in the attitudes of different stakeholders and parties who are involved in the programme;

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6. See Chapter 6, R. Salmons (2001), *New Areas for Application of Tradable Permits – Solid Waste Management*.

7. See Chapter 2, S. Smith (2001), *Ex post Evaluations of Tradable Permits Programmes*.

and (viii) influence of the institutional process of during the *ex ante* design period on the rules of the game, and *ex post* acceptability and performance of that programme.

Furthermore, the papers prepared during the first phase of the current OECD project that reviewed "old" and "new" TP programmes (see Chapters 2 – 7) highlight typical features and results that give precious lessons as to how TPs can fit into specific sectoral contexts. The main lessons from these papers are summarised below.

3.1 Experiences in air pollution control applied to industry and power generation [8]

Since the mid-1970s, experiments in the use of TPs for air pollution control from industrial point sources in California have demonstrated the major effects of restrictions on their use and trade. These have included geographical restrictions on trades, and heavy regulatory requirements such as administrative approval of each envisaged trade and strict technology-based command-and-control approaches in parallel. Establishing the rules of a TP programme on paper is not sufficient to achieve economic performance. It depends on the ease for, and the willingness of the regulated to take advantage of trading opportunities. The more confidence they have in the regulatory authorities and the future value of allowances and credits, the more inclined they will be to participate in such a scheme.

RECLAIM: In the Los Angeles District, the RECLAIM[9] programme adopted rules quite similar to the Federal Acid Rain Programme under the 1990 Clean Air Act Amendments. The programme targeted two gases, SO_2 and NOx from point sources, leading to two separate systems within the same programme. Experience since January 1994 provides two main lessons. Until 1999, the total amount of allowances (Reclaim Trading Credits) distributed was greater than the quantity demanded. Most transfers were then made without being priced. About 20% of market transactions took place at low prices. One explanation for this, as provided by the authority in charge of the programme, was the unexpected progress in pollution control technology. How are we to understand such a situation? Does it mean that the programme was successful or not before 1999? The answer depends on the link that can be established between the existence of the programme and technological innovation. If innovation is considered to have occurred anyway, the programme's contribution in this area can be considered rather insignificant. On the other hand, it can be considered a great success if innovation was triggered by it. As a matter of fact, the cap became binding in 1999, and the 2000 price increased to unexpectedly high levels - ten times the levels of advance transactions made in 1999 for the same year 2000! This short term volatility is explained by the following factors embedded in the design of the programme: (i) no banking nor borrowing; and (ii) severe geographical limits. When a TP system is overly constrained spatially and temporally, it does not bring much flexibility and efficiency, and is unable to avoid structural crisis resulting from conflicting requirements of economic activity and environmental protection.

Lead trading programme: Another TP programme that was implemented in the USA in the 1980s to ease the phasing-out of lead in gasoline is generally considered a success. Many transactions took place, mainly for small refineries affording them reduced compliance costs, while large refineries were able to incorporate technological innovations and process changes more quickly. The success is attributed to clear identification and general acceptance of the programme objectives, limited regulatory bureaucracy (no *ex ante* approval of trades required), and pre-existing routine transactions of various products among the participating refineries. In spite of that, transaction costs have been estimated to be in the range of 10 to 20% of the potential economic gain from the programme. This

8. S. Smith (2001).

9. The Regional Clean Air Incentives Market (RECLAIM).

means that if a TP programme is expected to reduce costs of a given environmental achievement by 50%, the actual benefit may not be greater than 40%, even when the system is well-designed and flexible.

Acid Rain Programme: *Ex post* evaluations[10] of the SO_2 Acid Rain Programme confirm that this programme turned out to be a great success at an unprecedented geographical scale, since SO_2 emissions trading can take place across the entire US territory. This programme has been well publicised internationally. Several features deserve attention here. It demonstrated the superior simplicity of running a cap-and-trade scheme when compared to a baseline-and-credit one: there is no problem associated with agreeing on a baseline in each particular case, nor is there any need for prior approval of transactions. The programme also showed the importance of adaptation to surprises from the general economic and institutional contexts, specifically, in this case, the dramatic cost decrease in freight transport after the railway deregulation. One factor that explains why the actual compliance costs under the programme were lower than *ex ante* forecasts, is the fact that the agents were allowed to take advantage of external sources for cutting abatement costs, a result that command-and-control approaches would not have permitted. Finally, like other programmes, this one confirmed the role of actual market prices in providing a sense of the real costs of the programme, reducing controversies among experts who mobilise very diverse cost estimates. At the same time, the programme also confirmed that price movements could not be interpreted as the direct reflection of full costs, as current prices also depend on short-term interaction of supply and demand. Since the programme's inception in 1995, prices have varied within a range of 1 to 3, with a lower bound defined by variable or operating costs of pollution control, and upper bound by full development costs including fixed costs of considerable investments such as scrubbers.

Lessons on design mistakes can also be learned from this experiment. Two features of the programme have proven to be a source of unnecessary loss of economic efficiency. Firstly, during the first phase (1995-1999), some bonus allowances were allocated to power plants that had installed flue gas desulphurisation devices, which led to over-investments in such scrubbers. These early investments were based on the anticipation that future abatement costs would be much higher, which proved to be quite exaggerated. Secondly, "substitution" provisions during the same initial phase permitted companies to substitute other facilities for those obligated by the programme. This led to opportunistic behaviours in choosing facilities for initial permit allocations, aiming at maximising the amount of allowances received rather than selecting facilities with the greatest potentials of low cost abatement. The lesson from this is that some opt-in provisions may not lead to environmental efficiency nor cost-effectiveness, and that initial allocation rules should avoid narrowly defined technologies.

VOC emission trading programmes: Lastly, VOC programmes bring a specific experience in dealing with multiple pollutants. Regional or inter-states programmes in the USA have shown the difficulties of integrating multiple pollutants with different health effects within a TP scheme. This adds to the concerns of the public and NGOs for potential localised pollution "hot spots" as a consequence of trading. Thus, the social acceptance of trading in such circumstances has been much less than for programmes targeting only one gas, such as SO_2. For this reason, the RECLAIM programme eventually did not include VOC emissions trading as initially planned. Under the existing VOC programmes, strong regulatory requirements have been put in place, including prior approval of each proposed transfer, in order to limit possible adverse effects on public health. This resulted in higher transaction costs and less flexibility.

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10. See, in particular D. Ellerman, R. Schmalensee, E.M. Bailey, P. Joskow and J.-P. Montero (2000), *Markets for Clean Air. The US Acid Rain Programme.* Cambridge University Press, Cambridge (Mass.).

3.2 Experiences in water resources management[11]

The rights to use and share groundwater and surface water in rural communities, especially for irrigation, have a long history in most industrialised countries as well as developing countries. They generally incorporate a mix of public and local community involvement as well as civil works planning and decentralised transfers of rights, which are often informal. Meanwhile, true water markets are rather rare, often limited to countries and regions with specific histories in water management. A great variety of regulatory and property rights regimes exist related to water. In some countries, recent developments in water rights trading have not been limited to water for agriculture but expanded to urban needs. Some of the world's most developed water markets are found in the western states of the USA. However, even in this country, despite the efforts of EPA, development of emissions trading for water pollution control has had mixed results due to the difficulties in addressing non-point source pollution. In Australia, where water trading programmes have also been under development for some time, the emerging recognition of the water rights for non-consumptive (i.e. in-situ) uses poses a challenge, and is now reflected in the government policy. These include, for instance, ensuring minimum flows in rivers in order to preserve environmental quality and amenities. There are some difficulties in transfer or trade of water rights, as water cannot be considered a standard commodity, since it combines attributes of private, common and public goods. They also illustrate how local environmental constraints may rightly limit potentials for trades, especially when transfers from agriculture to other sectors, and from rural to urban areas are at stake. In other countries such as Mexico, it is not infrequent that Water User Associations (WUAs) take an intermediary position between administrative authorities and end users in water management. Most trades then take place within a WUA, which can be considered as a "bubble." Meanwhile, in several countries, governments have recently developed new programmes to expand or modernise water management frameworks, including water trading. From these experiences, some key points are to be stressed here:

- Social acceptability of application of the TP concept for water management is certainly more limited than for other sectors, especially for water use. This is explained by several factors: (i) the potential significance of local environmental impacts of transfers (water is a life-supporting resource), which expresses itself in increasing demands for return flows constraints; (ii) perception of water as an essential resource that local communities do not wish to give up, except for temporary transfers; (iii) in some cases, non-separation of water rights and land ownership (then access to water rights would imply buying the land);
- Water is not a homogenous good and cannot be physically transferred without taking account of physical and geographical constraints. Most transfers are local by nature. Meanwhile, some water pollution rights trading schemes could be developed between point and non-point sources within the overall water quality management of an area, but this needs further work;
- Water trading can only occur within rather restricted geographical zones with common ecological and institutional significance, such as sub-river basins regulated by appropriate water institutions (Basin Authorities and WUAs);
- Specificity of water resources (multi-dimensionality) makes it difficult to treat them as a private good. A necessary condition for the development of water markets is, then, strong institutional frameworks for water management at micro and regional levels to ensure that environmental and social aspects of water usage are taken into account; and
- New developments in water management and trading should be incorporated into existing water institutions.

11. See Chapter 7, A. Kraemer, E. Interwies, and E. Kampa (2001), Tradable Permits in Water Resources Protection and Management - A Review of Experience and Lessons Learned.

3.3 Experiences in transport[12]

Quite evidently, emissions from transport involve mobile sources. In western societies, mobility has always been considered as a key attribute of an open and free society. The social concerns as well as technological difficulties have restrained environmental policies, such as urban tolls or road pricing, which would limit mobility and access to space. The approaches in applications of the TP concept in transport sector experimented for several years have been indirect, taking some forms of the baseline-and-credit model. Namely, these "averaging" programmes have obliged car manufacturers to meet targets based on various technical parameters related to emissions rates and fuel efficiency. They have included standards for gasoline composition and the mean gasoline consumption per kilometre for all models produced, rather than for each model. This solution is more in line with the traditional regulatory approaches, and could supplement the type of voluntary commitments, for example, by the European association of car manufacturers.

This approach, focused on performance of vehicles, has limitations too: it does not provide a complete set of incentives for vehicle users, nor guarantees the total amount of emissions and their spatial distribution. Particularly, it does not address the issue of possible pollution "hot spots".

Another problem is related to the multiple pollutants generated by fuel combustion: CO_2, NOx, VOC, and particulate matter (PM). If increasing energy efficiency per kilometre reduces all pollutants altogether to a certain level, at one point some trade-off would become necessary among targeted pollutants, as technical solutions to decrease one pollutant (e.g. CO_2) can imply an increase or less decrease of others (e.g. NOx). This situation raises the issue of completeness and appropriateness of policy instruments targeting the same sources (emissions from vehicles) in order to achieve coherent results. What kind of interactions can be set between several TP schemes targeting different pollutants from the same sources? What about the interactions of one TP scheme for one pollutant with other policy instruments focusing on other pollutants from the same sources?

One recent experience in Europe, namely the Ecopoint programme in Austria, shows another approach to TP schemes in the transport sector. Although no quota trading is allowed, this programme introduced for the first time a cap on transit flows of EU trucks weighing more than 7.5 tons. One Ecopoint, i.e. one allowance, corresponds to the right to emit 1 gram of NOX per kWh. Each year, the total amount of Ecopoints is allocated by the European Commission to member States on the basis of 1991 traffic shares. The annual cap decreases each year so as to achieve a 60% emission reduction by 2003. Since January 1998, management of the system is electronic. Each truck is mounted with a small electronic device that transmits information on identify of the truck and its remaining Ecopoints, allowing the information to be read each time it crosses the border. In fact, this programme shows that technical and managerial solutions exist for a TP scheme directly applied to point sources such as transport flows within a region. An interesting next step would be to make the Ecopoints-type allowances tradable.

3.4 New developments: renewable energy certificates[13]

A number of countries (Australia, Belgium, Denmark, Italy, Netherlands, USA) are introducing and others (Sweden, UK) are considering tradable renewable energy certificates (TRECs) as policy instruments to support the development of alternative renewable energy sources in competitive electricity markets. In most cases, new obligations are imposed on electricity suppliers; they are formulated as minimum shares of renewable energy sources in total electricity output. In

12.. Chapter 5, C. Raux (2001).

13. Chapter 4, R. Baron (2001).

parallel, certificates are issued as a proof that a certain amount of electricity has been generated from renewable sources and supplied to the grid by eligible electricity producers. Therefore, suppliers can buy TRECs from renewable energy producers without having to produce renewable energy themselves.

This new development raises interesting points related to:

- spontaneous differentiation of TREC sub-markets according to the nature of energy sources, i.e. the types of renewable energy source generating TRECs (some operators may be reluctant to use some TRECs if the status of some renewable energy sources are controversial, e.g. wood products);
- linking or not linking national TREC systems, i.e. whether or not to allow trades of TRECs under different national systems. Linking a national system to foreign ones may stimulate technological breakthrough by foreign competitors, leading to loss of local external benefits as well as national advantages in renewable energy related to GHG emissions. Also, it can lead to distortions in competition due to unequal levels of subsidies for renewable energy across countries. Moreover, linking will make the weakest element of a national system (for instance, penalty or monitoring) the weakest one of the overall international TREC system. Thus, linking of different national systems should be based on specific conditions ensuring equivalencies among the systems involved;
- TRECs as transitional instruments, from other pre-existing policy instruments (e.g. subsidies and feed-in prices, energy taxes) to a future state of incentives within electricity markets that would have fully incorporated climate change policies;
- how TPs can address several objectives simultaneously; TRECs combine the objectives of cost-effectiveness, promotion of sufficient diversity in renewable energy sources, and also long-term strategy of technological development by discouraging players from prematurely exiting the business of renewable energy development; and
- whether or not TRECs may be made interchangeable with CO_2 emission permits, as substituting renewable electricity for that generated from conventional sources (i.e. coal and gas) also cuts CO_2 emissions. If TRECs are not made fungible and there exist distinct TPs for each goal, one same action of substituting renewable energy for carbon energy sources could give rise to rights under two or several types of TP schemes, TRECs, CO_2 allowances and possibly SO_2 or others. If conversion of TRECs into GHG emission allowances is permitted, the latter could not be used to meet an absolute cap, since TRECs currently in existence are defined in relative terms.

3.5 New developments: municipal solid waste management[14]

There are two UK programmes that apply the TP concept in municipal waste management. They are aimed at restricting landfilling, and encouraging collection and segregation of waste for material recovery and recycling. They show how promising TPs can be in this new field, in spite of doubts expressed earlier. As for TRECs, the goal with material recovery is not to decrease the level of activity of a market, but to increase it. Policy is then about obligating the demand side of the market to achieve a minimum required level. The first programme, Packaging Waste Recovery Notes (PRNs), is somewhat an accidental TP programme, as it was not conceived as such when it was first introduced in 1998 by the guidance document from the Environment Agency to facilitate implementation of the Producer Responsibility Obligations (Packaging Waste) Regulations of 1997. The extended producer responsibility was translated into the latter regulation as a minimum requirement that increases

14. Chapter 6, R. Salmons (2001).

annually (e.g. 56% recovery of packaging wastes for 2001), and obligated businesses to provide evidence that the targets had actually been met. For this purpose, PRNs were to be issued by accredited waste material re-processors. The firms involved at different stages of the packaging chain share responsibility on a fixed percentage basis. Firms were allowed to join this collective compliance scheme by paying an annual fee, an opportunity that a vast majority of them seized. After several months, an Electronic Environment Exchange was privately created, attracting increasing attention. It resulted in a rapidly increasing number of trades in PRNs, reaching 100,000 tonnes in 2000. Although prices paid by re-processors for waste packaging materials have increased since the introduction of the programme, allowing additional revenues to go to waste collectors, actual compliance costs have been dramatically lower than expected. Meanwhile, the minimum recycling targets have been met each year (except for plastics whose recycling rate was slightly below target in 2000). This case demonstrates how a market can emerge once an adequate regulatory infrastructure is in place (i.e. standardised PRNs as proof of compliance).

In order to enforce the EU landfill Directive, the UK government is now considering a TP programme to limit biodegradable municipal waste going to landfills. This programme has gone through a phase of consultation and design adjustments. According to the latest consultation document, permits will be allocated free of charge to Waste Disposal Authorities (WDAs) with uniform reduction rates for all WDAs based on the current levels of biodegradable wastes handled. To implement this cap and trade programme, allocation of permits will be defined for the entire time horizon, up to 2020, with linearly decreasing annual caps in order to achieve the EU targets at interim and terminal dates. The permits can be traded among WDAs, and the latter can also decide to form bubbles within which transfers would be made without payments.

In both cases of TP use in municipal waste management described above, it seems that issues of market power, often considered negative for TP schemes, would not have significant importance in practice, except maybe for plastic reprocessing where price discrimination could occur.

3.6 New developments: greenhouse gas control[15]

In anticipation of an international GHG emissions trading programme under the Kyoto Protocol, several countries have considered (e.g. Australia, Canada, France, the Netherlands, New Zealand) or have plans (Norway, Sweden) for introducing domestic provisions for GHG or CO_2 emissions trading. Denmark and the UK have introduced their domestic trading schemes in 2001 and 2002 respectively, while the EU proposal for a new directive has recently been adopted, for introduction of an EU-wide GHG emissions trading programme for a list of industrial sectors in 2005. Discussions raised by these projects and proposals raise various issues including the following:

- How could TPs best interface with taxation, since several countries are already using or envisaging to use carbon taxes as a key policy instrument? Combining TPs with taxes may be achieved in several ways. Voluntary participation in a TP scheme may be introduced as a means to opt-out of specific tax obligations, or in reverse, paying a tax may be an appropriate way out of a TP programme (concept of safety valve) if, for unexpected reasons, emissions trading is trapped into excessively high prices (i.e. prices that impose marginal abatement costs far in excess of what is considered economically sound for GHG emissions abatement). Also, as for any other assets, holding of unused TPs may also be taxed in order to avoid strategic hoarding of TPs and maintain sufficient liquidity in the TP market.

15. Chapter 3, K. Kitamori (2001).

- What links could be established between GHG emissions trading and measures addressing other pollutants contributing to local air pollution (e.g. NOx, SO_2, PM)? Could GHG emissions trading supplement or substitute emissions trading for other pollutants?

- What are the necessary conditions for integrating domestic TP programmes into a regional or international trading scheme? Harmonisation would be required for a number of variables across domestic schemes, including, for example, the definition of the permit or allowance and its validity period, agreed methods of monitoring, minimum penalties and enforcement measures, and rules of initial permit allocation for extension of productive capacities and new investments.

- To what extent can TP programmes be developed on a voluntary basis? Would trading schemes based on voluntary participation face problems of credibility? Under the voluntary UK Emissions Trading Scheme (ETS), for example, a special auction procedure has been introduced whereby firms are to tender GHG emission reduction proposals for a given amount of public subsidies, to make participation in the scheme attractive. This procedure is established under an annual budgetary cap of public expenditures devoted to such subsidies. It may be questioned if this procedure is compatible with the polluter pays principle and the EU State aids law. For efficiency and equity reasons, this approach would imply that other policy instruments (e.g. Climate Change Levy) aimed at GHG emissions would need to be applied to firms that choose not to participate in the ETS. Hence, the issue of combining TPs with other instruments becomes important, which could add to administrative burden.

- Can two markets for credits with relative targets and cap-and-trade allowances with absolute targets coexist within one programme, on the basis of complete fungibility between the credits and allowances? One problem with relative target-based schemes is that they cannot guarantee an absolute level of total emission abatement, and thus unlimited trades between the two systems would risk the ability to meet the overall absolute cap.[16] This drawback is considered minor by some experts, as even schemes based on absolute targets usually do not cover all emissions from all sectors in a given country anyway, and the international emissions trading scheme would require a country to deliver its overall national GHG emission reduction target in absolute terms under the Kyoto protocol.

- What is the best pathway to introduce GHG emissions trading (i.e. which sectors and gases to start with)? A trading scheme may evolve over time, as it develops from a pilot phase to a mature phase, with respect to key design issues such as sector and gas coverage, relative or absolute targets and voluntary or mandatory participation.

4. Reflections on past experience

The first attempts in TPs were aimed at granting flexibility to decentralised agents in making their technological choices, and avoiding absolute obstacles to new pollution abatement investments in "non-attainment areas" (highly polluted urban areas which did not meet national air quality standards) in California. The Acid Rain (sulphur allowance trading) Programme introduced as part of the reform of the US Clean Air Act in 1990 was remarkable for the change in scale and objectives. The policy went further towards a strong articulation of environmental and economic performance of the

16. The UK ETS has chosen a solution of a "gateway": relative target-based credits can be bought by firms in the absolute target sector only if the total amount of relative target-based credits, sold into the absolute target market and those newly proposed, does not exceed the total amount of absolute target allowances sold into the relative target market.

whole power generation sector, making sulphur pollution a key component in various management decisions in this business. Today, the efforts in research and development towards cleaner technologies have not yet vanished, still serving as the main driver of these credit trading systems. Furthermore, the recent developments in technological progress in this area promise to feed into more active interactions of supply and demand in the markets for environmental technologies and services.

With waste management, transportation, and green energy, the ultimate issue is either to downsize demand for some services (landfills, use of private vehicles in some areas and general demand for transport) or to sustain the development of new markets (green energy, recycled materials, low or zero emission vehicles). Entirely new markets for goods and services could potentially arise from the introduction of TPs in such contexts. It is certainly a greater challenge than just shifting technology choices for environmental motives.

In several cases in the past, the actual trade volumes recorded under TP schemes have been well under the expected levels (Kraemer et al., 2001).[17] In some cases, TP schemes introduced by the authorities have led to no transfers or trades at all. Does it mean that those cases proved TPs to be inefficient? Such schemes may have been ill designed and would presumably gain from reform. Or, the case could be that the initial permit allocation was optimal and did not require trades or transfers on the part of participants. In any case, it is to be stressed that, for decentralised agents, having the capacity and opportunity to take part in transfers or trades of rights is in itself valuable, even if very few or no transfers actually take place. This capacity and opportunity to trade bring security in procurement, and should induce more dynamic attitudes in making investment decisions. Thus, TP schemes may be a useful tool even when not many transfers or trades are observed. Here the argument is the same as that of Weisbrod (1964)[18] in defence of public management of natural parks: among the people willing to pay for preserving the option of a future visit to a park, some will never actually take the option. In the case of TPs, realised transactions alone do not fully capture the total social benefits of the instrument.

In spite of the progressively extensive use of the instrument, there has been persistent ambivalence and scepticism for TPs in the eyes of various stakeholders regarding the concepts of transfers, trades, and markets. For one part, reservation is rooted in general *a priori* views about the market and the environment. For another part, it is rooted in the debates on actual experiences:

1) trades of rights in the marketplace may lead to a concentration of property and market power, denying small business and poor people access rights to necessary resources (e.g. water); therefore, minimum social considerations should be given against concentration of property and market power;
2) in some instances, TPs may be seen as a way to circumvent meeting required targets under an environmental policy, as it seems to be the case with the evolution of the Zero Emission Vehicle (ZEV) programme in California;[19]
3) by alleviating immediate individual technological constraints, TPs may erode short-term incentives to make necessary investments to prepare for longer-term environmental performance,[20] and;

17. Chapter 7, R.A. Kraemer et al.

18. B.A. Weisbrod (1964), "Collective-consumption services of individual-consumption goods", *Quarterly Journal of Economics*, 78, August.

19. See Chapter 5, C. Raux.

4) transfers involved in TP schemes may have undesirable local environmental consequences (e.g. case of transfers of water abstraction rights causing over-abstraction in one area, or localised air pollution hotspots).

The design of a TP scheme should be examined accordingly for the relevance of these potential drawbacks which should be addressed as appropriate. In contrast, some of the views against transfers and trades could be considered as reluctance to co-operate with other peoples or regions whose needs for environmental management cannot be met with their own local resources, either permanently or in exceptional occurrences.

Experience also demonstrates that it may take a long time to fully develop a TP programme, mainly because of constraints in acceptability of, and familiarity with the instrument. Having well-functioning TP schemes does not only depend on design decisions on the part of managing authorities, but also on the capacity and willingness of the decentralised agents to look for and make use of trade opportunities. Introducing TP schemes is not a simple change comparable to the introduction of more stringent regulations. It involves a change in patterns of social and economic relations among decentralised agents, and between them and regulatory authorities. Therefore, introducing TPs should be considered as a social change process. Accordingly, development of TP schemes would benefit from guidance by a long run strategy.

In this regard, one question often raised is to what extent TPs can serve as a long-term instrument, or only a transitional one. In fact, both cases can be found. If the goal of protection of the global climate is pursued by the international community in the long run, TPs can be a core instrument used over time, from an initial 5-year period to next 5-year periods. In other cases, as for TRECs (see earlier discussion as well as Chapter 4), it may be assumed that incentives are required in the short to medium term to accelerate the development and diffusion of alternative energy sources. However, in 20 years or so, the combined effect of technological progress and a stabilised regime of incentives to protect the global climate would be sufficient for alternative energy sources to be competitive in the energy market. Thus, TRECs could be considered as a transitional tool. There are more technical reasons to see some TPs as revisable instruments. As in the case of lead phase-out in gasoline (see Chapter 5 on transport sector), the goal of the programme was reached in the medium term. Alternatively, the means necessary in the long run to meet an environmental objective may be such that it will eventually dictate technology choice. In such a case, there would no longer be room for the flexibility afforded by TPs.

A strategic view of the long-term development of TP schemes will have implications for the choice in initial areas of application and in design options (e.g. 'baseline and credits' versus 'cap and trade', absolute vs. relative targets, type of initial allocation method, etc.). Hence, the development of a TP regime should be seen as evolving from initial steps to maturity.

In this regard, it may be useful to consider prerequisites for subsequent developments of TPs. For instance:

- The organisation in charge of the policy instrument should be familiar with economic theories and concepts, and concerned with cost-effectiveness. For local authorities, it may involve delegation of managerial competencies in some domains (e.g. waste collection, water management) to an independent agency or authority.

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20. However, there are also views that the pressure to invest in new technologies depends more on the stringency and legal status (i.e. binding or voluntary) of the environmental performance target, rather than on the instrument chosen to implement a given target.

- A market for TPs needs to be regulated by a competent authority, tasked with tracking information on environmental performance (e.g. levels of emission or abstraction) and transfers, verifying compliance, organising permit allocation, preparing strategic adaptation of rules, and ensuring confidence in the fairness of the market. In some cases, this authority may be placed in an existing administrative service without altering its credibility. In other cases, it would preferably be an independent administrative authority or an association of resource users (e.g. for irrigation water). Then, institutional reforms introducing such new bodies in the regulatory landscape would be a prerequisite for subsequent development of TP schemes.
- To begin with, "bubbles" may be a first step towards fully developed TP schemes, since bubbles involve transfers without external trades. Similarly, granting companies operating multiple plants the possibility of intra-firm flexibility in achieving compliance would have the same effect.
- Development of trading requires a generally accepted and relevant definition of the good to be traded. Beyond simple cases where the nature provides the common definition (e.g. CO_2 molecules), it is necessary to elaborate equivalencies that are scientifically acceptable and relevant for practical purposes. This step incorporates elements of convention, as shown by the six GHGs considered under the Kyoto protocol or PRNs (Packaging waste Recovery Notes) in the UK system (Salmons, 2001 in Chapter 6).

The social dimensions of TPs often lead authorities to carry out extensive stakeholder consultations in order to obtain a sufficient level of understanding and acceptance of a new programme. This process could certainly influence the design of a TP scheme. While acceptance-building may be critical in development of the new instrument, it could undermine its capacity to reach economic efficiency. For instance, consulted business organisations may insist that the rules of a proposed TP scheme not affect the pre-existing state of competition, no matter how unequal the levels of external diseconomies incurred by different products and technologies may be (Godard, 2000).[21] Another effect of stakeholder consultation is that, despite the strong advocacy for clear, simple and pragmatic TP schemes, some recent ones, both proposed and in place, are rather complex in design and administration, as exemplified by the UK Emissions Trading Scheme for GHGs. Part of this programme is built on a proposal elaborated by industry, while the other part responds to government objectives. More attention should perhaps be paid to the process of stakeholder consultation itself, and to the design variables that should not be compromised by this consultation.

Looking at past experience with the benefit of hindsight shows a structural opposition in the development of TPs between learning and adaptability on one side, and stability and predictability of the rules of the game on the other. For new instruments in early phases, it is necessary to experiment and adapt what experience reveals to be inappropriate design and rules. At the same time, permanent or frequent changes of the rules have proven to have negative effects on the credibility of TP schemes, resulting in poor achievements. There are two different reasons for such a result. On the one hand, by participating in a TP scheme and actively committing themselves to trading, regulated agents reveal information on their marginal abatement costs and this information can be subsequently used against them by opportunistic authorities who then change the rules by surprise. On the other hand, the participating firms may hope that by not complying, they may be able to renegotiate the rules and strongly influence the new ones. Revisions that are too frequent and not based on precise rules, the result of which can be foreseen by participating firms, undermine the credibility of a programme, as is shown by successive revisions of the California LEV and ZEV programme. Many difficulties can be

21. O. Godard (2000), "Dialogue and economic efficiency: two antagonistic goals for environmental policy-making? Lessons from the French packaging waste management system", in M. Skou Andersen and R. Sprenger (eds), *Market-based Instruments for Environmental Management. Politics and Institutions*. Cheltenham, Edward Elgar, 89-110.

attributed to immediate or future uncertainty of regulatory rules (rights and obligations) or to discretionary interventions by public authorities in the running of the scheme.

5. Challenges for TPs

Addressing failures

From an economics viewpoint, TPs are conceived to overcome several types of failures in environmental policies: (a) market failures (uncovered externalities); (b) governmental failures (faults in incentives incurred by policies based on regulatory and financial measures, e.g. those not touching intended agents or sending perverse incentives); and (c) information failures (physical flows of pollutants and abatement costs are not well known, combination of which can lead to approximate policies). Type (a) failures are addressed through TPs by introducing new objective and quantitative constraints in the name of public interest. Type (b) failures are addressed by the opportunity given to obligated agents to make a trade-off between actual pollution abatement or resource conservation actions, and buying the excess permits/credits incurred by such actions by others via TP transactions. Type (c) failures are also addressed because, on the one hand, TPs cannot work without a credible quantitative monitoring and, on the other hand, TP trades, whenever they take place regularly, give precious indirect information on compliance costs through mean trading prices.

Defining target groups

To be successful in overcoming failures, the design of a TP scheme should provide appropriate answers to different challenges and issues. The first one is to deliver appropriate incentives to the different categories of agents involved at different points of the material flow system generating the environmental problem. For instance, in the case of domestic waste management, it calls for reflections on incentives given to producers who should reduce waste at source, at the stage of product design and decisions about packaging; to retailers who should have a role in decisions on packaging use and take initiatives to set-up deposit facilities for consumers; to consumers who can contribute by segregating waste for collection and bringing back packages; to collectors, who decide on coverage and techniques of collection; and to reprocessors who define requirements about quality of recycled materials, prices paid, and may commit themselves to quantities and price. In the case of GHG abatement policies, some emission sources (e. g. methane from landfills) may represent a significant potential in addition to traditional measures targeting industry and power generation. To set-up a coherent system of incentives, it will be necessary either to think of specific instruments required for each target group or to consider the probability that ordinary economic mechanisms (market relationships) will adequately transmit policy signals introduced at one point of the system to all others.

This last question is at the heart of the discussion about upstream or downstream allocation of GHG emission permits. An upstream scheme obligating energy producers and importers would cover most GHG sources (except from forestry, agriculture and waste management), but would depend on the capacity of the markets, all along the chains down to final consumers, to provide appropriate incentives so that they adapt choices and behaviours efficiently. Imperfectly competitive markets could lead to price discrimination and regulation due to inadequate consumer demands if a quantitative cap is to be absolutely met. A downstream emission permits scheme, e.g. for car owners and domestic/residential emission sources, would provide direct incentives to final consumers to change their behaviours, but coverage of GHG emissions would only be partial, leaving out industry and power generation sectors.

Voluntary participation and "opt-in" provisions

The completeness of incentives delivered to all agents has an influence on the overall achievement of policy goals. Thus, whether a TP scheme is based on voluntary participation or has opt-in provisions becomes important. An economically coherent environmental policy cannot be set if only voluntary agents are taking measures to achieve an environmental goal. The only certainty is that a policy made of voluntary commitments will not be too damaging to the economy, but no great achievements can be expected in terms of environmental quality. Thus, the issue of voluntary participation can only be discussed in more specific contexts:

- when voluntary action is seen as a first step to experiment new ideas and policy tools before a public policy is developed and implemented; in such circumstances, early experiments may be profitable for business as well as for public authorities; some "light" policy tools may help the development of such initiatives, for example, establishment of a register;
- when a new compulsory TP scheme is introduced to address an environmental problem for a well-defined set of agents that constitute together the main source of the problem; by assumption, criteria of activity or scale are used to mark out subgroups of agents subject to the scheme and those to be excluded. The latter may be regulated by other types of policy instruments or their environmental impact is insignificant compared to the administrative costs. The excluded agents may be included into the programme at a later stage. Then opt-in provisions may allow them to anticipate a future obligation, which may be positive for environmental and economic reasons. Another case of voluntary participation is one in which obligated agents can substitute some basis of obligation (e.g. emissions from a given plant) with another basis (another plant). As mentioned earlier, the latter should be considered with caution since it may alter the outcomes of a TP programme. Generally speaking, voluntary opt-in provisions may bring into a scheme more permits than allowed within the target set by the public authorities, or bring in what has been called 'hot air' in the context of the Kyoto Protocol, i.e. emission rights that do not result from specific abatement measures. Similarly, if conditions of voluntary entry are not carefully sets for new participants, it may add unpredictability to the permit market, making it less attractive for existing participants.

TPs in policy mixes

All existing and proposed TP programmes are never used alone; they are always combined or supplemented with other policy instruments, such as command and control regulatory measures, taxation, voluntary agreements, subsidies, and information disclosure to name the most important ones, aimed at the same environmental problem. Hence, the design of a TP scheme cannot be framed as an exercise in building an isolated tool, but a tool whose result will depend on wanted and unwanted interactions with other instruments and incentives acting on the same target groups. Policy mixes may embrace a sectoral division of labour, reserving TPs for some sectors and using other instruments for other sectors, or a division according to scale or other distinctions within the same business (small versus large firms). It may also lead to an integrated scheme incorporating technology standards (BAT), voluntary commitments, taxes and TPs in one unique machinery targeting the same group of agents. How to manage all these possible interactions among these instruments is a real challenge in policy-making.

A case in point concerns controlling different pollutants emitted by one unique source, mainly combustion of fossil fuel by fixed or mobile sources: CO_2, SO_2, NOx, VOC, and PM. Several options are open:

- setting the TP target on the pollutant subject to the most stringent constraint, assuming that the resulting levels of abatement of this specific pollutant will result in sufficient abatement of

the others; this option can work if technology cannot address the different pollutants separately;

- developing multiple TP schemes, one for each pollutant, and keep them separate; in that case, economic agents responsible for one source will have to comply simultaneously with the requirements of the different TP schemes in order to have the right to burn one unit of energy input;
- developing one integrated TP scheme, while allowing inter-pollutant trades on the basis of offset coefficients expressing the relative intensity of marginal damage imposed by additional flows of each type of pollutant; such a solution may be envisaged if, from an environmental viewpoint, offset coefficients can be sufficiently stable to be practical; there are other conditions too, as it is exemplified by the challenge of integrating credits/allowances based on both relative and absolute targets into one unique TP scheme.

Initial allocation of rights

The initial allocation of obligations and property rights has an economic influence on surplus distribution between obligated agents and those who receive an allocation of rights. This is so firstly through the distribution of scarcity rents to some agents, even when there are second-order competitive markets for TPs within an overall cap. For instance, in the case of waste management, such distributive effects depend on whether quantitative limits are set as a cap on landfilling -limit on the supply of landfill services- or as a cap on the right of municipalities to send wastes to landfills. In the former case, landfill operators could gain a scarcity rent from municipalities that demand their services. In the latter case, municipalities will shift down their demand for landfill services and landfill operators will bear profit losses. Secondly, whenever markets are imperfectly competitive (i.e. existence of market power due to monopoly or monopsony), initial allocation will affect both economic efficiency and distribution of profits. These are the two reasons for considering the industrial structure in the sector concerned by a new TP programme and distributive effects of initial permits/rights allocation.

Part II

EX-POST EVALUATIONS OF EARLIER DOMESTIC TRADEABLE PERMIT SCHEMES

Chapter 2

EX-POST EVALUATIONS OF TRADEABLE PERMITS PROGRAMMES

by

Stephen Smith
University College London

1. Introduction

Tradeable permits have the potential to achieve environmental goals at less economic cost than conventional "command-and-control" regulation. They allow regulated firms flexibility in the way in which pollution reductions are achieved, enabling firms to select the most cost-effective approaches. In addition, they provide a continuing incentive for regulated firms to search for innovative approaches which permit further reductions in emissions, while conventional regulation provides negligible incentive to do more than the minimum needed to achieve regulatory compliance. These potential advantages of employing tradeable permits - in terms of "static" and "dynamic" cost reductions - are well-known elements of the wider theoretical case for making greater use of market mechanisms (tradeable permits, emissions taxes, abatement subsidies, etc) in environmental policy. In addition, tradeable permit regimes are distinguished from most other environmental market mechanisms by the greater certainty which the environmental outcome can be achieved. Unlike environmental taxes, where the environmental consequences depend on polluters' responses to the tax incentive, tradeable permits can place a fixed limit on total emissions, and in certain circumstances this may be a significant advantage.

Nevertheless, there is always a significant step from theory to practice in policy design. Recent OECD work on the evaluation of environmental policy instruments has sought to identify systematic evidence on the performance of different instruments in practical applications, to extend policy makers' understanding of the advantages and disadvantages of different instruments beyond the purely theoretical. In the past decade, actual applications of market mechanisms in environmental policy have spread rapidly, throughout the OECD area and, increasingly, beyond. These practical applications have generated a rapidly-growing body of evidence on the actual performance of a wide range of market mechanisms in practice.

The early evidence about the performance of tradeable permit mechanisms in practice was a considerable disappointment. For some time, the Fox River scheme for transferable discharge permits, introduced by the Wisconsin Department of Natural Resources in 1981, provided the sole clear-cut example of a tradeable permit regime in operation. This scheme, covering a number of paper mills and municipal wastewater treatment plants discharging effluent to a heavily-polluted stretch of the Fox River, had the potential to achieve considerable cost savings compared to a fixed central directive requiring uniform emissions. A simulation study by O'Neill, David, Moore and Joeres (1982) showed that there should be considerable scope for reducing total abatement costs through the reallocation of emissions between sources. Since abatement costs differed between sources by a factor of four, there would be considerable cost savings from such reallocation. The fact that, in practice, the introduction of the scheme was followed by only a single trade, stood in stark contrast to the predicted theoretical gains, and provided a clear warning that there could be a considerable gap between advance forecasts and practical outcomes. There has been much subsequent discussion of the reasons for the low level of trading within the Fox River scheme - some of which has pointed to particular practical features of the

scheme that may have inhibited efficient trading. Nevertheless, it was not for some time that further practical applications of tradeable permits became available, so that it could be judged whether the disappointing outcome from the Fox River scheme reflected the particular circumstances of that policy application, or reflected a more general failure of tradeable permits in practice to live up to expectations.

The last decade has seen a massive expansion of tradeable permit applications in the United States (EPA 2001). Some of the most significant of these have been:

- lead credit trading, as part of the US phase-down of lead in gasoline;
- development of various forms of emissions trading at state level;
- CFC production allowance trading;
- heavy duty motor vehicle engine emission trading, and the application of corporate average fuel economy (CAFE) standards for motor manufacturers;
- SO2 allowance trading within the US acid rain program.

Much can be learned from these applications, if appropriate priority is given to research to document and evaluate the outcomes.

Recent OECD work (including *Evaluating Economic Instruments for Environmental Policy, 1997*) has encouraged more extensive and systematic evaluation of environmental policy instruments, to facilitate policy learning between countries, and to stimulate reflective processes of policy re-evaluation and improvement within countries.

A number of *ex post* evaluations of domestic tradeable permit schemes are now available, almost all reflecting US applications, since these still constitute the overwhelming majority of significant policy applications of the tradeable permit approach. However, the evaluations that have been conducted differ widely, in terms of their institutional origins, coverage, methodological approach and findings. The evidence and lessons from the tradeable permit applications which have been studied need to be drawn together and thoroughly assessed, as an input to the development of new emissions markets (for CO2 and other GHGs, NOx, etc) currently being considered by many member countries. In addition, however, there are lessons also to be learned from the studies about the process of evaluation itself. What should *ex post* evaluations aim to cover, and what approaches can be adopted?

This paper provides an overview of the issues involved in evaluating domestic tradeable permit schemes in practical operation. The paper is in five main parts. Following this introduction, Section 2 reviews the benefits that may be obtained from more systematic *ex post* evaluation. A particular issue is the "value added" from *ex post* evaluation, over and above what is known about the effects of tradeable permits in advance of practical experience. It is argued that *ex post* evaluations will maximise value added if research effort is carefully targeted, rather than through a broad and comprehensive overview of all costs and benefits. Section 3 develops a framework for the evaluation of tradeable permit policies, in the form of a check list of issues and evaluation criteria. This discussion draws heavily on the outcomes of earlier OECD work on this topic, most notably the 1997 report *Evaluating Economic Instruments for Environmental Policy*. Section 4 then discusses methodological issues in *ex post* evaluation research, including the key issues of defining an evaluation "baseline" against which the effects of the policy can be assessed, issues of timing and institutional location, etc. Section 5 then reviews a number of the available evaluation studies for tradeable permit applications, in the light of the preceding methodological discussion, and looks at the approaches adopted and

results obtained. These studies differ both in the types of tradeable permit applications studied, and in methodology. They comprise an early academic evaluation of specific arrangements for emissions trading in the Los Angeles basin in California, a study of the completed episode of trading to support the US phase-out of lead in gasoline, a major recent study of the US-wide trading scheme for acid rain, and an evaluation report on arrangements in a number of US states for emissions trading for VOCs. Section 6 concludes the paper with observations on the lessons learned from the evaluation studies, and recommendations for the direction of future work in the area of *ex post* evaluation of tradeable permit programmes.

2. The value of evaluation

As the 1997 OECD report *Evaluating Economic Instruments for Environmental Policy* observes, systematic analysis, or "evaluation", of practical experience about the performance of economic instruments can provide valuable information about their performance. More extensive evaluation evidence would be beneficial within the policy process in a number of ways:

- Evaluation evidence on the performance of policy instruments can help to improve the administration of current policy, and can contribute to a process of policy reappraisal, modification and improvement in the light of experience;

- Evaluations can also improve the choice of instruments in future policy, by demonstrating how different instruments perform in specific contexts;

- Evaluations can provide evidence on the functioning of the political and policy processes, to ensure that they translate policy intentions into practice as effectively as possible. The knowledge that policies may be subject to future evaluation can act as a brake on the development of poorly-justified or cynical policy measures.

- Evaluation may also contribute to better communication with stakeholders and the public about the purpose, operation and effects of policy.

In each of these ways, evaluation studies can contribute to better design and implementation of environmental policies in the countries concerned. There may also be important benefits to other countries, which can learn from the practical experience of countries which have implemented particular policies. In the case of tradeable permit mechanisms, nearly all of the large-scale practical applications to environmental policy have occurred in the United States. Evaluation evidence on the performance of emissions trading and other tradeable permit mechanisms in the US could contribute significantly to the international diffusion of tradeable permits, and to more efficient policy learning about the contexts to which they may be most effectively applied, and the best form for tradeable permit systems to take.

This paper takes a wide view of "*ex post* evaluation", and does not restrict its scope to those studies intended as a systematic and comprehensive assessment of all relevant elements bearing on the performance and outcomes of tradeable permit applications. Instead, it includes within the category of "*ex post* evaluation" a broader group of research studies which aim to shed light on aspects of the performance of tradeable permit policies. Some of the studies included in the discussion focus on only a single dimension of performance - for example, a study of abatement technology adoption in an industry regulated using tradeable permits. The key criterion is not the portrayal of a particular study as an overall "evaluation", or the extent of its coverage in terms of the inclusion of estimates for all relevant costs and benefits, but the extent to which the lessons that can be drawn from the study are the results of experience, and could not have been learned without observing instruments in actual operation.

There are important things that can be learned from the practical operation of tradeable permits in environmental applications that cannot be learned without such experience. At the same time, however, many of the properties of tradeable permit mechanisms can be well understood, in advance of any real-world use of tradeable permit mechanisms, as a result of theoretical analysis, and *ex ante* assessment, forecasting and simulation.

2.1 What do we already know?

Ex ante empirical studies of the costs of pollution abatement under tradeable permits, in comparison with a "command and control" baseline, are widely cited in policy discussion. Such studies typically show that there are substantial potential economic gains from using a market-based policy instrument (such as tradeable permits) which would efficiently allocate emissions reductions between polluters, rather than the type of equal-abatement rule which frequently results from conventional command-and-control regulation. Tietenberg (1990) surveys a range of estimates.

These studies use data on the pattern of abatement costs across different polluters to calculate the total costs of achieving a given level of pollution abatement, on different assumptions about how the aggregate abatement requirement is divided amongst different polluters. Typically, in analysing pollution problems where there is no spatial or temporal damage from emissions, these studies compare the costs of achieving the efficient (in the sense of cost-minimising) pattern of abatement across polluters, with the total abatement costs when the reductions in pollution are distributed across polluters according to particular rules, representing the operation of certain types of command-and-control policies[1]. Thus, for example, policies may compare the efficient pattern of abatement with the costs of achieving the same reduction in pollution by requiring all polluters to reduce emissions by the same percentage. Where polluters face widely-differing marginal costs of abatement, the costs of the equal-abatement policy will be substantially higher than the costs of the efficient policy. If tradeable permits achieve the cost-minimising pattern of abatement, then these studies show the net economic benefit from achieving the given level of abatement using tradeable permits, rather than the "command and control" alternative. However, it is important to note that this result depends on the assumption that regulated firms will use the opportunity for permit trades to exploit all available cost-savings by reallocation of abatement, so as to secure the optimal, cost-minimising abatement pattern. If, in practice, polluters respond to economic instruments in ways which depart from the optimising model, perhaps because of the presence of transactions costs or possible malfunctioning of the permit market, then the gains from trading that are in principle available will not be fully realised, and the *ex ante* analysis will have overstated the benefits from the use of tradeable permits.

In practice there are other aspects of the use of tradeable permits (or other market mechanisms) in environmental policy where we know about as much in advance of experience as we would be likely to learn from subsequent post-experience research. To a certain extent this is the case with the environmental benefit achieved by using tradeable permits. In principle, "cap-and-trade" tradeable permit schemes deliver certainty in their quantitative impact on emissions levels, and we do

1. Ex ante analyses of this sort are more complicated where the problem being analysed is one where there are spatial or temporal differences in the damage from polluting emissions. In these cases, calculation of the efficient pattern of abatement needs to take into account both the marginal cost of abatement and marginal damage from abatement at each location or date. As with the more straightforward problem, uniform abatement from each source is unlikely to be optimal, but a tradeable permits scheme will only achieve the efficient outcome if it is designed so that permits trade at a ratio that reflects the difference in damage at different sources or dates.

not learn anything more about this through observation of such schemes in practice (aside from possible issues of illegal non-compliance, or in the cases where tradeable permit mechanisms incorporate some deliberate provision for partial retirement of emissions to secure an "environmental gain" from trades). Also, the methods that might be used to assess the value of the environmental benefit (using contingent valuation or other techniques) do not require observation of the instrument in practice. Thus, while an *ex ante* assessment of whether to tighten regulation of a particular type of emissions using a tradeable permits scheme may well include the environmental consequences as a key element in a cost-benefit calculation of the effects of the policy, we learn nothing additional through experience to add to this.

2.2 What might we learn from ex post evaluation?

On the other hand, there are a number of key issues concerning the performance of tradeable permits which we cannot assess adequately in advance, and which can only be assessed through the observation of practice. It is towards these issues that attention and research resources would be most productively directed in *ex post* evaluation research.

For example, on the basis of theoretical analysis, and *ex ante* quantitative modelling, we can show that the potential for savings in abatement costs may exist, because polluters differ in terms of their marginal costs of abatement. In addition, we can say, *ex ante*, that transaction costs may inhibit trade, and may therefore reduce the extent to which potential savings are realised in practice. However, it is unlikely that we can quantify the size of transactions costs in advance of experience, or assess their significance for different groups of market participants.

This suggests that evaluation research should pay attention to possible divergences between potential and realised efficiency gains, and to possible factors that may act as obstacles to realising some potential efficiency gains. An important area for *ex post* investigation will be the nature of transaction costs that are experienced by participants in systems of emissions trading or tradeable pollution permits. These are the key to "value added" in *ex post* evaluation studies, as compared to what can be known from *ex ante* quantitative studies of potential efficiency gains.

Likewise, while *ex ante* analysis may be able to forecast the consequences of a well-functioning market for pollution permits, these will depend on how well the market for pollution permits (or emission credits, etc) functions in practice. How well is information disseminated among market participants through the market process? Does the pattern of permit prices suggest efficient dissemination of information among market participants and efficient trading to eliminate potential differences in abatement cost between sources? Relevant indicators of market functioning might include the volume of trades the number of buyers and sellers, the extent of price dispersion between trades, etc. In addition to information issues, there are also potential permit market failures due to monopoly power. *Ex post* evidence on permit markets in practice can shed light on the circumstances in which market participants are able to exercise monopoly power in the permit market.

3. Criteria and priorities in evaluation

This section draws together a number of ideas prompted by the foregoing discussion of evaluation studies of some tradeable permit schemes in US experience, to develop a set of principles that might help to guide the design of tradeable permit evaluation studies in other contexts. The discussion is under three main headings. First, the US examples discussed have shown that there can be considerable diversity in the design of tradeable permit mechanisms. Some of the differences in the design of schemes had significant implications for their functioning. The first step in developing evaluation studies that can help in the process of policy learning across countries is clear

characterisation of the relevant features of the instrument. Second, choices need to be made about where evaluation resources can be most profitably directed. What can be learned from *ex post* evaluation that will contribute most to our understanding of the effectiveness of tradeable permits as environmental policy instruments? Third, there are a set of fundamental issues that are raised in evaluation that any study needs to address in a systematic and coherent way. Some of these are methodological (for example, determining the baseline against which effectiveness will be judged), while others concern process - when and where should evaluations be conducted, if they are to make an effective contribution to informed policy-making?

3.1 Characterisation

EPA (2001) sets out a framework for defining and classifying tradeable permit systems, so as to bring out the significant similarities and differences between the various schemes that have been implemented. It is suggested that eight basic parameters characterise the key features of trading systems:

> ***(i) Scope.*** *What is the scope of permitted trading activity? At one extreme, "trading" might only be allowed within a single production location. In this very restricted sense of trading, firms could benefit from flexibility in how they meet an aggregate emissions limit for a particular site. Broader scope would allow flexibility in compliance across production facilities owned by the same firm - so that, for example, an emissions increase in one plant could be offset by greater abatement in one of the firm's other plants elsewhere. External trading, between entities under separate ownership, widens the scope of trading still further. The scope of such external trading could be wholly unrestricted, or might, for example, involve geographical or other limits.*
>
> ***(ii) Cap.*** *Does the scheme place an explicit limit on aggregate emissions or effluents? "Cap-and-trade" systems such as the US acid rain programme involve such a limit, but credit systems do not do so explicitly, and may not do so implicitly either. In particular there are two reasons why credit systems may not fix aggregate emissions. One is that they may operate on variables such as the emissions rate (e.g. per tonne of output) from individual sources, so that aggregate emissions are not wholly fixed, but depend on the level of activity. Another is that the credit generated by a particular action may not exactly correspond to the achieved reduction in emissions - for example, where the rules require some "environmental gain" from trading.*
>
> ***(iii) Commodity being traded.*** *How, precisely, is the traded commodity defined? Possibilities noted by EPA (2001) include allowances for future pollution (as with the VOC "offset" trading discussed in 3.4 above), credits for quantifiable reductions in pollution (as with VOC "open market" trading), emission characteristics of products (as with CAFE standards), or rights to own and operate products themselves (as with the tradeable permits in some Colorado mountain communities covering installation and operation of wood-burning stoves).*
>
> ***(iv) Distribution of tradeable permits.*** *The main choice here is between auctioned permits, and administrative allocation. Auctions can vary in design, but they have in common that permits are paid for, at a price that reflects purchasers' bids. The revenue raised from the auction may be retained, or returned to participants as a group according to some formula. Rules for administrative allocation may assign permits on the basis of historical emissions ("grandfathering"), current or past output levels, etc. They may or may not involve some form of charge.*
>
> ***(v) Trading ratio.*** *Do all trades take place on a 1:1 basis, or is there variation in the trading ratio? Schemes may require a ratio different from 1:1 either uniformly (i.e. all trades involve some "retirement" of emissions, so that the value of a permit, in terms of allowed emissions, deteriorates*

uniformly through trade) or on a basis that varies depending, for example, on the location of the buyer and seller, the season of the year, or other factors.

(vi) Banking. *Can permits in excess of current requirements be stored for future use, and on what terms does this "banking" take place? Is there any symmetrical scope for borrowing permits against future allocations?*

(vii) Monitoring. *How are firms' holdings and trades of permits recorded and monitored, and what arrangements are used to ensure the emission compliance of both the supplier and purchaser of permits? For example with the US acid rain trading programme, there is no centralised monitoring of transactions, and firms merely need to be able to produce the required quantity of valid permits at the year-end. With the VOC "open market" schemes discussed in section 3.4, by contrast, the purchasing firm becomes liable for any failure by the seller to achieve the abatement performance that generated the credit.*

(viii) Environmental benefit. *Is the scheme designed to operate in such a way that trading activity can generate environmental benefits (e.g. through "retirement" of a proportion of traded emissions), or is the level of emissions fixed by the initial issue of permits, and unaffected by subsequent market behaviour?*

3.2 Criteria

The 1997 OECD report *Evaluating Economic Instruments for Environmental Policy* (OECD, 1997) set out a framework for *ex post* evaluation of market mechanisms, covering seven key categories of costs and benefits from their use:

(i) Environmental effectiveness. *How far, and in what way, does the mechanism achieve the required reduction in emissions or environmental damage which is the objective of policy?*

(ii) Economic efficiency. *A central element in the case for using economic instruments in preference to "command-and-control" regulation in environmental policy is static efficiency (or "cost-effectiveness") in achieving a given level of abatement - in other words, reducing aggregate abatement costs by switching abatement to firms which can reduce pollution at least cost.*

(iii) Administration and compliance costs. *These include administrative costs incurred by regulatory agencies for emissions measurement, monitoring and regulating permit transactions, and costs of enforcement activities, and compliance costs in the form of administrative and managerial costs incurred by the regulated firm. From the point of view of the economy as a whole, all such costs (whether borne by government or the private sector) are a dead weight cost of the system, absorbing potentially-productive resources. Consideration also needs to be given to enforcement effectiveness in judging the burden of administration and compliance costs.*

(iv) Revenues. *Market mechanisms such as auctioned tradeable permits generate government revenue. This revenue has the potential advantage that less revenue may need to be raised from other, less-attractive, fiscal instruments.*

(v) Wider economic effects. *This category includes macroeconomic effects (for example on the price level, and possibly the rate of inflation, employment and economic growth), effects on competitiveness and trade patterns, and income distribution effects.*

(vi) "Soft" effects. *These include possible effects of the use of an instrument on attitudes and awareness, for example by signalling the importance of reducing a particular type of emissions.*

(vii) Dynamic effects, and innovation. *Economic instruments are, in principle, likely to be more effective at stimulating innovation in pollution-abatement technologies than regulations which merely require a given level of compliance.*

All of these impacts are costs and benefits that an evaluation study of a tradeable permit system might wish to address, and clearly all should, in principle, be included in any overall assessment of the balance between costs and benefits from using tradeable permits, rather than alternative instruments. In some cases, as we have seen, evaluation research may choose to select one or a limited number of these effects for examination in detail. Thus, for example, one of the studies which will be discussed in section 5.2 focuses on one aspect of the impact of lead credit trading - namely, its impact on the pace of technological change in gasoline production (Kerr and Newell, 2001). Also there may be policy reasons to place greater emphasis on some of these headings than others. Effects on international competitiveness may for example be a particular focus for public controversy, that objective evaluation evidence might help to resolve.

In addition, even where the objective of evaluation research is to expand knowledge about the full range of relevant costs and benefits from the use of tradeable permits, there may be reasons, as Section 2 has suggested, to focus effort on certain aspects within the full range of effects. The amount of genuinely new light that can be shed on the consequences of employing tradeable permits, through study of their use in practical situations, seems likely to vary widely across the seven categories noted above. Under some of the headings, evaluation research can genuinely add new information, over and above what is already known from *ex ante* theoretical consideration and modelling. Less may be learned from practical experience about other effects.

Drawing on the analysis in earlier sections of this paper, it is suggested that the following aspects of the operation of tradeable permit systems would be particular priorities for study in *ex post* evaluation:

(i) The efficiency of the market for pollution permits. How well is information disseminated among market participants through the market process? Does the pattern of permit prices suggest efficient dissemination of information among market participants and efficient trading to eliminate potential differences in abatement cost between sources? Relevant indicators of market functioning might include the volume of trades, the number of buyers and sellers, the extent of price dispersion between trades, etc. In addition to information issues, there are also potential permit market failures due to monopoly power. *Ex post* evidence on permit markets in practice can shed light on the way in which different patterns of abatement costs, permit allocation rules and market structures can determine how much monopoly power affects the permit market.

(ii) How far polluters respond to potential abatement cost savings. *Ex ante* we can show that the potential for savings in abatement costs may exist, but transaction costs may inhibit trade, and may therefore reduce the extent to which potential savings are realised in practice.

(iii) The scale and nature of transactions costs. *Ex post* evaluation research should pay particular attention to possible divergences between potential and realised efficiency gains, and to possible factors that may act as obstacles to realising some potential efficiency gains. A central issue in this is the analysis of the various transaction costs that are experienced by participants in systems of emissions trading or tradeable pollution permits. These are the

key to "value added" in *ex post* evaluation studies, as compared to what can be known from *ex ante* quantitative studies of potential efficiency gains

(iv) The quantification of costs of administration to both the regulated firms and the regulatory agency. A potentially significant element in this is the cost of enforcement activities, and their relationship with enforcement effectiveness. Little can be known about these costs, and the risks and extent of illegal non-compliance, except through the observation of actual experience

(v) The practical significance of "soft" effects. *Ex post* evaluation of tradeable permit schemes in operation is really the only way in which we can assess the importance of "human" factors such as attitudes, commitment and "ownership" in the successful implementation of emissions trading.

(vi) The environmental consequences of polluter responses Cap-and-trade permit trading systems in principle fix the level of aggregate emissions, and, aside from possible illegal non-compliance, this outcome is known in advance of experience. If total emissions are fixed in this way, analysis of environmental effects may not be a high priority for detailed study in *ex post* evaluations of tradeable permits. However, even with a cap on aggregate emissions, there may still be relevant issues about possible regional redistribution of pollution damage, or the development of "hot spots". And with some tradeable permit schemes, in which a proportion of permits is retired with each trade in order to generate an environmental dividend, the level of realised trading will have a direct bearing on the environmental outcome achieved.

4. Methods for *ex post* evaluation

The evaluation baseline – i.e. "what would have happened otherwise?"

Not all of the change in polluter behaviour following introduction of a system of tradeable permits can be attributed to the effects of that economic instrument. Some of the changes might have occurred in any case, regardless whether policy had changed or not. In evaluating the effects of an instrument we are not interested simply in describing changes, but in ascribing cause. To do this, we need to compare the situation following introduction of the tradeable permits with an alternative, hypothetical, scenario (the evaluation baseline, or "counterfactual"), showing what would otherwise have happened, without the tradeable permits.

There are two aspects to this. One is that we need to be clear about how we specify environmental policy in the hypothetical alternative. What, if anything, is the alternative environmental policy against which the tradeable permit scheme is being evaluated? Should the evaluation of the performance of tradeable permits be made against the likely economic effects of other policy instruments (e.g. command-and-control regulation) with equivalent environmental effects, or should it compare the instrument with a no-policy baseline (and, consequently, poorer environmental performance)?

There are important methodological implications. One attractive feature of conducting evaluations on the basis of a comparison of alternative policies with equivalent environmental impact is that it allows the evaluation to side-step the difficult question of the measurement of the value of environmental benefits. A full cost-benefit evaluation of a single policy against a no-policy outcome would, by contrast, have to place a value on the environmental effects, and this may not be straightforward. A difficulty of "equivalent policy" evaluations, however, is that there may be as much

uncertainty about the environmental and other effects of alternative policies - even conventional "command and control" regulation - as there are about the effects of tradeable permits.

If comparisons are to be made against a hypothetical counterfactual or "baseline", the evaluation needs to predict or model what would have been the evolution of all the relevant variables in the alternative baseline scenario. This cannot of course be observed, but various different methods are available to construct a baseline scenario (sometimes referred to as the "counterfactual"). As is discussed in more detail in section 5.3 below, the Ellerman *et al* (2000) study of acid rain trading used a number of approaches, ranging from a relatively straightforward and mechanical baseline, to sophisticated econometric modelling. The range of different approaches to modelling the counterfactual is used as a check on the robustness of the conclusions, because it is recognised that a significant margin of error surrounds any attempt to model a hypothetical alternative scenario.

"Disentangling" policy packages

A frequent difficulty in assessing the effects of an economic instrument in environmental policy is that economic instruments are in practice rarely used in isolation, but are combined in a "package" of policy measures. Often, the effects of new economic instruments are reinforced by regulatory measures, or other measures, taken at the same time.

In many cases, it will simply be impossible to separate the individual contribution of policy measures implemented as part of a package, and the evaluation will have to be content with evidence on the joint effect of all the elements of the package taken together. Sometimes there may be some scope for international comparisons to indicate the extent to which particular components of policy packages are responsible for outcomes, if similar tradeable permit schemes are implemented in different countries along with different accompanying measures, or the same measures in different proportions, allowing the separate contribution of each component to be separated. However, such opportunities will be rare, and the evidence derived from such cross-country comparisons will depend critically on the assumption that the countries are otherwise equivalent in terms of the responses expected from the implementation of particular policies.

Timing

How soon after the introduction of a system of tradeable permits should its results be assessed? Is there any point in setting up arrangements for early evaluation, with the aim of generating evaluation findings that can be used to inform modifications and improvements to the scheme, or that may be of value in considering how tradeable permit schemes in other areas might be most effectively designed?

With many environmental market mechanisms, such as emission charges and other environmental tax incentives, there is probably little to be learned about the effectiveness of the scheme from early evaluation. Behavioural responses to tax-induced changes in prices are likely to occur slowly, especially where the long run response would involve changes in capital stock. Thus, for example, environmental taxes on energy may have an impact on industrial energy use spread out over 15 years or more. The response observed in the first year or two of a tax increase may be so small that it cannot easily be distinguished from normal random fluctuations. With tradeable permits, by contrast, there is rather more that can be learned from early monitoring. The existence, pattern and prices of permit trades can provide useful early indications of the efficiency of the permit market, and the existence of efficiency gains, even though it may take considerable time for the full long-run effects of the pricing incentive to be felt. In effect, the permit market generates data on some of the processes of interest, that cannot so easily be collected for other incentive-based environmental policy instruments.

The timing of the evaluation may also have implications for the data available for the evaluation. Evaluation too long after the introduction of a new instrument is likely to mean that some of the relevant economic actors are no longer available to survey or interview (firms go out of business, managers retire or move jobs, and so on). Also, with the passage of time the "counterfactual" may become increasingly imprecise, and the effects on levels of pollution and production costs of the introduction of a system of tradeable permits may be more difficult to distinguish from the effects of other factors also affecting these variables.

Another consideration in the timing (or, more specifically, concerning the time-frame over which the policy is analysed) is the possibility that the behavioural response to an economic instrument may partly anticipate its introduction. (An example of this will be seen in Section 5.3, where it is found that some effects of the US acid rain trading scheme arose before it came into force.) Polluters may respond to the advance announcement of measures, and may make long-term investment decisions, or decisions about the retirement or replacement of existing capital stock in the light of future policy requirements (or even based on their assessment of possible future policy developments). Deciding an appropriate baseline for evaluating effects may be difficult. If anticipation effects are likely to arise, confining the analysis to the period following the date of introduction of the measure will be too late to include all of the relevant effects.

Institutional location

The institutional context within which evaluation studies are conducted can also have a significant influence on the success of the evaluation, and on its effectiveness in stimulating reforms and refinements to the policy. There are a range of possible models, stretching from internal evaluation conducted by the agency responsible for designing or implementing the tradeable permit mechanism, to wholly-independent evaluation by "outsiders" such as independent academics, working without any involvement or support from the policy makers responsible for the scheme. In between, there are a variety of possible intermediate arrangements, including external academic researchers, research institutes, and private-sector consultancy firms working with some degree of support from the relevant agency. This support might be in terms of information (such as data, access to agency staff running the scheme, or to regulated firms), or financial, and it could be provided either as "untied" funding or other assistance for a wholly-independent evaluation activity, or through some form of consultancy contract, with more active agency involvement in guiding and monitoring the progress of the research.

These are important differences, and evaluations conducted in different ways have different characteristics, and relative strengths and weaknesses.

First, comprehensive evaluation studies can rarely be conducted totally independently from the relevant institutional actors involved in the policy. Studies that have the explicit support of the environmental agency running the permit scheme may have better access to agency staff with detailed knowledge of the system, and may be better able to obtain confidential data. The relationship with the agency may also influence the willingness of the subjects of the regulation to co-operate with the evaluation - though in which direction is not always clear. Second, the relationship between the evaluator and the institutions responsible for the policy may affect the actual or perceived objectivity of the work. Third, translating evaluation findings into future policy requires that policy-makers accept the findings of the evaluation. Wholly-external evaluations, conducted without involvement of policy-makers at each stage, may have less impact on subsequent policy developments than those which have stronger links with policy-makers (including perhaps funding). In the latter case, the relevant policy-makers may be more willing to take "ownership" of unpalatable findings, and to accept the need for policy reforms.

As the earlier OECD report Evaluating Economic Instruments for Environmental Protection (OECD, 1997) observed, these considerations do not point to a single clear conclusion about the best institutional location for evaluation research. There is a trade-off between information and objectivity which may look different, depending on the policy instrument concerned, and the research, administrative and political traditions of individual countries. Suggestions which may help to ensure that studies conducted towards the "internal" end of the spectrum maintain (and are seen to maintain) objectivity include (i) establishing a steering group to oversee the evaluation, including outside members, (ii) making data and research methods open to peer review and scrutiny, and (iii) making a prior commitment to publication, regardless of the research findings.

5. Evaluations in practice

This section looks at the approaches adopted and results obtained in a number of evaluation studies for tradeable permit applications. All of the studies relate to US experience, reflecting the much more extensive use made of tradeable permits in the US than elsewhere, but they differ both in the types of tradeable permit applications studied, and in methodology. The studies relate to four different tradeable permit systems:

- State-level emissions trading in the Los Angeles basin in California, which has been the subject of a number of separate evaluation-type studies, including the early academic evaluation of by Foster and Hahn (1995). The evaluation discussed here, however, relates to the later RECLAIM system, established in 1994.

- Arrangements for trading to support the US phase-out of lead in gasoline, various aspects of which have been the subject of different research studies. One interesting feature of this group of studies is that they comprise retrospective evaluations of a studies of a completed episode of trading.

- A major recent study of the US-wide trading scheme for acid rain

- An evaluation report on arrangements in a number of US states for emissions trading for VOCs.

5.1 Evaluation audit of RECLAIM

RECLAIM, the Regional Clean Air Incentives Market, was established by the South Coast Air Quality Management District (SCAQMD) in 1994, as a trading system to facilitate more flexible, lower-cost, adaptation to air quality requirements (National Ambient Air Quality Standards) in the area around Los Angeles covered by SCAQMD. It has replaced source-by-source controls on emissions of nitrogen oxides (NOx) and sulphur oxides (SOx) with an emissions trading system for these pollutants, which limits the aggregate emissions within the area, and allows trading to determine the distribution of emissions across individual sources.

During the 1980s the Los Angeles area had been one of the parts of the US in which there had been significant trading activity under the EPA's emissions trading program, a nation-wide scheme, allowing certain specified types of flexibility and trading for five principal air pollutants - hydrocarbons, carbon monoxide, SOx, NOx and particulate matter. The experience of emissions trading in Los Angeles was evaluated by Foster and Hahn (1994), who noted the major effect of geographical restrictions on trade on the functioning of the market, and the high level of transactions costs associated with trades. RECLAIM covers a smaller number of pollutants (SOx and NOx) but is designed as a less-constrained market, with a clearly-defined group of tradeable commodities, and explicitly-defined state-level aggregate limits on emissions. Although the experience of active

participation in emissions trading in the Los Angeles area may have provided experience and familiarity with emissions trading, that may have encouraged both industry and the policy community to be receptive to the proposals for RECLAIM, it seems that a more significant influence on its development was the parallel creation of the national trading regime for acid rain (Title IV of the 1990 Clean Air Act Amendments, discussed in 3.3 below). RECLAIM has many similarities in terms of design with the national acid rain trading scheme, despite its focus on a specific regional area, and state-level policy relating to that area.

The performance of RECLAIM has been discussed in a number of academic studies, including Fromm and Hansjürgens (1996), and Thompson (1999). The evaluation study described here, however, is that conducted by the agency running the programme itself (SCAQMD, 1998). The analysis of the evaluation is referred to quite extensively in a number of other subsequent studies, including for example, a recently-published report on RECLAIM for the National Academy of Public Administration (Kerr, 2000), part of a wider NAPA report on trading systems.

Design of the instrument

RECLAIM applies to all major sources of SOx and NOx emissions (broadly, those that emitted four tons or more of these pollutants in 1990). Other facilities have the option of voluntary participation in the programme, although few have so far done so. In 1998, there were 329 plants within the RECLAIM system, covering about 65% of total stationary-source NOx emissions and 85% of stationary-source SOx emissions in the area. Participating sources receive an allocation of RECLAIM trading credits (RTCs). They are required to hold RTCs sufficient to cover their actual emissions, and are freely able to supplement or dispose of their initial allocation through trading.

Allocations are based on sources' past level of activity (calculated as peak activity level over 1989 to 1992) and the emission control requirements for year 2000 and 2003 specified by the 1991 Air Quality Management Plan. They are intended, therefore, to correspond to the emissions levels that sources would have been achieving if subject to the command-and-control regulations that RECLAIM replaced. Allocations were made to each RECLAIM facility at the start of the programme, for each year from 1994 through to 2010. The amounts allocated for each year showed a sharply-declining time profile, to reflect the increasingly-stringent emissions control required by the Air Quality Management Plan. The total NOx allocation began at 103 tons per day in 1994, declining to 35 in 2000 and 26 (one quarter of 1994 allocations) in 2003. The total SOx allocation began at 25 tons per day in 1994, declining to 14 in 2000 and 10 (40% of 1994 allocations) in 2003. Allocations are valid in the specified one-year time period only, and there is no possibility for inter-temporal flexibility in the use of allowances through banking or borrowing. As a result of this restriction on inter-temporal trading and the sharply-declining allocations, the trading system operates against the background of an increasing scarcity of emission credits.

The audit

The agency running the RECLAIM program, SCAQMD, is required under the rules of the programme to make regular reports on various aspects of the programme. Annual programme audits are required on various specific topics, as well as a more comprehensive audit after three years of operation. SCAQMD (1998) constitutes the comprehensive 3-year review, covering the first three years of operation of the programme. RECLAIM was designed not only to meet state and federal clean air requirements, but also various other performance criteria (equivalent enforcement effectiveness, lower implementation costs, lower impact on employment, and no adverse health effects). The 3-year audit thus addresses a wide range of separate topics relevant to the various performance criteria and objectives, under ten headings (each corresponding to a chapter of the audit report):

- changes in the number of sources subject to RECLAIM, including changes due to closures
- The allocation of RTCs, in total, and between sources
- Emission reductions achieved by RECLAIM sources
- Trading activity and prices of traded RTCs
- Compliance monitoring and costs
- New Source Review Activity
- Socio-economic analysis
- Air Quality and Public Health effects
- Amendments to the RECLAIM programme
- Changes to the programme which are under consideration

The audit was written by SCAQMD staff, under the guidance of a RECLAIM Advisory Committee, established for the purpose in 1997. The advisory committee had 19 members, drawn from firms regulated by RECLAIM, environmental organisations, academic and policy research institutions, regulatory agencies, and others with relevant expertise. The committee met with SCAQMD staff on six occasions, providing input to the audit, and commenting on a preliminary version of the audit report. In addition, three subcommittees worked on specific aspects of the audit. The process culminated in presentation of the report to the SCAQMD Governing Board in May 1998, at a public hearing.

An interesting feature of the audit report is that it is explicitly recognised that not all members of the advisory group agreed with all aspects of the audit. A number of chapters conclude with a section in which dissenting members of the advisory committee make critical comments on the analysis or conclusions of the chapter, followed by a response by SCAQMD staff. This format appears to have been designed to allow the audit report to make comments of substance on issues where there was significant disagreement between members of the advisory group, rather than to negotiate a bland, lowest-common-denominator, consensus. Although it has the drawback that the audit does not completely resolve a number of controversial issues, it has significant advantages. Firstly, it makes it possible to assemble a broad and inclusive panel, comprising members from a wide range of perspectives, which helps to increase acceptance of the audit findings overall. Secondly, it reduces the temptation to make claims that are not well-justified by the data. Thirdly, it helps to identify the areas of genuine uncertainty and controversy, and thus focus subsequent discussion.

Two key issues of general interest covered by the audit are the effects of RECLAIM on emissions levels and the level and pattern of trading activity.

Impact on emissions levels. The system is a cap-and-trade mechanism, with trading throughout on a 1:1 (i.e., ton-for-ton) basis, and no provision for inter--temporal trade. As a result, overall emissions are constrained to a maximum level in each year given by the initial allocation (except to the extent that there may be any illegal emission or fraud). The only remaining issue is whether emissions are constrained by the aggregate issue of allowances, or whether they fall below this level. The report finds that for the first three compliance years emissions levels were below aggregate allocations. Some 24% of total NOx RTCs, and 27% of SOx allowances, were unused in 1996. Although emissions were on a declining trend, the allowance allocation was declining more rapidly, and the audit anticipated that the "crossover" years (in which RTC demands would exceed supply) were about to be reached: 1999 for NOx and 2001 for SOx. The extent to which the scheme had reduced emissions was one of the subjects about which representatives of environmental groups on the Advisory Committee expressed

dissent. Here, as in some other places, the existence of the advisory committee has subjected the audit's claims to detailed analytical cross-examination.

Trading activity and prices. Trading volumes have been substantial, with SCAQMD (1998) reporting that by the end of 1997 some 1,200 trades had taken place, covering about 0.25 million tons of NOx and SOx RTCs. A considerable proportion of RTC trading activity has been "zero-price" trading, such as transactions between two plants under common ownership - only some 15% to 20% of trades have been "priced" trades. These priced trades include trades directly between two firms, and trades mediated by brokers. Prices are reported to SCAQMD, allowing it to monitor the compliance costs of RECLAIM, and the relationship between prices and the "backstop" price ($15,000 per ton), which triggers a review of the RECLAIM programme. The prices of traded RTCs have been well below the prices projected at the time the system was established. The initial staff assessment of likely RECLAIM prices, made in 1993, was that NOx allowance prices would lie at around $9000 per ton in 1996-98, but actual prices were no higher than $600 per ton. Both SOx and NOx RTCs valid for years after 2000 trade at much higher prices. In 1997, for example, NOx prices for 2000 to 2010 were around $1800 per ton.

An *ex ante* assessment of the cost savings from RECLAIM, quoted by Stavins (2000), estimated that these would amount to some $58 million annually, a saving of some 42% compared with command-and-control compliance costs. This is not a matter which was explicitly addressed in the 3-year audit, although the implication of the lower-than forecast prices for RTCs would be that the gains from flexibility on the marginal allowance trade are substantially lower than initially projected. Two possible reasons that could account for this outcome have very different implications for the aggregate gains from RECLAIM. (i) If RTC prices are low simply because too many allowances have been issued (as suggested in the dissenting comments from representatives of environmental organisations on the Advisory panel), then RECLAIM has provided little incentive for abatement. (ii) On the other hand, the implications of low prices may be different if, as the SCAQMD staff response on this issue argues, there have been significant innovations in abatement technologies which were not anticipated at the time of the *ex ante* assessments. This could be consistent either with substantial efficiency gains from RECLAIM (if responsibility for the innovations can be attributed to RECLAIM), or with lower *ex post* gains (if the innovations would have taken place without the RECLAIM programme). Resolving this issue may be of little importance in an audit designed to assess whether the functioning of the RECLAIM system could be improved, but would naturally be of considerable interest to policy-makers elsewhere seeking to learn lessons from RECLAIM.

In practice, the real test for RECLAIM came when the so-called "crossover" year was reached, and the allowance allocation began to bite on current emissions levels. In year 2000 the price for current-year NOx RTCs shot up to some $45,000 per ton, way beyond the previous year's price for current RTCs of $1,827 per ton, and the price of $4284 per ton at which year 2000 RTCs traded during 1999, and well above both the previously forecast price and the $15,000 per ton backstop price that triggers a review of the programme (SCAQMD Board Agenda, January 19, 2001). The SCAQMD has given urgent consideration to possible measures to stabilise RTC prices. It is argued that the problem is essentially short-term, and that abatement actions are already in train which will, in the long run, act to reduce RTC prices. Unfortunately, given the impossibility of inter-temporal substitution within the RECLAIM mechanism, there is no automatic way for this to lead to stabilisation of the price for current RTCs.

5.2 Tradeable Permits and adjustment - the case of lead trading

There have been a number of research studies evaluating the operation of the system of inter-refinery trading which was used to support the US phase-out of lead in gasoline. The relevant stage of the gasoline lead phase-out began in November 1982, when the US EPA introduced a limit on

the average level of lead in a refinery's gasoline production of 1.1 grams per gallon, subsequently lowered to 0.5 in July 1985, and 0.1 in January 1986. Between November 1982 and the end of 1985 a system of "inter-refinery averaging" allowed refineries to reallocate lead within the overall lead content limit, and credits could be banked through until the end of 1987. The system was designed to provide flexibility to refiners during this rather drastic phase-out, as refiners adjusted to the new lower lead limits which were to apply uniformly from the end of 1987. It is therefore now a completed episode; once the phase-out of lead in gasoline was completed the need for the trading regime disappeared.

All of the studies considered here are retrospective studies, written in the light of experience, and discussing the scheme over its whole lifetime. An extensive discussion of *ex ante* studies undertaken prior to the scheme can be found in Nichols (1997). EPA estimates from 1985, reported by Hahn and Stavins (1991, page 17), were that the savings from the lead trading programme would be approximately 20 per cent over alternatives without lead trading.

The studies include a number of general analyses of the functioning of the lead credits market. Anderson, Hofmann and Rusin (1990), in an American Petroleum Institute research paper, provide an extensive discussion of trading activity in the lead credits market, as part of a broader survey of the use of incentive mechanisms in environmental management. Hahn (1989), Stavins (2000), and EPA (2001) also include (in some cases, brief) discussions of the performance of the lead credits scheme in the context of broader discussions of the performance of market-based environmental policy mechanisms. Nichols (1997) is primarily a discussion of the economic analyses undertaken in support of the development of the gasoline lead phase-down policy, and the crucial role that they played in the policy formulation process, although it also contains some discussion of market outcomes.

There are also two academic studies which focus specifically at single aspects of the performance of the lead credit market. Kerr and Maré (1998) studies the effect of transactions costs on the cost effectiveness of the lead credits market. As already noted, transactions costs are one of the reasons that actual performance of tradeable permits systems may differ from *ex ante* assessments. Kerr and Newell (2001) looks at how the effects of the tradeable lead credit regime on incentives for efficient technology adoption by the regulated refineries.

Design of the instrument

Lead credit trading (or "inter-refinery averaging") allowed refiners and importers to trade lead reduction credits in order to meet limits for the lead content of gasoline, under which the maximum average lead content in a refinery's gasoline production should fall sharply, from a limit of 1.1 grams per gallon introduced in November 182 to no more than 0.1 grams per gallon from January 1986. It was a trading scheme directed at providing flexibility during a (relatively short and sharp) adjustment period. Without trading in lead credits it was thought that either a longer period would have to be allowed for the phase down, or else there would be a risk of short-term disruptions to the supply of gasoline (Carlin, 1992, page 5-8).

The initial allocation of lead rights to firms was determined by the amount of leaded gasoline produced by the firm and the current EPA standard required (Hahn and Hester, 1989, p. 381).

Trading could be internal or external - a firm could use lead rights itself by adding more lead to its gasoline at some point during the quarter than would otherwise be allowed, or it could sell its rights to another firm. (Hahn and Hester, 1989, page 382). Initially, rights simply expired if they were not used or sold during the quarter in which they were created. However, starting in 1985, refiners

were allowed 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 program at the end of 1986.

Transactions were reported to the EPA and each refiner was required to have a net balance of lead rights greater than or equal to zero for the quarter. Enforcement was largely operated through paper audit of the reported transaction statements, to identify discrepancies and inconsistencies between transactions reported by the buyer and the seller.

Ex post evidence on effects

Several hundred refiners, constituting over half the total number, participated in the reduction credit market. The market was very active and trading activity generally increased throughout the life of the program. In particular there was extensive use of banking.

The policy involved a number of rule changes - the introduction of lower lead-content limits on two dates, and the introduction of arrangements for banking. Refiners tended to see trading and banking as a way of smoothing their adjustment over the date of the rule change. When a lower limit on lead came into effect it was exceeded for a while by refiners of all sizes who withdrew rights banked previously.

The prices at which trades were made were not required to be reported to EPA, and systematic data on prices could not, therefore, be obtained from official sources, and had to be collected from market participants. Anderson, Hofmann and Rusin (1989, page 26) report, on the basis of anecdotal evidence, that lead rights may have traded at a range of prices between 0.75 and 4 cents per gram. Other estimates are that prior to the institution of banking, prices were consistently below one cent per gram after which they fluctuated between two and five cents.

On the other hand, systematic official data was available on quantities traded and banked, as a result of the reporting requirements of the scheme in operation. The large volume of rights traded and banked imply that savings were been substantial. The actual amount banked turned out to be some 10 billion grams, very close to the EPA's *ex ante* estimate, for an average saving of 2.5 cents per gram banked (Carlin, 1992, page 5-9).

Ex ante it had been assumed that it would principally be smaller refineries that would need to take advantage of the flexibility that lead credit trading offered, and during the first six quarters of the trading program small refiners consistently exceeded the EPA lead limit by purchasing credits. Indeed, throughout the period to 1987 small refiners continued to add more lead to gasoline than did large refiners. Nevertheless, roughly equal proportions of small traders bought and sold rights indicating that they were not uniformly hard-pressed to meet the new lead standards. However, the proportion of large refiners engaged in trading was generally greater than that of small refiners, and it is possible that for some smaller firms the information costs or other transactions costs involved in trading were prohibitive. (Hahn and Hester, 1989, page 384).

The environmental effects of the trading program cannot be calculated exactly, and depend crucially, of course, on what lead content standards are assumed would otherwise have applied. If it is assumed that the same time profile of standards would have operated, without any provision for trading, then the effect of trading on environmental outcomes may have been negligible. However, if it is assumed that trading permitted a more aggressive time profile of reductions than would otherwise have been feasible, then lead trading may have accelerated the reduction in lead emissions from motor vehicles.

The general consensus of the studies reporting and evaluating the experience of the lead trading programme is that the scheme worked well - and certainly much better than most other

practical applications of tradeable permit mechanisms to that date. Among the features of the situation which, it is suggested, account for the good performance of lead trading are that the time-scale of the exercise was clearly set out in advance (although, in practice, subsequently adjusted through the introduction of banking provisions), rights and obligations were clearly defined, and 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.

Kerr and Maré (1998) is a study of the role of transactions costs in the market for lead permits. Such costs are of particular interest in understanding possible divergences between the theoretical gains in terms of cost-effective compliance through tradeable permits and the cost savings actually achieved in practical applications. The particular focus of the Kerr and Maré study is one type of transaction cost, namely "first trade" transactions costs, in other words the cost of making one trade rather than not trading at all. Using data collected from 30 major oil companies, and showing the trading partners and quantities traded for all permit trades carried out by each of 87 refineries over a two year period, they estimate an econometric model of the probability of individual trades, to assess how far observed trading behaviour can be explained by differences in the potential gains from trading, and how far it appears to be influenced by "first trade" transactions costs. They find significant effects of transactions costs on the overall gains from trade, sufficient to reduce the overall achieved gain from trade to some 10 to 20 per cent below the total potential gain. Some of this loss comes from the transaction costs themselves, and other losses are the potential trading gains that are foregone. For some oil companies, especially the smaller and less sophisticated operations, the losses from transactions costs were of considerably greater significance.

Kerr and Maré conclude that their results show that tradeable permit markets can still be a more cost-effective instrument of environmental regulation than non-market instruments, in reasonably promising contexts. Nevertheless, transaction costs do reduce the efficiency savings below the maximum level theoretically available, and they warn that this effect may be particularly severe in contexts where potential traders are unsophisticated and where they have few existing connections, so that search costs are high.

Kerr and Newell (2001) is another tightly-focussed academic study, providing *ex post* evidence on one specific aspect of the US lead credit trading scheme, namely the impact of the scheme on incentives for technology adoption. The theoretical literature suggests that economic instruments such as tradeable permits and pollution taxes should create more effective incentives for technology adoption than conventional command-and-control regulation, mandating the use of particular technologies or fixing uniform emissions levels. Kerr and Newell's study tests how far practical experience bears out the predictions of theory, in the context of refineries' decisions to adopt a key technology for reducing lead levels in gasoline (pentane-hexane isomerisation).

Kerr and Newell's research makes use of a database they have assembled, covering the annual technical and operating characteristics of 378 US refineries over the period 1971 to 1995. The period predates the first use of isomerisation in US gasoline refining, and the centre of the period includes the years during which the EPA required sharp reductions in gasoline lead content, including the 1982-87 life of lead credit trading. The data, compiled from a range of sources including published data and information provided by the oil companies themselves, include information on the adoption of isomerisation, the lead-trading behaviour of individual refineries, and other relevant economic and market variables.

Theoretical considerations would suggest that the speed of adoption of a new abatement-cost-reducing technology will tend to be increased, with increased stringency of the environmental

policy. However, this effect is unlikely to be uniform across firms. In particular, firms with relatively low marginal costs of pollution abatement (in other words, those firms which would tend to be sellers in a permit market) will have greater incentives for cost-saving technology adoption within a tradeable permit (or other incentive) regime. Other things being equal, one would expect to see these firms adopting new abatement technologies sooner than firms with high marginal abatement costs. Kerr and Newell find that these theoretical effects are borne out in the results of the econometric model they estimate. The increased stringency of the lead credits scheme over time, which raised the effective price of lead, encouraged increased adoption of lower-lead refining methods. Larger refineries, with more sophisticated technologies were more likely to adopt new refinery technologies. In addition, there was evidence that the tradeable permit regime led to more efficient adoption than with a less flexible regulatory regime: as theory would predict, a greater proportion of the adoption was by low abatement-cost refineries under the tradeable permit scheme than under regulation without trading.

5.3 "Markets for Clean Air" - the Ellerman et al evaluation study of the Acid Rain Program

The Ellerman *et al* evaluation study, published in the form of a book in 2000, contains the results of a five-year programme of research at the Center for Energy and Environmental Policy Research (CEEPR) at MIT. The research programme was initiated as a result of a request by the National Acid Precipitation Assessment Program (NAPAP) to draw up an evaluation for NAPAP's Quadrennial Report to the US Congress, and was funded initially by NAPAP and subsequently by the Acid Rain Division of the US EPA. A preliminary version of the study was published by CEEPR in 1997, and a number of individual components of the study have been reported in separate working papers and journal articles. Two of the authors of the report acted as contributors to the NAPAP report itself (NAPAP, 1998).

The programme

The US acid rain program aims to control emissions of SO2 from coal- and oil-fired electricity generation plants (which in total accounted for some 70 per cent of US SO2 emissions in 1985) through a "cap-and-trade" system of tradeable permits, set up under Title IV of the 1990 amendments to the Clean Air Act. Title IV places an aggregate cap on power plant emissions of SO2 of around 9 million tons, effective from year 2000 onwards. This emissions cap is equivalent to a reduction in annual emissions of some 10 million tons as compared with 1980. It was implemented in two phases:

- Phase I, covering the period 1995-1999, required aggregate emissions reductions of around 3.5 million tons per year from the 263 generating units with the greatest emissions (the so-called "Table A" units).

- Phase II, starting in 2000, extended the scope of the aggregate cap on SO2 emissions to all coal-fired and oil-fired electric power plants above 25MW capacity.

The basic structure of the tradeable permit regime is built around an annual distribution of tradeable emissions allowances (mainly on a grandfathered basis). Each of these allowances permits the holder to emit one ton of SO2. The permits may be used in the year of issue, or saved ("banked") and used in a subsequent year. They can also be traded freely across the entire US territory, on a 1:1 basis, either directly between emitting firms or through brokers. The EPA plays no role in the trades. The control mechanism requires each firm to produce to EPA (within a month of the year-end) sufficient allowances to cover its measured emissions during the preceding year. The penalties for

firms that fail to produce sufficient permits include both significant monetary fines and additional requirements for future abatement.

In addition to this basic structure there are a number of complications, some of them of practical significance for the functioning of the system. These include:

- "substitution" provisions within Phase I, which allow operators to substitute other generating units in place of the designated Table A units;
- incentives, taking the form of bonus allowances, for Phase I units to install flue gas desulphurisation "scrubbers";
- a small EPA auction of about 2.8% of the total issue of allowances, which was begun in 1993, and was intended to stimulate development of the market for permits.

Scope of the evaluation

The Ellerman et al (2000) evaluation is a major study, undertaken over a significant period of time, requiring substantial resource inputs, and able to draw on a significant body of prior and parallel academic research. It includes chapters covering the following specific topics, most of which are analysed using detailed quantitative data as well as conceptual argument and informed discussion.

- The political genesis of the programme, and the determination of allowance allocations
- The impact of the programme on the aggregate level of emissions
- The impact of the programme on the geographical pattern of emissions
- The functioning of the allowance market
- The aggregate cost of compliance with the acid rain program during Phase I
- Cost savings arising from allowance trades
- The impact of the provisions for "voluntary compliance" through substitution of Phase I units, etc.

Conclusions of the evaluation

The report concludes with a ringing endorsement of the programme, as a "star on the environmental policy stage". "The Acid Rain Program has thus far been a notable success. Title IV more than achieved the SO2 emissions-reduction goal established for Phase I, and it did so on time, without extensive litigation, and at costs lower than predicted. Moreover there has been 100% compliance by all affected sources. Not only have there been no violations of the law, but no administrative exemptions or exceptions have been granted to permit non-compliance. ... We are unaware of any other US environmental program that has achieved this much, and we find it impossible to believe that any feasible alternative command-and-control program could have done nearly as well." (Ellerman *et al* p314)

They draw a number of general lessons from the experience of the programme

> "The experience ... clearly establishes that large-scale tradeable permit programs can work more or less as textbooks describe. ... Owners of affected sources have taken advantage of the flexibility Title IV provided and focused on reducing emissions when and where it was cheapest to do so - even though some observers had contended that regulated electric utilities would never be agile enough to do this." (Ellerman *et al* p. 315)

Although the legislative process which led to the introduction of the programme was accompanied by extensive rent-seeking activity, especially surrounding the distribution of permit rights worth billions of dollars, this had little adverse effect on the operation of the programme (with the possible exception of the bonus allowances offered for FGD installation, which may have led to some over-investment in scrubber technology). They point out that regardless of the merits of the process by with the permits were allocated, the scope for trading ensured that a cost-minimising pattern of emissions reduction could result.

A reasonably-efficient competitive market for allowances developed quickly - by around the middle of 1994. They highlight two key design features of the system which underpinned this result. First, the scheme was defined in terms of "rights" to emit SO2, rather than "credits" for SO2 abatement. This allowed the system to be established without any need to define the baseline level of emissions for each participant, against which their credits for abatement would be calculated. The value of this is partly its simplicity, but also that it avoids any need for administrative approval of trades. Second, the scheme involved no geographical restrictions on trading. Every allowance was worth one ton of SO2, regardless of where it was used, resulting in a single, US-wide, allowance price, and a single, straightforward, market.

The trading regime proved robust to quite significant unanticipated developments, or "surprises". The most significant of these was the reductions in rates for freight transport arising from the deregulation of the railway system during the 1980s. This sharply reduced the cost at which low-sulphur coal from the Powder River Basin in the Midwest could be obtained by power stations in other parts of the US, and facilitated fuel switching to lower-sulphur fuel inputs as over a much wider than before. This development complicates the process of assessing the impact of Title IV, but in terms of the performance of the trading regime was handled reasonably smoothly. Allowance prices turned out to be substantially lower than originally anticipated, transmitting the information about the reduced value of marginal abatement reasonably quickly and efficiently to firms contemplating more costly abatement options, and hence minimising the amount of unnecessarily costly abatement activity. The authors observe that "it is hard to imagine any command-and-control regime adapting as sensibly to such an important exogenous event". In particular, they highlight the important role played by allowance banking in smoothing the adjustment to the new situation.

The report is critical of the so-called "voluntary" provisions of Title IV, especially the arrangements during Phase I which allowed utility companies to substitute other generating units in place of the designated Table A unit. They point out that a major driver of the decision to substitute appears to have been the basis on which allowances were to be allocated to the substitute units. The units chosen for substitution tended to be those which would benefit from overgenerous allowance allocations, rather than those with the most scope for reducing abatement costs, and units may have been selected for "voluntary" inclusion which were due to make emissions reductions anyway. The net effect of these substitution provisions was to weaken the incentives for abatement , and the authors argue that this experience suggests that such "opt-in" provisions should therefore be considered very carefully before being imitated in any subsequent legislation.

Methodological discussion

A central feature of the Ellerman et al (2000) assessment of the impact of Title IV on emissions and abatement is the careful and well-documented specification of a counterfactual, or baseline, against which the impact of Title IV can be calculated. It would be highly misleading to calculate the effects of Title IV simply by calculating changes against some historical baseline year, since there were a number of important factors which would have changed the level of emissions over time, even without the introduction of Title IV. For example, by 1995, SO2 emissions from Phase 1 units had fallen by 4.66 million tons compared with their emissions in 1990, the year when Title IV was enacted, a fall of more than 50%. But this does not mean that Title IV is responsible for a halving

of SO2 emissions from these units. On one hand, the level of electricity generation in these units rose significantly, by around 7% over the period, which would have led to a corresponding percentage increase in emissions, in the absence of any change in emission rates. On the other hand, there were well-documented downward trends in emission rates, pre-dating Title IV, which might have been expected to continue, even if no policy had been enacted. The shift towards lower-sulphur coal, partly prompted by changes in rail transport costs, would have continued to reduce emissions rates over 1990-95, even without Title IV.

Throughout the study an explicit "counterfactual" time profile and pattern of emissions is employed, as a baseline against which the impact of Title IV on emissions can be assessed. In much of the work, the study uses two different approaches to the estimation of counterfactual emissions, as a check on the sensitivity of the results to the precise method of calculating the counterfactual.

The first method employed is comparatively straightforward. Emissions from each unit, in the absence of Title IV (i.e. in the counterfactual), are calculated by multiplying the observed heat input in the year in question by the rate of SO2 emissions per unit of heat input observed in 1993 (i.e. immediately before the start of Phase 1).

Implicitly, this simple counterfactual is making three key assumptions:

- The sulphur trading mechanism had no effect on the heat input of individual units (i.e. the pattern of electricity generation across units was unaffected by Title IV).
- All of the change in the emission rate observed after 1993 is the effect of Title IV (i.e. emission rates would have been constant after 1993, were it not for Title IV).
- Title IV had had no effect on the emissions rates observed in 1993 (i.e. no anticipatory effects of the announced regime).

None of these assumptions is likely to be entirely accurate, and the direction in which they might bias the conclusions is unclear. This version of the counterfactual has the attraction of transparency and simplicity, but, given the obvious potential biases, it was checked against alternative specifications, able to relax some of the assumptions.

The alternative counterfactual was based on econometric modelling of emissions from individual units, based on a model estimated using panel data covering both Phase 1 and non-Phase-1 units, over the ten year period 1988-97. Including data for non-Phase-1 units makes it possible to disentangle general influences on emissions rates from those specific to the units affected by Phase 1, and the ten-year run of data allows long-run emission trends to be modelled (such as the switch to lower-sulphur coal, and also allows for the possibility of anticipatory effects, in advance of the introduction of Title IV.

On average over the period 1995-97, the emission reductions estimated using the simple and econometric counterfactuals were similar - about 4 million tons per year of SO2 emissions abatement. Both, therefore, show significantly higher abatement attributable to Title IV in these years than the approximately 3 million tons reduction in emissions, as compared with the actual level in 1993. However, the time profile of the estimated reductions differs between the two counterfactual estimates. The econometric estimate shows a certain amount of anticipatory abatement, in 1993 and before, that can be attributed to Title IV. On the other hand, it attributes less abatement in the later years to Title IV than does the simple counterfactual.

Table 5.1. Title IV emissions reductions estimated against two alternative counterfactuals, compared with emission reductions from 1993 level

	Simple counterfactual	Econometric counterfactual	Emissions reduction from 1993 actual emissions
1993	0	398	0
1994	0	537	200
1995	3961	4135	3232
1996	4076	3791	2934
1997	4257	3955	2987
Average 1995-97	4098	3966	3051

Source: Ellerman *et al* (2000). Reductions against simple and econometric counterfactuals, Table 5.1. Reductions against 1993, calculated from Table A.3

The authors indicate a preference for using the results of the econometric counterfactual in assessing the overall effects of Title IV, because it can include a wider range of effects. However, they note that it is less useful when analysing effects at a finer level of detail (e.g. at state level, to assess the geographic distribution of effects), because the underlying model is relatively imprecise in its predictions of the performance of small groups of generating units. For these analyses they revert to the simple counterfactual.

Specification of a counterfactual, in this case at the level of individual units, also plays a central role in the study's assessment of the effects of the "voluntary" compliance program. This was used heavily, with 138 substitute units as compared with 263 Table A units, but contributed little to the overall reduction in emissions. The report argues that this reflected an "adverse selection" problem, of the following form:

Under this provision of Title IV, firms were permitted to substitute alternative units for those designated as Phase 1 units ("Table A units"), subject to the condition that total generation capacity of the substitute units equalled that of the initially-designated Table A units. The substitute units received allocations of allowances equal to their historic emissions, and were then treated in the same way as original Table A units.

If all units receive an allocation of allowances equal to their counterfactual emissions, then the substitution of units will be attractive only where abatement costs are lower for the substitute units than for the originally-designated units. Emissions reductions will then be achieved at lower aggregate cost than if this flexibility had not been available. However, where allowance allocations differ from counterfactual emissions, a further consideration (in addition to abatement costs) will enter into the decision whether to substitute units. Other things being equal, units will be more likely to be put forward as substitute units where they would receive an over-generous allocation of allowances (i.e. an allocation greater than their counterfactual emissions), while Table A units with an allocation below their counterfactual emissions will tend to be taken out from the programme and replaced by substitute units.

An initial indication of the operation of this adverse selection is given by Table 5.2, based on table 8.1 in Ellerman *et al* (2000), which compares trends in heat input and SO2 emissions before and after the start of Phase 1. Prior to Phase 1, trends in heat input (broadly-speaking, in utilization) differed radically between the units put forward as substitute units, and the remaining non-Table-A units. Utilization fell by about 9% in the substitute units, but rose by about 24% in those not put forward as substitutes. Utilization in the original table A units was broadly constant. Since the allowances allocated to substitute units were based on data on emissions at least five years prior to the start of phase 1, it is evident that these units received allowances on a basis which was a substantial exaggeration of their level of activity at the start of Phase 1. In addition to these trends in utilization, it is also evident that the substitute units had on average a greater improvement in SO2 emissions rates prior to Phase 1 than units not put forward for substitution. The substitute units reduced their

emission rate by 28% over 1985-93, compared with 14.5% achieved by non-substitute units. The effect of this can be seen by comparing the 1995 allowance allocation with 1993 emissions. The allowances allocated to original Table A units in 1995 were about 5% lower than their 1993 emissions, while the substitute units were allocated allowances 46% higher than their emissions at the start of the programme.

Once included in phase 1, the substitute units broadly matched the percentage reductions in emission rates achieved by original Phase 1 units (39% and 36% reductions respectively), while no emission rate reductions were achieved by the units not put forward as substitutes. The similar average improvement in emissions rates in Table A and substitute units is not, however, evidence that differences in abatement costs played *no* role in substitution decisions, since abatement costs in those units withdrawn from Phase 1 could have been higher than the average abatement costs for Phase 1 units.

Table 5.2. Title IV: the voluntary compliance programme

	Table A units	Non-Table-A units substituted in place of Table A units	Non-Table-A units, not put forward as substitute units
Number of units	263	138	421
Capacity (1000MW)	88.0	28.8	92.7
Heat input:			
1993 against 1985-87 baseline	+0.7%	-8.6%	+23.5%
1997 over 1993	+14.9%	+14.8%	+24.5%
Emissions rate			
% change 1985-93	-22.2%	-27.9%	-14.5%
% change 1993-97	-36.4%	-39.2%	+0.9%
Total SO2 emissions			
1985	9302	994	1967
1993	7579	655	2409
1997	4769	543	2993
% change 1985-93	-18.5%	-34.1%	+22.5%
% change 1993-97	-37.1%	-17.1%	+24.2%
1995 allowances, as% of 1993 emissions	95.2%	146.4%	

Source: based on selected data from Ellerman et al (2000), Table 8.1.

At the very least, excess allowance allocations seem to have been a significant factor in the substitution decisions. To assess their significance as compared with relative abatement costs (and a further incentive to substitute arising from the NOx grandfathering provisions), the study estimated econometric models of the decision to volunteer individual units for substitution.

The models took the form of a discrete-choice (logit) model of substitution, in which substitution decisions in 1995 are modelled as the outcome of an underlying (unobservable) assessment of the net benefit from substitution. The models included a number of variables reflecting different potential influences on the net benefit, including

- the difference between the allowance allocation and counterfactual 1995 emissions, estimated in three different ways
- variables to proxy unit-specific SO2 abatement costs
- variables reflecting the NOx grandfathering aspects of substitution

- other variables, reflecting factors such as a common smokestack between Phase 1 and Phase 2 units (affecting measurement costs), the role of long-term coal contracts, unit-level uncertainty about utilization levels, etc.

The estimated models are generally stable across alternative specifications, and the key coefficients are generally significant and have the expected signs. The relative importance of the various influences on substitution was then assessed, by showing the impact of each variable on the probability of substitution. Where a unit attracts excess allowances, this has a large impact on the probability of substitution. Low control costs - and, to a lesser extent, NOx grandfathering incentives - also appear to make a sizeable difference to the substitution decision. The authors conclude that, given the asymmetry of information between the authorities and firms, there is a difficult trade-off between benefits and costs of opt-in provisions, such as those for Phase 1 substitution. Substitution can have the benefit that abatement shifts to units with lower abatement costs, but this gain may be partly offset or even reversed, if the excessively-generous allowance allocations made to some units becomes the dominant influence on substitution decisions.

5.4 The NAPA evaluation report on performance of trading systems for VOCs in four US states

Hix, Ruder and Sugarman (2000) is an evaluation study commissioned by the US National Academy of Public Administration, which has compared the performance of trading systems for VOCs in four US states, Louisiana, Michigan, New York and Texas. The study includes two types of VOC trading programme – "offset" programmes in Texas, Louisiana and New York (the first involving much more extensive trading activity than the other two), and more recently-established "open market" emissions trading programmes in Michigan and Texas.

The methodology of the study is based on literature reviews, interviews with various participants (regulators, trading brokers, industry representatives and environmentalists), and various data obtained about the operation of the schemes in each of the four states.

The institutional context of the study is that it forms one of a series of independent evaluations of innovations in environmental management commissioned by the National Academy of Public Administration, following a request by the US Congress in 1998. The research costs of the exercise were funded by the US EPA. The series of evaluations was commissioned and guided by a panel of NAPA Fellows and other experts, and researchers commissioned to write the studies were encouraged to offer their own findings and recommendations. The commissioning panel in turn has used the individual evaluation studies as a foundation for its own report and recommendations to Congress and the US EPA. All the studies – and the panel's subsequent report – have been published in full on the NAPA web-site.

Overview of the instrument

VOCs (Volatile Organic Compounds) are a group of pollutants, which generate environmental concerns of two sorts. VOCs are one of the precursors of ground-level ozone, a major pollution problem in many US urban areas. In addition, a number of VOCs have significant carcinogenic properties, or other adverse health effects.

VOC emissions trading is one particular element within the wider heading of emissions trading in the US. Only some states operate trading arrangements for VOCs, and the systems of trading take different forms in each state (subject to broad overall guidance and oversight from EPA). There are, however, two broad types of VOC trading which are the subject of this report, offset trading, and open-market trading.

Offset trading. The 1977 amendments to the Clean Air Act authorised offset trading as a way of allowing some relaxation in the prohibition on new sources, or major modifications to existing sources, in "non-attainment areas" – highly-polluted urban areas which did not meet national air quality standards (NAAQS). Under offset trading, such emissions sources could increase emissions by purchasing a sufficient quantity of emissions reductions ("offsets") from another emissions source within the area. The emission reduction credits arising within offset trading take the form of prospective, permanent, reductions in annual emissions, arising through plant closure or permanent and enforceable modifications to reduce emissions. Sufficient have to be purchased to cover the planned annual emissions from the proposed new facility or modification, and in addition to cover the compulsory "retirement" of credits. The amount to be retired varies with the environmental conditions of the area. Depending on the non-attainment category, a further 10% to 50% of credits must be purchased for retirement, in addition to the credits used.

Open-market trading. This allows firms flexibility in the achievement of certain air pollution regulations (RACT requirements, and certain state-level requirements that are more stringent than those of EPA) through the exchange and banking of emission-reduction credits. In the mid-1990s EPA encouraged the development of open-market trading through its Economic Incentive Program, and published a proposed open market trading rule. Credits can be generated through pollution control in excess of regulatory requirements and plant shut-downs. Unlike the abatement which generates offset credits these can be temporary, rather than permanent, reductions in emissions. As a result, the trading unit is specified simply in terms of the quantity (x tons) of emissions reduced (rather than the emissions rate which is the unit traded in offset trading), and the credits can be used in the same year, or banked for use in future years. Some proportion (usually 10 per cent) of the credits is retired, so that trades result in a net decrease of emissions. Purchased credits can be used for a range of purposes, including offsets for new or modified sources, to allow operational flexibility (by permitting temporary pollution spikes), or as an alternative route to compliance with permit limits.

The evaluation finds some differences in the extent of VOC trading activity across the state programmes studied. All three of the state offset programmes had generated a significant number of credits – between 2,500 and 6,000 tons of VOC credits per year, generated by some 30-60 individual sources. Some 2,700 tons of offset credits generated in Texas were used (about half the total generated), but less than one fifth of the offset credits generated in New York were used. Data on credits used was not made available by Louisiana. Of the two open market schemes studied, that in Texas had involved negligible trading activity, while that in Michigan had recorded around 5000 tons of VOC credits generated, by some 25 individual sources, with less than 200 tones used.

Key findings

Underlying the analysis of the report are two key critical observations - that in some respects VOCs are less well-suited to regulation though emissions trading than some other pollution problems, and that the form of the emissions trading programmes that have been implemented is less capable of delivering significant economic and environmental gains than alternative trading instruments.

The first major conclusion, highlighted at the start of the "Findings and Recommendations", is that "VOC trading creates complexity and risk concerns not associated with trading in other pollutants", such as sulphur dioxide or NOx (Hix, Ruder and Sugarman, 2000, p14). The regulation of VOCs through emissions trading is less straightforward than emissions trading for sulphur dioxide, because VOCs are a whole class of compounds. This has two direct implications. First, continuous emission monitoring is more costly and difficult than continuous monitoring of a single pollutant such as SO2. Consequently, the VOC trading programmes studies relied on emission quantification methods based on engineering estimates of emissions, rather than continuous measurement. Second, a given quantity of VOCs emitted in a particular locality may have differing implications for the health

of local residents, depending on the proportion of the total which comprises those VOCs with the greatest potential to damage health. Consequently, the programmes have needed to monitor the health risk created by changes in the level and composition of VOC emissions from each source, leading to much greater regulatory oversight of individual trades than with SO2. The severe risks to health if hot spots arise through trading have led to significant opposition to VOC trading from EJ (environmental justice) organisations, and it is clear that currently-available arrangements for monitoring and approval cannot wholly allay these concerns.

One of the key recommendations of the study is that encouragement should be given to the adoption of emissions monitoring tools that provide greater certainty than existing engineering-based methods, and that provoke less controversy. There has been disagreement between states and EPA (and, apparently, within EPA too) over the quantification protocols that should be used for engineering-based emissions estimation. Developing affordable continuous emission monitors for VOCs would avoid the need for controversial indirect ways of estimating emissions, and increase the certainty of emissions reductions. "If that proves technically or economically infeasible, regulators should focus trading programs on more easily-measured pollutants until affordable continuous emission monitors become available."

A second major critical conclusion of the evaluation is that alternative designs of emissions trading would have been more likely to lead to significant benefits. The forms of emissions trading employed, offset and open-market programs, which have been grafted on to existing command-and-control regulatory structures rather than being designed as a treading regime from scratch, have the disadvantage that they "lack sufficient motivation for sources to trade" (Hix, Ruder and Sugarman, 2000, p. 14).

This observation seems to incorporate two separate problems. One is that the nature of offset and open-market programs restricts the types of pollution redistribution than can be allowed to take place through trading. In contrast, a cap-and-trade system such as operates for SO2 does not place limits on the way in which reductions in emission are achieved, or the reasons for the additional emissions for which permits are purchased. A second issue is that the heavy regulatory scrutiny involved in every individual transaction under an emissions credit system creates substantial transactions costs which can outweigh the benefits of trading. In particular, sources which already maintain compliance with regulatory requirements may find that the additional transactions costs incurred in trading can exceed the profit to be made.

Costs and benefits of trading

The methodology of the report involves less quantitative modelling and analysis than most of the other evaluation studies discussed. This partly reflects difficulty in obtaining detailed data from some of the regulatory agencies involved, as discussed below. It also reflects the study's emphasis on investigating the views and experiences of different participants. As a result, although the study does draw together an overall assessment of the costs and benefits of the trading systems, it also contains a significant discussion of other issues relevant to the overall assessment of emissions trading - namely issues to do with the use of commercially-sensitive data, and concerning the role of staff attitudes in programme effectiveness.

The evaluation concludes that VOC trading programs can lead to substantial economic benefits and modest environmental gains beyond those achieved through traditional regulatory structures. The economic benefits from offset trading are the scope for construction of new major sources, or modifications to existing sources, and the benefits from open market trading take the form of better compliance and operational flexibility.

Staff costs are incurred by the state environment agencies in initiating and operating trading programmes. The study describes (p. 48) the quite lengthy process involved in initiating Michigan's trading programme. The on-going additional staff requirement for a trading programme was estimated by Michigan and Texas as the equivalent of some 4 to 6 full-time employees - a non-trivial cost, but probably small in relation to the economic benefits from unlocking new investment in non-attainment areas.

The report analyses transactions costs of various sorts which arise in offset and open market trading schemes – the costs to participants associated with learning about the rules of the scheme, search costs involved in finding potential trading partners, the costs of price uncertainty, and the costs of uncertainty about the validity of emissions credits being purchased. It is noted that emission credit brokers can play an important role in promoting the trading programme, by seeking out firms with potential to gain from buying or selling credits, by promoting the dissemination of price information and by acting as market makers, and by advising and educating unfamiliar participants in trading.

Procedures are necessary to check on the validity of credits that are traded, and these can be a significant part of the administrative cost to the public agency and of the compliance cost burden to firms. The report notes two different approaches. In the case of offset programmes, the state determines the validity of a credit (in particular that the emission reductions claimed have actually taken place) before allowing it to be used by the purchaser. Indeed, in some cases (such as New York) the state verifies the validity of the credit before it is placed on the register. The effect is that there is a substantial compliance cost burden even before trading takes place, but, on the other hand, purchases can be sure that the credits they have purchased will indeed be valid when they come to use them. On the other hand, credits in open market trading tend to be subject to less prior verification, and some systems explicitly operate on the basis of "buyer beware". The effect of this is that credits have a somewhat uncertain value, and buyers can find themselves risking the need to provide compliance information for the source of the credits.

Both offset and open market trading have provisions designed to lead to environmental benefits, because a proportion of the emissions saving is "taxed" in each transaction. Nevertheless, given the limited number of firms that engage in trading, the environmental gains will tend to be modest - in the examples studied, the environmental improvement was less than one per cent of aggregate emissions in the relevant geographical area.

It is not always clear, however, that these emission reductions arise only as a result of the trading programme. Indeed, there appears to be recognition that a proportion of credits that arise, especially for shutdowns, reflect emission reductions that would happen anyway, regardless of the presence of the trading regime. Significantly, the report notes that shutdown credits are often referred to as "anyway credits". However it notes that shutdown credits have often in practice been less attractive to purchasers, because there has been some uncertainty about whether credits arising from shutdowns will continue to be recognised by EPA, and because if it subsequently becomes necessary to check on the status of credits this may be more difficult if the seller was a firm that has since gone out of business.

Information issues

The report draws attention to issues of data availability and public access to information. Individual citizens have the right, under the Clean Air act, to file federal lawsuits against individual pollution sources for non-compliance, or against public programmes that they believe harm themselves or the environment. EPA's guidance on economic incentive programmes includes rules to ensure that emissions trading does not weaken the ability of individual citizens to identify, and take action against, individual polluting sources. State programmes are required under this guidance to provide details of the emission changes by participating sources. The report notes that all the state

programmes studies comply with the EPA's guidelines, but that they vary considerably in terms of the amount and accessibility of the information placed in the public domain. Some provided extensive, online, information sufficient to track the life-history of each credit from generation to use. Others provide more limited information. One issue which limited the comparative data available for the study was that some states limited the data put into the public domain because of concerns about the disclosure of confidential business information. It was observed that requiring firms to release certain types of information about manufacturing processes could act as a significant disincentive to participation in trading. There is clearly a difficult balance to be drawn here between maximising the potential cost savings through trading, protecting the rights of citizens to access to relevant information, and ensuring the availability of sufficiently-detailed data in the public domain that well-informed evaluations can be undertaken by independent researchers.

Staff attitudes

The study noted some issues about program staff attitudes to trading. Some agency staff, particularly those with responsibility for enforcement, were sceptical about the value of trading, and tended to view it as leaving loopholes for poor compliance. "We found that these concerns result more from a lack of information and communication rather than an analysis of program enforceability." There are gains from providing staff who may come into contact with the programme with education about its rationale and objectives, since individual staff may play an important role in maximising the benefits from the scheme. The study notes that that attitudes of agency staff can change over time. In Michigan, for example, although there was initial hostility, "once field staff and permit engineers gained exposure to the program, their support of the program increased dramatically" and many began actively to facilitate and encourage trading-based options for the firms with which they were in contact.

5.5 Comparative discussion of the evaluations

The four case studies of *ex post* evaluations of tradeable permit schemes exhibit some striking contrasts in methodology and approach, as well as suggesting some general lessons. They differ significantly in terms of institutional location, timing, and analytical style.

Institutional location

Only one of the four case studies, the RECLAIM study, was conducted within the agency responsible for operation of the instrument. The other studies were broadly-speaking "external", though with significant differences.

In broad terms, as discussed in Section 4, a trade-off might be anticipated between various costs and benefits of conducting evaluations outside the agency responsible for operating the policy. The advantages of outside evaluation are primarily in terms of actual and/or perceived objectivity. The disadvantages are in terms of poorer linkages with the policy - in terms both of poorer access to information, and, possibly, less scope to influence future modifications (if, for example, agency staff are less likely to "take ownership" of critical findings and recommendations from an external report). These matters are complex and far from clear-cut, and how they work out in practice will vary from case to case, and depending on the political and administrative cultures of the country concerned. Since all of the studies covered in this report relate to the US, all have been undertaken in the relatively open administrative culture of the US, and not all of the lessons will be transferable outside the US. Nevertheless, it is interesting to note some of the strategies used to offset some of the disadvantages normally associated with a particular choice of institutional location.

The RECLAIM audit was written by SCAQMD staff, but the work was conducted under the guidance of a RECLAIM Advisory Committee, with members drawn from regulated firms, environmental organizations, academic and policy research institutions, regulatory agencies, and others with relevant expertise. The committee provided input to the audit, and commented on a preliminary version of the audit report. A noteworthy feature of the audit report, which has had the effect of clarifying differences in judgement and perspective between members of the advisory group, is that the comments of dissenting members of the committee are published within the report, along with the response of SCAQMD staff. This format seems a creative and potentially valuable response to the danger that internal reports will avoid criticism, and the equal danger that a broad representative panel of external stakeholders, with different perspectives, might only be able to agree on a bland and insubstantial report.

The Ellerman et al study of acid rain trading is in the form of a book published by an academic publisher, but the work underlying the book has also fed into earlier policy studies, including NAPAP's Quadrennial Report to the US Congress. The report is a major study, undertaken over a significant period of time, requiring substantial resource inputs, and able to draw on a significant body of prior and parallel academic research. It has reached correspondingly substantial, well-supported and comprehensive conclusions, covering many of the key areas of interest and uncertainty in the evaluation of tradeable permit schemes.

Although funded in part by the agency responsible for the programme, the resulting study is an independent and objective assessment, using transparent and clearly-documented research methods. A considerable amount of the data relevant to the acid rain programme is in the public-domain, which helps to promote peer discussion, criticism and verification of research findings relating to this scheme. The study has been able to draw on a considerable body of other research, in academic institutions and elsewhere, on the operation of the acid rain programme. This active research community of scholars working on the same topic provides valuable underpinning for the rigour and independence of the report's findings.

It is noteworthy, however, that nearly all of the US studies reviewed in this paper by researchers working towards the "independent" end of the spectrum, including a number by individual academic researchers, have been conducted in an environment where there is considerable scope for information flows and interaction with the public agencies responsible for the policy, without undermining the credibility that independence provides. The ability of researchers to conduct effective evaluations is enhanced by the US culture of openness with public information (including the enormous amount of data about, for example, permit trades which is posted on open-access web sites). In addition, a number of the studies, for example by researchers working through research centres such as Resources for the Future, benefit from the close relationships which these centres have built up with relevant public officials, while at the same time maintaining a track record of objectivity, by supporting research using well-documented data and methods, and subject to the rigorous scrutiny by academic peers that arises in the course of academic seminar discussion and publication.

Timing

The lead trading studies are evaluations of a completed policy episode, in which trading was used for a short period as an adjustment mechanism. Almost all of the lead studies are wholly-retrospective evaluations, published after the trading scheme ended. The studies therefore may provide lessons for other possible tradeable permit applications, but have no potential role in policy-improvement in relation to the particular application studied. In some respects, the lack of any role in subsequent policy-development may be an advantage in evaluation, since it may be possible to obtain data from the affected firms without fear of this being contaminated by their attempts to influence future amendments and developments of the instrument. Once the trading scheme has ended, firms may also be less concerned about revealing information of commercial value, such as data on

marginal abatement costs for different technologies or the prices at which permits were transacted, which they might not wish to be communicated to other firms.

In contrast, the three other studies are of ongoing applications of tradeable permit mechanisms, and all have been conducted with a view to policy improvement in relation to the particular application studied. The study of RECLAIM, undertaken by SCAQMD, was a mandated three-year audit, intended to constitute a systematic appraisal of the performance of the scheme, to identify failings and areas for improvement. The Ellerman et al study of acid rain trading covers the early years of a scheme intended to be a long-term mechanism for the regulation of SO2 emissions. The work underlying the study has fed into a policy appraisal of the initial years of the mechanism, in the form of NAPAP's Quadrennial Report to the US Congress (NAPAP, 1998). It is clear from both these studies that a three-year perspective is sufficient for many of the effects of a tradeable permits regime to be observed and assessed, even though the long-run adjustments of market participants to the system may be far from complete. (In this respect, as noted in section 4, it may be possible to learn rather more about tradeable permit systems than could be learned about environmental taxes and charges over a similar time scale.) In both the RECLAIM and acid rain studies there was evidence that permit market participants took some time to settle into the new regime. Initial prices for acid rain allowances took a couple of years to stabilize, but thereafter the market appeared to have reasonably coherent and stable allowance pricing, and significant volumes of trading activity. Nevertheless, although three yeas may be a sufficient perspective to see a market develop and, perhaps, stabilize, it may be insufficient time to assess its ability to cope with shocks and unforeseen contingencies of various sorts. The recent dramatic price escalation in the RECLAIM market, which took place after the three year evaluation, may be an indication of underlying deficiencies in the mechanism itself (in particular the absence of scope for banking, or other forms of inter-temporal shifting of emissions and abatement activities), or it may reflect failures of market participants to anticipate the consequences of the steady tightening of the limit on aggregate emissions. Substantially longer experience with the system may be needed for these issues to be fully investigated.

Style

Ellerman and some of the lead studies are quantitative in style, based on detailed, systematic quantitative data, making heavy use of sophisticated econometric techniques, explicit quantitative modelling of counterfactual scenarios, etc. At the other extreme, the VOCs study is almost wholly qualitative in style, being based on interviews with various affected parties, and having little access to relevant quantitative data.

The difference in style has implications for what can be learned from each study. The focussed quantitative analysis of the Kerr and Maré (1998) study of lead trading, for example, is able to generate firm evidence on the practical significance of transactions costs in diminishing the overall gains from trading. Similarly, the quantitative modelling by Ellerman et al (2000) of the participation in Title IV's voluntary compliance programme is able to demonstrate the practical importance of the "adverse selection" effect, rather than simply its theoretical possibility. In the other hand, the NAPA study of VOCs trading, employing a more qualitative methodology, is able to provide evidence that could not be obtained through more quantitative research approaches on some of the channels by which soft effects (such as the role of staff attitudes) may affect the functioning of a trading system.

6. Conclusions and suggestions for further work

6.1 Conclusions

Evaluation of tradeable permit systems is desirable because it can shed light on the kinds of environmental problem most amenable to regulation using tradeable permits, and on the design features of permit trading arrangements that function most effectively. This information can feed back into policy reappraisal and modification in the particular application studied. In addition, while tradeable permits remain relatively novel instruments, with few long-standing applications to environmental policy except in the United States, evaluation evidence on the performance of tradeable permits can be of considerable value in the processes of cross-country policy learning and diffusion. Although theoretical and quantitative *ex ante* studies have shown that there are potentially substantial cost savings from regulating environmental problems using tradeable permits, some of the key features that might affect the performance of trading schemes cannot be assessed except through the observation of practical experience. The significance of transactions costs in preventing firms realising all theoretically available cost-reducing trades, and the extent to which permit markets function efficiently, without excessive problems of monopoly power, are both key factors which could have a significant bearing on the performance of tradeable permit schemes, and their practical significance can only be assessed by observing real-world experience.

The studies reviewed here demonstrate the feasibility of evaluation, and show some of the things that have been learned. Perhaps the most important lesson is a general, and positive, impression: that tradeable permit instruments can function effectively in environmental policy applications, and that the flexibility they offer compared with conventional regulation is clearly used by firms, to achieve at least some of the savings in compliance costs that theoretical and quantitative *ex ante* studies have predicted. Naturally, questions could be raised about the representativeness of the tradeable permit applications, or of the evaluations, selected for this paper. However, it is difficult to argue that this positive general conclusion arises only because favourable cases were selected for study, since the applications covered include a significant proportion of all large-scale tradeable permit applications in the US. Of course these may well be applications in policy areas that are particularly amenable to regulation in this way, but no claim is being made that tradeable permits are appropriate to all environmental problems. As Ellerman *et al* (2000) make clear, the results of acid rain trading under Title IV show that emissions trading is a valuable policy tool in appropriate circumstances, but there remain environmental problems (including those involving highly-toxic chemical emissions, or effluents which cannot be easily measured) where emissions trading may not be a sensible policy approach.

The paper has also set out a suggested framework for evaluation of tradeable permit schemes, identifying some important priorities for investigation, and some key issues in the design of the evaluation.

It has argued that evaluation studies should focus on those aspects of the performance of tradeable permit systems about which least can be learned from theory and *ex ante* simulation studies. Amongst these issues are the efficiency of the market for pollution permits, the pattern and scale of polluter responses to the existence of potential abatement cost savings, the scale and nature of transactions costs, costs of administration, and the practical significance of "soft" effects such as the awareness and attitudes of staff in affecting how well tradeable permit schemes function in practice.

Key issues in the design of the evaluation, which may have a considerable bearing on how much is learned, are the need for clarity regarding the policy baseline (i.e. with what other instrument, if any, are we comparing the performance of tradeable permits?), the need for careful modelling of the

"counterfactual" (i.e. what would have happened otherwise?) rather than simply attributing all "before-and-after" changes to the effects of the policy, ensuring good access to data (which may be affected by the institutional location of the study, and may be significantly enhanced by advance planning at the time when policies are introduced), and high standards in the conduct of the research (promoted, in particular, through open discussion and the rigours of peer review and criticism).

6.2 Suggestions for further work on ex post evaluation of tradeable permits

This section makes some concluding suggestions for future directions in work on *ex post* evaluations of tradeable permits, under five possible headings. The data collected and analysed under these suggested studies could also be cross-checked with the findings of earlier OECD work on the design and implementation of domestic TP systems (*Domestic Transferable Permits for Environmental Management: Design and Implementation,* 2001, OECD).

(i) Institutional location, authority and credibility in evaluation research

The institutional context within which evaluation studies are conducted can have a significant influence on the success of the evaluation, and on its effectiveness in stimulating reforms and refinements to the policy. Typically, one might expect evaluations conducted by the agency responsible for designing or implementing the tradeable permit mechanism to have better access to certain types of information than "outside" evaluation by academic research centres, but to face problems of credibility regarding the objectivity of the findings. There is no single clear conclusion about the best institutional location for evaluation research, and the trade-off between information and objectivity differs, depending on the policy instrument concerned, and the research, administrative and political traditions of individual countries.

A number of the US studies reviewed in this paper have been conducted by researchers working towards the "independent" end of the spectrum, including a number by individual academic researchers. However, the RECLAIM study was conducted under the auspices of the responsible agency, and some of the academic researchers responsible for other studies were working in institutions which had good institutional access to public officials and the policy community.

A suggestion for further work would be for a study which would look at these various institutional models in more detail. It would be useful to know more about, for example, the ways in which research institutions such as Resources for the Future and the MIT Centre for Energy and Environmental Policy Research maintain both their close engagement with policy and implementation, and their high reputation for objectivity. Likewise, there were a number of interesting features of the institutional arrangements for evaluation of RECLAIM, which seem intended to offset the risk that this internal evaluation would otherwise have lacked rigour and credibility. Were these arrangements (e.g. the steering group) effective, and a good model for other agencies to follow in setting up credible arrangements for evaluation? These issues are difficult for someone not well-informed about the context of the US studies to assess, and there may be some important lessons for policy-makers in other countries.

(ii) Methods for defining the counterfactual in ex post evaluation research

Establishing what would have happened in the absence of a particular policy initiative is perhaps the most intractable element in an *ex post* assessment of its effects. However, in the absence of an effective way of doing this, there is a real danger that effects (either positive or negative) which would have happened otherwise are incorrectly attributed to the policy concerned.

Sophisticated modelling of the counterfactual was a key feature of the Ellerman *et al* study of acid rain trading discussed in section 5.3, and it may be worth investigating the approaches taken by this study in more detail. What are the relative advantages and disadvantages of econometric and less-sophisticated approaches to modelling the counterfactual? How important is it to model the counterfactual for individual emissions sources as well as in aggregate terms? How far ahead can the counterfactual be estimated with any precision? What does this imply for the ability of a study to model anticipatory effects of policy as well as effects following the date of its implementation, and for the timing of evaluation studies? A relatively technical study covering these issues might well provide useful guidance on research design for other evaluation researchers.

(iii) Transactions costs

Another suggestion for further work is for a study which would discuss the various forms of transactions costs which might impede the efficient functioning of permit markets, and how observation-based evidence on their pattern and significance might be obtained. The paper by Kerr and Maré provides a sophisticated analytical framework for discussing transactions costs, as well as a (somewhat indirect) methodology for their quantification. The paper might compare this approach with the potential for other, more direct approaches, perhaps based on questionnaires to polluters and other participants in permit systems.

The study might also review whether there is scope for comparing *ex ante* and *ex post* approaches to evaluation of tradeable permits. Probably the most likely reason for divergence between the two is the presence of transactions costs not accounted for in the *ex ante* studies. Does *ex post* evidence on the scale and pattern of transactions costs indicate that we should discount or qualify the claims about the efficiency of tradeable permit mechanisms, made on the basis of *ex ante* studies?

(iv) Efficient permit markets

A fourth direction for further work would look at the institutions and procedures for trading permits, and their efficiency. What are the signs of an efficient, or inefficient, permit market? How can we tell whether the market functions well, or whether its efficiency is undermined by transactions costs, monopoly power, or other factors?

A considerable amount of data about permit prices is available from brokers and other sources. Does this tell us anything about the efficiency of permit markets? For example, can we learn anything by measuring the stability and predictability of permit prices, or from the relationships between permit prices for emissions in different years, or for the same year at different times? Do episodes such as the rapid escalation of RECLAIM permit prices in the last two years disclose anything about the efficiency with which permit markets transmit information about marginal abatement costs between market participants?

(v) Costs of evaluation and access to information

Not all of the evaluation research described in Section 5 of this paper was commissioned as a result of legislative or policy decisions to undertake a formal evaluation of the tradeable permit mechanism. Some of the studies - for example, some of the lead trading studies - were initiated by independent researchers as an academic research project, aimed primarily at academic publication. Such independent research activities can expand the amount of evaluation evidence available, and may make a substantial contribution to innovation and the development of good practice in evaluation research.

One factor that may greatly affect the amount of evaluation work originating from academic and other independent sources is the amount of data about the operation of programmes routinely placed in the public domain. A considerable amount of administrative information about the US programmes discussed in Section 5 is available on-line, or from readily-accessible public sources. The accessibility of this data increases that chances that independent researchers will identify possibilities for evaluation research, and that such research will partly, therefore, arise spontaneously, without explicit legislation or other policy decisions. By contrast, administrative systems that are less open in the provision of public domain data will provide less encouragement to evaluation. Evaluations may need to be explicitly commissioned, and access to data discussed and negotiated, processes that are likely to increase evaluation costs, and reduce the amount of flexibility, experimentation and innovation in evaluation research.

These issues are, of course, not limited to the evaluation of tradeable permits, but arise in many areas of policy where evaluation is, or could be, employed, including wider environmental and resource policies, labour market policies, education policies, etc. A useful direction for future work would be to examine the relationship between data accessibility and the amount and type of evaluation research which is undertaken. What kinds of institutional relationships are most effective at promoting vigorous external evaluation, and how far does this affect the feasibility, costs and standards of internal or explicitly-commissioned research? What constraints, such as considerations of commercial confidentiality and administrative cost, are there on making the kinds of data that might be useful for evaluation research more widely available? How are these issues addressed in other areas of policy, and what lessons might be learned that would help to promote greater evaluation of environmental policy instruments?

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Part III

NEW AREAS FOR APPLICATION OF DOMESTIC TRADEABLE PERMITS

Chapter 3

DOMESTIC GHG EMISSIONS TRADING SCHEMES: RECENT DEVELOPMENTS AND CURRENT STATUS IN SELECTED OECD COUNTRIES

by

Kumi Kitamori
Environment Directorate, OECD

1. Introduction

This paper attempts to provide an overview of the current status of domestic GHG emission trading schemes in some of the OECD member countries initially as of June – August, and updated in December 2001, with a focus on how they are being combined with other policy instruments for domestic actions towards meeting their Kyoto goals[1]. While some countries have officially adopted domestic emission trading as one of the policy tools for meeting their Kyoto targets, others are still debating its suitability in the context of their own domestic climate policies. Among those that are considering introduction of domestic emissions trading schemes, some are at advanced stages of designing and legislating such schemes, while others have just recently started discussions among policy makers and stakeholders. To date, only Denmark has actually launched a domestic trading programme in January 2001, soon to be followed by the UK in April 2002[2]. Only domestic emissions trading schemes being considered or adopted as an official national policy measure are discussed in this paper. Although there are several pilot, voluntary (e.g. by the private sector), and regional (i.e. at sub-national level) initiatives, those are not covered here.

At this point in time, the proposed domestic trading schemes vary from one country to another in terms of core design criteria such as sector coverage (and thus the GHG emissions captured), permit allocation/crediting methods, etc. Also, the ways in which domestic emission trading is combined with other policy instruments such as energy or carbon taxes and voluntary

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1. A country can meet its GHG emissions reduction target set forth under the Kyoto Protocol through two channels: (i) ***domestic actions***, i.e. actual emission reductions at domestic emission sources, and accounting for domestic carbon sinks (but the rules for the latter are not yet finalised), and (ii) "***flexible mechanisms***" which allow a country to procure additional emission allowances, or Annual Assigned Amounts, from other countries through ***international emissions trading (IET)*** or through taking credits for projects that result in emission reductions in other countries through ***Joint Implementation (JI)*** in other industrialised nations and ***Clean Development Mechanisms (CDM)*** in developing countries. This paper only focuses on OECD counties' use of domestic emissions trading and other policy instruments for domestic actions.

2. The UK Emissions Trading Scheme was officially launched on 14 August 2001, when pre-registration period for interested participants started. Followed by this preparatory period, trading scheme to is scheduled to be operationalised in April 2002.

approaches differ from one country to another, and this is due to a number of factors. In this context, comparisons and contrasts among the diverse models adopted by different countries will be made, and the possible implications in terms of environmental effectiveness and "implementability" of the policy mixes will be reviewed.

First, this report briefly reviews the theoretical literature on tradable permits. Then, the significance of the application of the TP concept in GHG emission abatement efforts will be discussed. It will be followed by an overview of domestic GHG emissions trading schemes proposed or in place in selected OECD member countries, in the context of their domestic climate change policy packages. Then, the issues related to the combined use of TPs with other policy instruments including taxes and voluntary agreements will be reviewed. This is done by revisiting the main characteristics of taxes and VAs, and briefly reviewing their use in GHG emissions abatement. This will lead to the issues of compatibility of TPs, taxes and VAs, and which instrument is suitable under what circumstances and in what combinations will be revisited. Although there is currently limited experience with TPs combined with other policy instruments, GHG emissions trading is one area where we are beginning to see some concrete cases and proposals. Finally, an attempt will be made to draw some preliminary assessments on how best domestic GHG emission trading might be combined, (or not) with other instruments, in the context of domestic climate change and other environmental policies of a country.

2. Why Domestic GHG Emissions Trading?

There are a number of reasons for the recent interest in domestic GHG emissions trading in many of the OECD member countries. It has derived mainly from the ongoing negotiations on the Kyoto Protocol whereby Annex I countries[3] would be required to meet legally binding emission reduction targets by 2008-2012 or the "First Commitment Period". In this context, domestic emissions trading would be a strong candidate, among a range of possible policy tools, particularly for its potential to fix the guaranteed level of emission reduction. The Kyoto discussions also include international emission trading (IET) to be introduced in 2008 as one of the so-called "flexible mechanisms." Similarly, a regional GHG emissions trading programme has been proposed by the EU, to be introduced in 2005. The ongoing lively debates on these proposed international trading schemes have urged many countries to consider developing their own domestic trading schemes in preparation. Finally, there are certain intrinsic characteristics of GHGs that would make them particularly suitable for emission trading.

2.1 Domestic emissions trading in response to international and regional emission trading schemes

Many countries consider that an early start with domestic trading schemes would not only allow them to take advantage of the inherent potentials of TPs for cost-effective emission abatement, but also afford domestic industries the practical experience of "learning by doing" prior to the introduction of international and EU trading schemes. Particularly in Europe, we find more concrete proposals for domestic emissions trading, in anticipation of the EU-wide trading scheme. Some of these front-runners, such as the UK, also have the added goal of possibly influencing the design of the EU scheme by implementing domestic schemes earlier.

On the other hand, many countries also consider the possible disadvantages of earlier introduction of domestic GHG emissions trading schemes, especially after the failure of COP6 in the

3. Annex I countries are the traditional OECD member countries plus the economies in transition of Eastern Europe and the former Soviet Union.

Hague in November 2000 to reach conclusive decisions on the rules of international emission trading and other flexible mechanisms[4]. Establishing a domestic trading scheme before clear rules are established for the Kyoto and EU schemes could potentially complicate the eventual harmonisation that would be needed between these and the domestic scheme. Thus, some countries (e.g. Australia) have done considerable preparatory work for a possible domestic scheme, but maintain a "wait and see" position until definitive decisions are reached on the international trading scheme under Kyoto, and the likelihood of the Protocol itself to become effective becomes more certain.[5]

Also, early actions by a country to move ahead with mandatory domestic emission trading could put their domestic industries in a competitive disadvantage, if other countries do no follow suite and also mandate their industries to bear compliance costs to reduce emissions. However, this should be a short to medium-term concern, as it could eventually be overcome by the first-mover advantage due to technological innovation induced by the dynamic efficiency of the trading scheme. Moreover, further benefits could accrue through spill-over effects of the technological innovations to other industries. These longer-term benefits may outweigh the possible short-term competitive disadvantage.

2.2 Tradable permits as a cost-effective policy instrument

The theoretical literature on tradable permits has grown over the past 25 years, complemented by recent but growing empirical evidence from applications of the TP concept in air and water pollution management in the US and elsewhere. In essence, TPs can be environmentally effective, economically efficient, flexible, and designed so as to limit unwanted distributive effects[6]. They can guarantee *environmental performance* as they are based on setting of quantified limits or caps on emissions, which are, in the case of GHGs, the emission reduction targets for Annex I countries set forth in Annex B of the Kyoto Protocol, or a part of it.[7] However, this would be the case for cap-and-trade type of emissions trading schemes, while overall emission reduction goal in absolute terms cannot be guaranteed under baseline-and-credit based emissions trading schemes[8]. Whether the reduction targets are in absolute or relative terms, it is crucial that rigorous monitoring and verification of compliance are in place to maintain integrity of the trading scheme.

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4. The Bonn Agreement of COP6 –2 in July 2001 clarified some of these issues, but the rules for the flexible mechanisms are to be finalised at COP7 in October 2001. United Nations Framework Convention on Climate Change (2001), *Review of the Implementation of Commitments and of Other Provisions of the Convention*, [FCCC/CP/2001/2/Add.4], 11 June 2001.

5. The Kyoto Protocol becomes effective once it has been ratified by 55 percent of the signatories, representing at least 55 percent of developed countries' CO2 emissions.

6. See, for example, OECD (2001a), *Domestic Transferable Permits for Environmental Management: Design and Implementation*, for full discussion on the strengths of, as well as design and implementation options for TPs.

7. In the real world, a domestic emissions trading scheme would not be expected to cover 100% of emissions, so its cap would be set at a level lower than the country's reduction target under the Kyoto Protocol.

8. Under cap-and-trade schemes, firms are obliged to meet absolute reduction targets, while under baseline-and-credit schemes, they earn credits when and if they overachieve their baseline or targets which may be defined in relative terms (e.g. as a percentage of outputs).

A TP scheme can achieve *economic efficiency* (static efficiency) by minimising the overall cost of compliance by encouraging those firms that can abate pollution more cheaply to do so first, while allowing those with higher abatement costs to opt for buying additional permits or allowances. In the Kyoto context, the "assigned annual amounts" (AAUs) represent the quotas or permits which, within a participating country, can be distributed or "down loaded" to legal entities or emission sources, which in turn could trade them among themselves. In theory, taking advantage of the differentials in marginal emission abatement costs among diverse emission sources in this manner would lead to equalisation of their marginal abatement costs, allowing achievement of a desired level of emission reduction at the least cost to the country.

Also, TPs allow firms *greater flexibility* in their choice of means for achieving the environmental objectives. Unlike setting of legally binding technology or emissions standards under regulatory measures, domestic GHG emissions trading would leave an emission source the freedom to choose how it meets the emission reduction goal assigned to it; through switching fuels to those that are less carbon-intensive, investing in technological improvements for enhanced energy efficiency, purchasing of additional AAUs or permits, or any combination of the above. The incentives for investing in energy-efficient technologies induced by emission trading can in turn stimulate technological innovations (dynamic efficiency).

Finally, depending on its design, a TP scheme can also *better control the distributive effects* of the policies, achieving desired income distribution or transfers among different groups through the choice of initial permit allocation methods. At the same time, the converse of this means that this equity aspect arising from initial permit allocation is one of the most controversial and politically sensitive areas in the process of designing any TP programmes. Domestic GHG emission trading schemes are no exceptions to this, as illustrated in the country cases provided later. Most schemes currently propose grandfathering based on historical emissions, which would mean assignment of newly created assets to emission sources, and thus is favoured by industry. Meanwhile, initial permit allocation by auctioning is proposed in a few countries (e.g. partial auctioning proposed in Norway and Sweden). Auctioning would be closer to taxes in terms of distributive impacts as it involves transfers from industry to government, but with the former, permits would be distributed according to the firms' willingness to pay for permits, reflecting their marginal abatement costs. However, this would work only if participating in such a domestic trading scheme is mandatory. If participation is voluntary, it may face a practical problem of attracting participants if they have to assume additional costs of having to purchase permits at the initial auction. This could be overcome by combining it with fiscal measures to provide incentives for firms to participate. As the proposed UK scheme is voluntary and could potentially face this problem, fiscal incentives have been introduced in order to attract participants, as discussed later. Also, the revenues from auctions could be used to achieve certain distributive or environmental goals, for example, by using them to reduce distorting taxes.

2.3 Characteristics of GHGs and their suitability for emission trading

Traditionally, the more successful applications of TPs have been found in air pollution control, the most notable examples being the SO2 allowance trading, the Regional Clean Air Incentives Market (RECLAIM, for SO2 and Nox), and the northeastern Ozone Transport Commission (OTC, for Nox) programmes in the US. While the lessons learned from these programmes are applicable when designing and introducing domestic GHG emissions trading schemes, it is worth

noting that some of the basic characteristics of GHGs are particularly suited for emissions trading, more so than for most other pollutants[9].

First, unlike for most other pollutants, the location of emissions sources does not matter for GHGs, allowing *full geographical flexibility*. As GHG emissions quickly become uniformly mixed in the atmosphere, it makes no difference in terms of global climate change impacts if more emission abatement occurs at one source in place of another due to trading. This means that GHG emissions trading will not lead to the potential problem of local pollution "hot spots" typically associated with TP schemes for other pollutants if their geographical coverage and flexibility do not fully respect the absorptive capacity of the local environment. Similarly, *full temporal flexibility* in borrowing and banking of GHG emissions permits would make very little difference in terms of environmental impacts, because the residence time of GHGs in the atmosphere is so long, in the order of several decades. These characteristics of GHGs allow their emissions trading to take full advantage of geographical and temporal flexibility which, in turn, would allow the potentials for least cost emission reduction to be fully exploited.

On the other hand, there are some characteristics of GHGs, particularly CO2, that pose a challenge compared to other local air pollutants such as SO2 emissions. The latter are concentrated in a few industries, while CO2 is practically emitted by any sector, from more numerous point and non-point sources. Also, reducing CO2 emissions means reductions in energy flows and a shift towards more sustainable consumption patterns in general, whereas for other pollutants, end of pipe abatement technologies are more readily available. Another difference is that GHG reductions achievable are generally small compared to those in other programs (e.g. SO2 allowance programme could achieve reductions in the order of -50%). This means that the number of emission allowances needed compared to the reduction goal is larger, and thus the costs of holding permits are higher.

3 An overview of domestic GHG emissions trading schemes

To date, the responses of OECD countries to the challenge of climate change have been diverse, and so are the roles of domestic emissions trading in the context of overall domestic climate change policies. While some, such as the UK, Denmark, and Norway have decided to adopt domestic emission trading as an integral part of their climate policies, others including the Netherlands, France, and Germany have reached no definitive decisions to adopt it despite past or ongoing considerations on such schemes. Meanwhile, some OECD countries, including Belgium and Switzerland are unlikely to have domestic emissions trading schemes, or have officially opted for not including domestic trading as part of their climate policy packages. Instead, they rely on taxes and voluntary agreements as the predominant instruments for meeting their Kyoto targets.

Table 1 below summarises the current status of domestic GHG emissions trading programmes as national schemes introduced by central/federal governments in selected OECD countries as of late 2001. It is highly plausible that this will change quickly over the next 12 months or so after COP7 in November 2001, as the rules of the international emission trading scheme under Kyoto are firmed up and countries consider ratification of the Protocol. Private sector-led or pilot emission trading schemes are not covered here.

9. Mohr, E. (1992), Tradable Emission Permits for Controlling Greenhouse Gases and Complementary Policies, Climate Change - Designing a Tradable Permit System, OECD, Paris.

Table 1. Inclusion of Domestic Emission Trading in Climate Change Policy

Country	A domestic emission trading scheme?	Status
Denmark	YES	Already launched in January 2001.Three trades took place in late 2001.
The UK	YES	Pre-registration period officially launched in August 2001, with trading to be operationalised in April 2002. Related schemes, Climate Change Levy (CCL) and Climate Change Agreements (CCA) in place as of April 2001. A first "forward" trade of allowances already took place in September 2001.
Norway	YES	Expected in 2008, with introduction of international emissions trading (IET). Legislation on details of the scheme to be passed in 2002, based on the outcomes of COP6 and 7.
Sweden	YES	Expected in 2005 or later, linked to the introduction of the EU scheme. Emission Trading Bill expected to be ready in after December 2002.
Canada	MAYBE	There have been active discussions and study groups. National trading scheme still being considered in combination with, or as alternative to, targeted measures. A couple of voluntary trading pilots already exist.
The Netherlands	MAYBE	May be introduced if other measures fail, officially included in the "reserve" or "alternative" policy package.
France	MAYBE	The proposal was not adopted, although industry was interested and the government considered it.
Australia	MAYBE	Active discussions and consultations took place since 1999 led by Australian Greenhouse Office, but no conclusive decision on scheme design. Current government policy is to wait for ratification and entry in to force of Kyoto Protocol before introduction of a mandatory scheme.
New Zealand	MAYBE	Active discussions took place, but no conclusive decision; waiting for decision on Kyoto Protocol.
US	MAYBE	Depends on changes in domestic political climate; some support for domestic emission trading, nationally or regionally. Some local voluntary pilots already exist.
Slovakia	POSSIBLE	Domestic SO2 trading in place. Studies to explore feasibility of a CO2 domestic emissions trading ongoing. Ministry of Environment to lead consultations with other key stakeholders. Likely to be an international net seller of emission reduction credits.
Czech Republic	POSSIBLE	Initial studies to explore feasibility of a domestic emissions trading carried out recently. Ministry of Environment to lead consultations with other key stakeholders. Likely to be an international net seller of emission reduction credits.
Germany	POSSIBLE	Committed to VAs and taxes. Discussions on domestic trading started but face strong opposition from industry.
Japan	POSSIBLE/ UNLIKELY	Discussions and studies on domestic trading, but no conclusion. Voluntary actions by industry likely to play an important role.
Ireland	UNLIKELY	Some discussions and studies, but no conclusion.
Austria	UNLIKELY	Some discussions and studies; no conclusion.
Finland	NO	Taxes predominant. Likely to join directly in the EU trading scheme, or some other regional (Nordic or Baltic) as appropriate, but unlikely to develop a domestic scheme.
Belgium	NO	Taxes.
Switzerland	NO	VAs only.

The diversity among these countries in terms of their readiness to adopt domestic emissions trading results from a number of inter-related reasons, including the following elements in each country:

(i) GHG emission profile;

(ii) industry and market size as well as structures under which emission sources operate, and their relative importance in the overall GHG emission profile;

(iii) other existing policy tools such as energy or carbon taxes and voluntary agreements, and their performance;

(iv) presence of industry lobby and their influence (or lack thereof) in the country's policy process; and

(v) the country's response to the Kyoto process, and also to the proposed EU-wide emissions trading scheme.

Especially after the delay experienced at COP6-1 in the Hague in November 2000, many countries have taken the position of waiting for the final words on the proposed international emissions trading before making further progress on their domestic actions. On the other hand, some of the front-runners in introduction of domestic emissions trading schemes have the declared intention of possibly being able to influence the design of international schemes to come into effect in the future by going ahead earlier. Meanwhile, some in Europe (e.g. Finland) are not acting, as they are waiting for the EU framework to be finalised, so that their domestic schemes could be developed in line with the EU scheme, or simply join the EU scheme without introducing a domestic scheme.

In countries that have decided, or are considering introduction of domestic GHG emissions trading, it is being combined with existing energy and carbon taxes, voluntary agreements, and regulations, and it interfaces with these instruments in different ways. For example, the emissions trading scheme for electricity sector in Denmark has been designed to *supplement* pre-existing energy tax, which exempts this sector. On the other hand, the Norwegian scheme has been designed with the widest possible sector coverage, capturing almost 90% of emissions, is intended to *replace* the existing CO2 tax. Meanwhile, the proposed emissions trading scheme in the UK *integrates* energy taxes, voluntary agreements and emissions trading. Ideally, the overall policy packages adopted will bring about the intended level of emission reduction, with the least overall abatement and administrative costs. Also, the choice of the policy mixes could reflect underlying distributive intentions of the governments.

Table 2 summarises the status of domestic trading schemes, their coverage, and how they interface with other policy instruments aimed at GHG emission reduction in selected OECD member countries. Although it is not a domestic scheme, a brief description of the proposed EU-wide scheme is provided first for comparison reasons, as a few of the domestic schemes (notably that of Sweden) are closely linked to it in design. Then, it is followed by descriptions of these domestic trading schemes, organised roughly in terms of their readiness for implementation, and for countries that have decided not to adopt emission trading domestically, their reasons.

3.1 The Proposed EU-wide Emissions Trading Scheme

The European Community plans to introduce a community-wide GHG emissions trading scheme in 2005, ahead of the proposed launch date of the international emissions trading under the Kyoto Protocol in 2008. The EU published a Green Paper[10] in March 2000 on an emissions trading scheme at the EU level. It was meant to launch a discussion and consultation process among stakeholders on GHG emissions trading within the EU, and the relationship between emissions trading and other policies and measures to address climate change. Originally, more detailed rules and guidelines for the proposed EU scheme were to be published by mid-200l following COP-6 in November 2000. However, as no conclusive decision on many critical aspects of the international emission trading programme was reached at COP-6, the preparations for the EU-wide scheme has also been delayed. Following a final round of consultations during September 2001, the Commission

10. Commission of the European Communities, (2000), Green Paper on Greenhouse Gas Emissions Trading within the European Union [COM(2000)87].

published a proposal for a Directive on establishing a framework for an EU-wide GHG emissions trading in October 2001[11].

Table 2. Overview of Domestic GHG Emissions Trading Schemes

Country	Coverage	Initial permit allocation	Interface with other instruments
The EU	Initially CO2 only (from 2005), then eventually all Kyoto gases after 2008. Approx. 46 % of EU's estimated CO2 emissions in 2010 from 4,000-5,000 sites. Sectors include electricity and heat; iron and steel; refining, glass and building material, and pulp and paper. The chemical sector was initially considered but not included.	During 2005-7, free allocation (grandfathering) of allowances by Member states, which will be required to apply common criteria for their national allocation.	Sectors not covered by emissions trading is to be addressed by other parallel measures, including a number of directives on energy performance of buildings, energy efficient public procurement, etc. Synergies with the IPPC Directive intended.
Denmark	CO2 from electricity production only, about 30% of 1997 CO2 emissions.	Grandfathering	Trading covers electricity generation, supplementing the tax covering others.
UK	Emission Trading Scheme (ETS) on voluntary basis for any firms that commit to binding targets, with the choice of CO2 only or all Kyoto gases.	Free allocation of allowances. Direct participants bit their reduction commitments in an auction for incentive monies.	Firms that negotiate Climate Change Agreements (CCAs) qualify for 80% discount on Climate Change Levy and eligibility for baseline-and-credit trading. This is integrated into cap-and-trading by the direct participants in the ETS.
Norway	All Kyoto gases and all sectors possible; over 80% to be captured.	To be determined, partial auction, partial grandfathering.	In parallel with tax, eventually to replace it after 2005.
Sweden	From 2005, CO2 in fuel combustion. From 2008, all Kyoto gases and sectors, nearly 80% to be captured.	Auctioning	In parallel with tax, eventually to replace it (as proposed in a report dated April 2000 by an advisory group on domestic emission trading).
Canada	Not yet certain. Both narrower and "broad as practical" coverage considered.	Both free allocation and auctioning being considered.	Transitional voluntary credit-trading scheme (for reduction beyond what is required by regulation) considered prior to mandatory cap-and-trade scheme.
The Netherlands	Not yet certain.	N/A	Trading to supplement in case VAs, taxes and regulations are not sufficiently effective to achieve ½ of required Dutch reduction. The other ½ to be achieved through Kyoto flexible mechanisms.
France	Energy-intensive industry, including electricity generation that are exempt from energy taxes was to become part of emission trading.	N/A	Baseline-and-credit trading was to be built onto VAs on emission reduction, while granting exemption from the new energy taxes.

Overall Climate Change Strategy: The European Union is to meet its Kyoto commitment of 8% (or 336 million tons CO2 equivalents) reduction in GHG emissions from the 1990 levels during the

11. Commission of the European Communities, (2001); A Proposal for a Directive of the European Parliament and of the Council – establishing a framework for greenhouse gas emission trading within the European Community; [COM(2001)581].

First Commitment Period of 2008-2012. Although it was earlier anticipated that this would require an emission reduction of 14% compared to a business-as-usual forecast,[12] to date the EU emissions have decreased by 4% since 1990[13]. However, more recent projections suggest that policies implemented or in the pipeline (discussed below) would only lead to reductions that would merely stabilise the GHG emissions at the 1990 level, leaving a gap of 8% (about 340 million tonnes CO2 equivalents).[14] As the EU is to meet the target as a bubble, the member countries' contribution to overall emission reduction has been agreed to under the so-called burden-sharing agreement.

The European Climate Change Programme (ECCP) was established in June 2000 to help identify the most environmentally and cost effective measures enabling the EU to meet its Kyoto target. It has identified a mix of priority measures based on the relative readiness for implementation, including the emissions trading scheme and a number of new directives on energy performance of buildings, biofuels, energy efficient public procurement, and fluorinated gases. This priority policy package is estimated to achieve a reduction of about 240 million tonnes CO2 equivalents. To supplement these and meet the Kyoto target, additional measures are being considered, including directives on combined heat and power (CHP), energy services and minimum efficiency standards for electrical equipment, etc.

Coverage and participants: The emission trading scheme is intended to start only with CO2 in 2005, but eventually to include other Kyoto gases and sinks to the extent possible for easy and accurate monitoring. During the first phase (2005-2007), CO2 emissions from the following sectors have been proposed to be included in the trading scheme; (i) electricity and heat production, mineral oil refining and coke ovens, (ii) metal ore roasting and iron and steel production, (iii) cement, glass and ceramics production, and (iv) paper and pulp production. (See Table 3) These sectors[15] represent approximately 46 % of EU's estimated CO2 emissions in 2010, from 4,000 to 5,000 installations. The chemical sector was originally considered in the Green paper, but based on the consultation that took place later, it was not included in the October 2001 proposal for a Directive. Transport and household sectors are to be addressed through other policy instruments.

Allowances and their allocation: During the first phase of 2005-2007 prior to the start of the proposed international trading scheme, the allowances would be allocated by each Member state based on a common allocation method, i.e. for free of charge based on transparent grandfathering criteria. This is intended to avoid undue distortions in competition by having some Member states opting for free initial allocation while others opting for auctioning during the first phase when the price levels are relatively uncertain, and thus creating a European market for allowances. Also, Member states would be required to communicate in advance to the Commission of their proposed intentions in allocation of allowances in "national allocation plans", which could be rejected by the Commission if it is found to be not in line with the common criteria, including the Community requirements concerning State aid. A harmonised allowance allocation method applicable for the period from 2008 onwards will be determined in 2006 based on the review of the first phase (2005-2007) experience.

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12. "Business as Usual" scenario for CO_2, CH_4 and N_2O only, from UNFCCC/CP/1998/11/Add.2r, as in Hasselknippe and Hoibye (2001).

13. Commission of the European Communities, (2001), Communication from the Commission on the implementation of the first phase of the European Climate Change Programme [COM(2001)580 final].

14. As above.

15. These sectors have been proposed based on the Integrated Pollution Prevention and Control Directive.

Penalties, banking and other flexibility: The October 2001 proposal introduced a lower penalty at € 50 per excess tonne or twice the average market price, whichever is higher, in case of non-compliance during the first phase. It will go up to € 100 per excess tonne or twice the average market price. Banking of the allowances would be generally allowed within the first phase and the following 5-year commitment periods set under the Kyoto Protocol. Member states would be free to decide whether to allow the banking of allowances between periods. Linkages with some key non-EU member countries (e.g. Norway and Accesion Countries) would be considered, as well as compatibility with other Kyoto flexible mechanisms, i.e. JI and CDM. However, the latter may be considered at a later stage under a separate proposal; at this stage, credits from JI and CDM would not be included. Similarly, tradable renewable energy certificates already in existence in some countries would not be honoured for compliance with the obligations under the EU-wide emissions trading scheme.

Table 3. Sector Coverage under the Proposed EU Emissions Trading Scheme

Activities	Greenhouse gases
Energy activities	
Combustion installations with a rated thermal input exceeding 20 MW (excepting hazardous or municipal waste installations)	CO2
Mineral oil refineries	CO2
Coke ovens	CO2
Production and processing of ferrous metals	
Metal ore (including sulphide ore) roasting or sintering installations	CO2
Installations for the production of pig iron or steel (primary or secondary fusion) including continuous casting, with a capacity exceeding 2.5 tonnnes per hour.	CO2
Mineral industry	
Installations for the production of cement clinker in rotary kilns with a production capacity exceeding 500 tonnes per day or lime in rotary kilns with a production capacity exceeding 50 tonnes per day or in other furnaces with a production capacity exceeding 50 tonnes per day	CO2
Installations for the manufacture of glass including glass fibre with a melting capacity exceeding 20 tonnes per day	CO2
Installation for the manufacture of ceramic products by firing, in particular roofing tiles, bricks, refractory bricks, tiles, stoneware or porcelain, with a production capacity exceeding 75 tonnes per day, and/or with a kiln capacity exceeding $4m^3$ and with a setting density per kiln exceeding 300 kg/m^3	CO2
Other activities	
Industrial plants for the production of	
a) pulp from timber or other fibrous materials	CO2
b) paper and board with a production capacity exceeding 20 tonnes per day	CO2

Source: The European Commission, COM(2001)581

3.2 Denmark

Denmark has already introduced a domestic CO2 emissions trading scheme for the electricity sector as of January 2001. Three trades have taken place so far during the first year, as

reported in December 2001 (discussed further below). In May 1999, the Danish Parliament adopted the bill on CO2 quotas for electricity production (Bill No. 235) which set out the details of the scheme including annually decreasing caps for 2000-2003,[16] rules of initial permit allocation, etc. The scheme has been introduced in the context of the Electricity Reform.

Overall Climate Change Strategy and Objectives: Denmark has a Kyoto target of 21% reduction by 2008-2012 from 1990 levels, as adjusted under the EU burden-sharing agreement. However, the Danish government has set its own voluntary CO2 emission reduction target of 20% below the 1998 level by 2005 in two energy action plans, Energy 2000 and Energy 21.[17] The latter forecasts that existing policy mix of CO2 tax and emissions trading will enable Denmark to reduce emissions by 16.6%. Total Danish CO2 emission has fallen over the last decade, but emissions from the transport sector and electricity sector have increased. The latter is the key sector, as it has a very high share (33%) of the total GHG emissions of Denmark. A considerable progress has been made in the energy sector with respect to reducing CO2 emissions through energy savings, increased use of combined heat and power (CHP) and renewable energy, as well as fuel switching and increased efficiency of the power plants. However this effort was more or less offset by extra CO2 emissions during the 1990's caused by a significant increase in electricity export from Denmark, as discussed further below.

Complementary policy instruments - CO2 tax and voluntary agreements: Denmark has had CO2 tax on fuels since 1992 originally for households only, and in 1995 adopted the "Energy Package" with the main purposes of increasing the CO2 tax, and extending the energy taxes to energy sources used in trade and industry.[18] Fuel used for electricity production is exempted. Also, a related voluntary agreement scheme was introduced in 1996, whereby energy intensive firms have the choice between being subject to the CO2 tax or to negotiate a voluntary agreement on CO2 abatement and being partly exempted from the tax in the form of a tax rebate. The Energy Package was evaluated in 1998, and it was found that in fact it resulted in about 4% cut in CO2 emissions in 2005. However these schemes have not sufficiently addressed the emissions from the electricity generation sector, the major emitters but the only "unregulated" sector.

Emissions trading for the electricity sector: The rise in the emissions from the electricity sector derived from the fact that Denmark has become a net exporter of electricity in the period of 1994-97 in the Nordic electricity market, which is highly integrated.[19] The level of electricity generation in Denmark, which is mainly from coal, is closely tied to that of Norway and Sweden from hydropower. Therefore, the country introduced an emissions trading scheme for the period of 2001-2003, specifically targeted at the electricity sector, with special design features to accommodate its role in the Nordic market. Thus, the stated objectives of the Danish emissions trading scheme are: (i) to limit CO2 emissions from this sector, but also (ii) to ensure continued Danish participation in the Nordic electricity market, (iii) to give incentive to produce electricity by more environmentally sound technologies, and (iv) to prepare the energy sector for the implementation of Kyoto mechanisms.

Participants and initial quota allocation: With the coverage limited to the electricity generation sector, the trading scheme captures about 33% of Denmark's CO2 emissions. The Danish electricity sector consists of a few large producers and many small-scale producers. Thus, the

16. Due to the delayed EU approval in April 2000, the scheme could not become operational in 2000 as originally planned. It started as of January 2001. Pedersen, S. L. (2001b), *Danish CO2 Cap & Trade – Frequently Asked Questions*, Danish Energy Agency.

17. Pedersen, S. L. (2000).

18. Majocchi, A. (2000).

19. Pedersen (2000), Ellerman, Hoibye.

emissions trading scheme captures about 90% of emissions from the electricity sector, although participants include only about 15 major emission sources belonging to eight electricity producers, out of a total of about 500. The remaining small-scale producers are exempted from the scheme, on the basis of having CO2 emission levels less than 100,000 tons per year, but exclusively from CHP production. Given the small number of players, trades are to occur through bilateral agreements, as was the case for the recent trades in late 2001, to be reported to the Danish Energy Agency within four weeks after the trade. The participating electricity producers pay an administration fee of 0.079 DKK per ton of CO2 allowance allocated to them to the Danish Energy Agency, to cover the administration costs including those for monitoring, control, hearing and distribution of allowances, and development of the scheme. The scheme operates under the following annual caps, gradually decreasing over time: for 2001, 22 million tons; for 2002, 21 million tons; and for 2003, 20 million tons. (See Table 4) Post-2003 targets will be negotiated before the end of 2001. Under the annual cap for each year, initial permit allocation to the eight electricity producers was done through grandfathering, on the basis of the historical CO2 emissions during the period 1994-98 (during which the historical average annual emissions was 30.3 million tons, for comparison). The total cap for each year includes unallocated allowances for about 1.8 million tons CO2 to cover emissions from the small-scale producers exempted from the scheme. Also, new producers on the Danish electricity market will be allocated allowances when starting up production on the basis of objective and "non-discriminatory" criteria.

Table 4. Emission Allowances for the Danish Electricity Sector

Unit: million tons CO2	Final Allocation	Final Allocation	Preliminary Allocation
Producer	2001	2002	2003
Energi E2 A/S*	8.221	7.577	7.135
Elsam A/S**	10.533	9.873	9.420
E. ON/Preussen Elektra	0.965	0.838	0.751
I/S Avedorevaerket 2	0.094	0.527	0.510
Ostkraft Produktion A/S	0.062	0.060	0.058
Energi Randers Prod. A/S	0.198	0.198	0.198
Dansk Shell A/S	0.102	0.102	0.102
NRGI Amba (Anholt)	0.001	0.001	0.001
Without allowances	1.825	1.825	1.825
TOTAL CAP	22.000	21.000	20.000

Source: Danish CO2 Cap & Trade Update, 2001, Danish energy Agency.
* Energi E2 has sold certificates worth 0.05 million tons to E. ON, a German energy company in 2001
**Elsam has sold certificates worth 0.11 million tons E.ON, and Entergy, a US energy company in 2001.

Trades to date: In early December 2001, three trades were reported[20] to have taken place, after almost one year of no trades since the inception of the domestic trading scheme in January 2001. All three trades were between Danish certificate holders and foreign buyers. The two largest Danish certificate holders, Energi E2 and Elsam both sold 50,000 tons and 100,000 tons respectively to a Germany energy firm, E.ON. Elsam also sold certificates worth 10,000 tons to a US energy company, Entergy. Although the prices were not reported, it is believed to have been less than 40 DKK per ton, the current level set for the penalty in case of non-compliance, as discussed below.

Penalty, and banking and other flexibility: The scheme has a penalty structure intentionally designed to function as an escape valve, very low at 40 DKK (or US$4.78) per ton of CO2 for companies out of compliance. This is, as stated as one of the objectives of the scheme, to ensure

20. Dow Jones energy Service, "*Entergy First International Buyer of Danish Pollution*," 7 December 2001; and Reuters English News Service, "*Denmark seals its first CO2 pollution deals*," 6 December 2001.

Denmark's continued participation in the Nordic electricity market; should the demand for Danish electricity rises within the Nordic market, which varies a great deal depending on the level of rainfall in hydropower-dependent Norway and Sweden, Danish generators would continue to supply and export even if the cap is overshot. Thus, the Danish scheme is one with a "soft" cap[21], which would be observed only if the market-clearing price is less than 40 DKK. Other features of the scheme include banking, which is freely allowed. Use of credits from JI and CDM under the Kyoto flexibility mechanisms will eventually be allowed. However, this requires specific guidelines under the Act on CO2 Quota for Electricity Production, and such guidelines have not yet been issued.

3.3. The UK

A domestic voluntary trading scheme for GHG emissions, the UK Emissions Trading Scheme (ETS), was originally proposed by the industry-led UK Emissions Trading Group (represented by the Advisory Committee on Business and the Environment and the Confederation of British Industry) in its report submitted to the government in March 2000. The government then took over the initiative and developed a consultation document, a Greenhouse Gas Emissions Trading Scheme for the UK, published in November 2000. Responses to this consultation document were received from industry, energy sector, NGOs, and academia, and the government later published Draft Framework Document for the UK Emissions Trading Scheme in May 2001[22], with the revised start-up date of April 2002. The final Framework for the UK Emissions Trading Scheme was published in August 2001. The proposed design is quite complex, resulting from the need to maintain consistency with other elements of UK climate change policy, e.g. existence of two but partially linked emission trading markets; (i) baseline-and-credit trading made possible under the Climate Change Agreements, and (ii) cap-and-trading originally proposed by the industry-led UK Emissions Trading Group. In addition, companies could also participate in emissions trading by undertaking a "project" which reduced emissions from a source outside of the scheme and not covered by other regulatory limits.

Overall Climate Change Strategy and Objectives: The UK's Kyoto target, as per the EU burden-sharing agreement, is 12.5% reduction from the 1990's level. As of 2000, the UK has already achieved 13.5% reduction in GHG emissions compared to the 1990 baseline (212 million tonnes). This is partly due to the decline of the coal industry and the switch to natural gas experienced by the country. The UK government has set out its own national target beyond the Kyoto target; CO2 emission cut of 20% below the 1990 levels by 2010. The UK Climate Change Programme[23] outlines a policy package, including emissions trading, energy tax, voluntary agreements and technology standard regulation, intended to deliver GHG emission reduction of 23% below the 1990 levels by 2010. Climate Change Levy, Climate Change Agreements, and emissions trading discussed below are expected to deliver respectively about 7 million, 9 million, and 7.7 million tonnes of CO2 emissions reduction.

Complementary policy instruments –Climate Change Levy and Climate Change Agreements: Fossil fuel levy has been in place since 1990, paid by electricity consumers to fund renewable energy projects. A new energy tax, *Climate Change Levy (CCL*) has been introduced as of April 2001 for energy consumption by industry and business. The CCL is a flat rate levy of 0.43p/kWh on electricity,

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21. Ellerman (2000).

22. Department of Environment, Transport and the Regions (2001a), Draft Framework Document for the UK Emissions Trading Scheme, May 2001 [UKETS901)01].

23. Department of Environment Transport and the Regions (2000), *Climate Change: The UK Programme* [SE200/209].

0.15p/kWh (equivalent) on coal and 0.15p/kWh equivalent on gas[24]. For electricity, every unit generated is taxed at the flat rate irrespectively of the fuel used to generate it, while electricity from renewable sources and CHP is exempted. Fuels used for electricity generation and by the transport sector are exempted. The household sector is also exempted and energy-intensive industrial sectors may be eligible for a discount of 80% on the CCL if they enter into *Climate Change Agreements (CCAs)*, voluntary agreements for industrial sectors for committing to certain "challenging" emission reduction targets. Ultimately, there will be about 40 "umbrella agreements"[25] with sector trade associations and bodies, consisting of ten large energy intensive sector and over 30 smaller sectors. Only those that are under Integrated Pollution Prevention and Control Act of 1999[26], which sets certain energy efficiency performance, are eligible to negotiate CCAs. Under this regulation, firms are required to adopt Best Available Techniques (BAT) sector by sector gradually between 2001-2007. The BAT will include considerations on the energy efficiency of the processes.

Under the CCAs, firms can choose either absolute or relative targets: (i) an absolute target in terms of energy or CO2 emissions, or (ii) a unit or relative target, again either in terms of energy or CO2 emission per unit of output. In practice, most firms have chosen the relative targets. If a firm over-achieves its baseline target, it will be credited for the portion above and beyond the baseline which had been agreed, and these credits can be traded. Those that opt for absolute targets under CCAs are also eligible to participate in Emission Trading Scheme (ETS) discussed below. Those with relative targets can trade credits among themselves, as well as with those with absolute targets but subject to the "gateway" rule described below.

The ETS - participants and initial permit allocation: The ETS is proposed to be a voluntary programme, with two possible ways of entry for participants; (i) largely those outside the scope of CCAs[27], and on voluntary basis ("direct participants") who are allocated permits for free through a "sellers' auction" whereby they bid in their promised emissions reductions (described further below), and (ii) those through the "agreement route", i.e. with absolute targets under CCAs discussed above. Although the rules have yet to be finalised, companies could also enter the scheme through undertaking projects which reduce emissions from sources not covered by any other regulatory limits[28]. In theory, the ETS is open to any entity wishing to participate and is able to comply with the rules of the scheme concerning monitoring and reporting. A participating firm will have choice of either CO2 only, or all Kyoto gases.

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24. In order to make the CCL financially neutral for companies, a reduction in National Insurance contributions by 0.3 % was introduced so as to maintain competitiveness of the companies involved. Thus, the introduction of CCL is revenue-neutral for the government.

25. See http://www.defra.gov.uk/environment/ccl/agreements.htm for a list of ten-year "umbrella agreements" on energy and carbon efficiency, which contain lists of facilities which will be subject to the 80% discount, sector targets, and conditions which apply to participating firms.

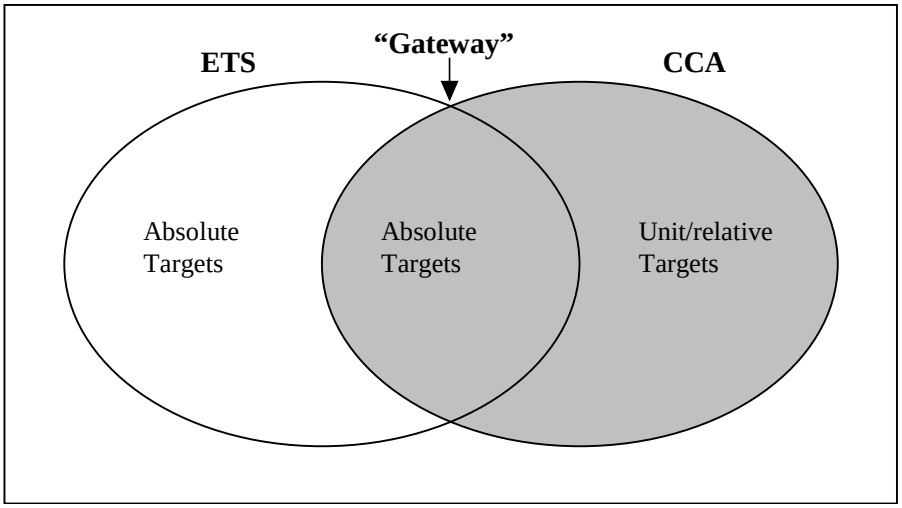
26. EU's Integrated Pollution Prevention and Control Directive (96/61/EC), IPPC, is being implemented in to UK law by regulations made under the Pollution Prevention and Control Act of 1999.

27. However, the CCA participants can still enter the auction for emission allowances under the ETS, as long as they bid in for emission reductions which are not already covered by the CCL or any other agreements.

28. Moreover, companies may also enter the scheme simply by opening a trading account and buying and selling allowances, without having an emissions reduction target themselves.

As the ETS is a voluntary scheme and its participants would also have to pay Climate Change Levy on their energy consumption, it was anticipated that not many would actually volunteer to participate. Therefore, in July 2000, the government has committed an incentive fund up to 30 million pounds sterling per year (after tax) for a five-year period to attract participants into the ETS. The incentive funds will be distributed in a "sellers' auction" scheduled to take place in February 2002, whereby companies sell their reductions against their baselines to the government for the entire period of 2002-2006, divided into five equal annual targets. Those that are already covered under CCAs and therefore eligible to receive the 80% discount on Climate Change Levy would not be able to receive the incentive fund in addition for the same emissions.[29] The first pay-out of the incentive fund is scheduled as follows: firms meet the reduction target for calendar year 2002, which will be verified during January-March 2003, and the incentive fund will be paid out in April 2003 for the first year.

Figure 1. Relationship between ETS and Climate Change Agreements



Source: Salmons, R., 2000.

It appears that the payment of the incentive funds is contrary to the Polluter Pays Principle, as the emitters are being paid by the government to participate in the ETS. However, they are at the same time paying the CCL. Until the outcome of the auction is available, it cannot be determined the relative magnitudes of the CCL paid by industry and the incentive funds paid to industry under this scheme, as well as the proposed levels of emission abatement[30]. It would be of great interest to undertake a distributive analysis of the UK scheme once it has been in full operation for a while.

29. That is, if a company has a portion of its emissions not covered under CCA could bid in reduction of remaining emissions in the auction.

30. Salmons, R. (2000). CSERGE has made rough estimates on the potential level of abatement under these auctioned subsidy. Those covered by Climate Change Levy but not eligible for CCA are estimated to be responsible for around 40-45 MtC in 2000. If half of these emissions are captured by voluntary participation in the ETS, and if they offer 20% reduction in their emissions by 2010, then the total reduction in emissions over the 2002-2010 period would be about 16 MtC. On this basis, the government would pay about 10 pounds sterling per ton of carbon for the auctioned incentive fund.

The "gateway": There will be a "gateway" linking the market for the "unit or relative sector" for those with a unit target under the CCAs, and the "absolute sector" under the ETS. There is no limit on sale of absolute sector permits into the unit/relative sector, as these permits are solidly based upon absolute emission reductions. However, the sale of compliance credits under CCA (relative sector) into the ETS (absolute sector) is restricted, and allowed only as long as the total historical sale in this direction does not exceed the quantity of absolute sector permits previously sold into the relative sector. This is aimed at ensuring that there will be no net transfer of allowances from the relative to the absolute sector, as otherwise this could undermine total emission reduction achieved in absolute terms.

Penalty, banking and other flexibility: When a firm under a CCA is found to be out of compliance, i.e. did not meet the agreed target and the portion of emission over the target is not supported by additional credits purchased, will be subject to penalty. A firm subject to penalty for non-compliance will lose its eligibility for the 80% discount on the CCL in the next period, and will also be required to repay the discount it received in the previous period if they are out of compliance at the end of their CCA period. For those that are in the "absolute sector", the penalty for non-compliance would include the following: non-payment of the incentive fund; "naming and shaming" and expulsion from the scheme; "penalty docking" of the next years' allowances (i.e. for every 1 allowance a firm is short this compliance year, 1.3 allowances will be docked from the next year's allocation); and possible legal actions as criminal offences. Recovery of the incentive money paid out in the previous period could also occur if a firm is not in compliance at the end of the five year period of the scheme or leaves the scheme before the end of that period. There will be no restriction on banking up to the end of 2007, and some baking is also allowed into the First Commitment Period (2008-2012) under the international emission trading scheme. The UK ETS is envisioned to recognise credits from JI and CDM, once clear rules regarding these have been established, as well as domestic offset projects as mentioned earlier.

3.4 Norway

The proposed Norwegian domestic GHG emissions trading scheme is the most extensive one in terms of coverage, capturing more than 80% of Norway's 1997 GHG emissions. In late 1998, the Parliament established the Quota Commission with a mandate to consider a domestic GHG emissions quota trading scheme in accordance with existing green taxes and the White Paper No 29 on Norwegian implementation of the Kyoto Protocol. The proposed domestic tradable quota system is intended to replace other tools previously used in domestic climate change policy, predominantly the CO2 tax. The Quota Commission proposed an initial design of a cap-and-trade based scheme in a report released in December 1999. In June 2001, the Norwegian government issued another White Paper on climate change policy, including a proposal for domestic emissions trading in Parliament. It could then be legislated one year later in 2002. The start-up of the quota scheme is not expected before 2008, when the international trading is to be launched, but it would all depend on international developments, including that of the proposed EU scheme.

Overall climate change strategy and objectives: Norway is allowed a 1% increase in GHG emissions from the 1990 levels (55 million tons CO2 equivalents) under the Kyoto Protocol. This translates to a 17.4% reduction from the business-as-usual scenario, or about 12 million tons Co2 equivalents[31]. However, Norway's GHG emissions already grew by nearly 10% between 1990 and 1999. More recent official projections show, however, that Norway's total GHG emissions could

31. This assumes that no new measures are introduced, and two planned gas-fired power plants are built adding 2.1 million tons CO2

increase by as much as 26% between 1990 and 2010[32], posing a major challenge in meeting its Kyoto commitments. Given the relatively high marginal abatement cost of domestic GHG emissions reduction in Norway, it would fully take advantage of cheaper emission reduction opportunities abroad afforded by the Kyoto Flexible Mechanisms. Nevertheless, the White Paper proposes the following domestic measures[33]; (i) continue to depend on CO2 tax until it is replaced by a quota system; (ii) invite the industry to co-operate on emissions reductions through voluntary agreements; (iii) initiate an agreement on reductions of non-industrial emissions of the potent greenhouse gas SF_6, and (iv) introduce measures to decrease the emissions of the greenhouse gases HFC and PFC. Another important emphasis is to increase the share of renewables for stationary energy use. In Norway, renewable energy sources already supply 70 % of stationary energy use[34], with hydro-power accounting for almost all electricity production during the 1990's.

Complementary policy instruments – taxes and other instruments: Norway's domestic climate change policy has predominantly relied on the CO2 tax. Norway introduced CO2 tax on fossil fuels in 1991, which has been extended in coverage over the years. Under the current system, about 65% of CO2 emissions in Norway are subject to this tax, estimated to achieve emission reduction of about 0.55 million tonne by 2010[35]. However, certain industrial sectors are still exempted, while some sectors such as petroleum and private transport have adapted to relatively high CO2 taxes, resulting in large differences in GHG emissions reduction in different sectors.[36] In order to improve cost-efficiency, the Government widened the tax base by introducing from January 1999 a tax of NOK 100 (about US$ 11) per tonne CO2 emissions in sectors that were previously exempt, including diesel and mineral oils used in air traffic, domestic shipping and supply ships (in 2001 at NOK 154-182/tonne CO2), and offshore petroleum installations (roughly NOK 300/tonne CO2 in 2001). Tax rates are quite volatile, and could be modified every six months during political discussions on the State Budget, limiting the desired impacts of the CO2 tax on long-term planning for energy efficiency investments.[37] Also, major energy-intensive industrial sectors as well as gas used in the transport sector still remain exempted from the CO2 tax, undermining its environmental effectiveness. The CO2 taxes are scheduled to be phased out before 2008 or possibly earlier, to eventually be replaced by emissions trading.

In addition, a tax on the final disposal of waste, in order to increase energy recovery and reduce emissions of methane from landfills, has been introduced in January 1999. This tax on waste disposal is expected to achieve emission reduction of 1- 1.5 million tons by 2010 (2-3 times the expected reduction from CO2 tax). The tax rates adopted are the following: (i) landfills, NOK 300 per tonne of waste; (ii) incinerators, a) basic charge of NOK 75 per tonne of waste, and b) additional charge depending on the degree of energy recovery at NOK 0-225 per tonne of waste. Also, new licensing requirements under Pollution Control Act for large point sources, as well as limited duration voluntary agreements with certain sectors including aluminium industry are also considered, and expected to contribute to significant emission reductions.

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32. OECD (2001e).

33. Ministry of Environment, White Paper to the Norwegian parliament, No. 29 (1997-98) on Norwegian implementation of the Kyoto Protocol.

34. Ministry of Environment, and Schreiner, P. (1999) Obstacles to the Implementation of Tradable Permits: The Case of Norway.

35. Ministry of Finance, Norway, *Proposition No. 54 (1997-98) to the Parliament on Green Taxes,* at the rates for 1999.

36. Ministry of Environment, Norway (1999), *A Quota System for Greenhouse Gasses – A policy instrument for fulfilling Norway's Emission reduction under the Kyoto Protocol.*

37. OECD (2001e).

The emission trading scheme: In light of the performance of existing policy instruments, the principle guidelines for the Norwegian emissions trading system put forward by the Parliament include the following: (i) at a minimum apply to branches of industry that has been exempted from CO2 taxes (while emissions that are subject to regulation through emissions trading shall not be subject to taxation); (ii) emissions that are subject to regulation through emissions trading shall not be regulated by the Pollution Control Act; (iii) apply to all GHGs listed in the Kyoto Protocol, (iv) attempt to achieve about 30% reduction in emissions from 1990 levels, (v) not penalise enterprises that have reduced emissions early, and (vi) allocate permits on a long-term basis, allowing banking.

Coverage and participants: The scheme is intended to cover all emissions sources and Kyoto gases technically possible to measure and verify emissions[38]. This would capture more than 80% of Norway's GHG emissions in 1997. Participating entities will include the metallurgic industry, gas-fired power plants (if the two planned facilities are built), the production of cement and composite building materials, the petrochemical industry, oil refineries, and transport and processing installations for crude oil. Inclusion of mobile as well as domestic sources is also being considered.

Initial quota allocation: The most controversial part of the discussion on the Norwegian trading scheme has been the question on initial permit allocation, through auctioning or grandfathering. The Commission was evenly split between the two, with a bare majority recommending auctioning and the proceeds accrue to the government to reduce other taxes and to achieve a double dividend by reducing tax on labour.[39] Also, the issue of international competitiveness of Norway's energy-intensive industry was a focus of debate, with a possible outcome that these firms would be initially granted access to permits for free. Another element for consideration has been that free quota allocation maybe out of line with the EU requirements limiting state aid.

Banking and other flexibility: The Norwegian scheme is intended for maximum flexibility including banking and recognition of the flexible mechanisms proposed under Kyoto, i.e. credits from JI and CDM as well as AAUs from other countries through international trading. As mentioned earlier, Norway's relatively high marginal abatement cost puts it in a position to opt for cheaper emission reduction opportunities outside its boarders. Thus, the proposed domestic quota trading system would benefit from linkages with, JI, CDM and international trading.

3.5 Sweden

An advisory committee to the Swedish Ministry of Industry, Employment and Communications on domestic emissions trading issued a report in April 2000, which opened active discussions on a proposed domestic emission trading scheme, but decisions are yet to be taken on the design details of the scheme. Then, in July 2001, the government appointed a parliamentary delegation to further develop a proposal for a Swedish domestic GHG emission trading system. It is expected to submit its recommendations in December 2002. A bill instituting the recommended scheme could be passed some time after that. However, implementation would most likely not be before that of the EU-wide scheme in 2005. Sweden is allowed a 4% increase from the 1990 level under the Kyoto commitment and the EU burden-sharing agreement. This translates to a reduction of 8.3%, or 6.1 million tons CO2 equivalents, compared to business-as-usual scenario.

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38. This would exclude N2O and CH4 from combustion, CO2 from agricultural liming and from solvents, CH4 and N2O from agriculture, FHCs/PFCs used as substitutes for CHCs and halons, and SF6 other than form magnesium production.

39. Ministry of Environment (1999), Ellerman (2000).

Complementary policy instruments – the CO2 tax: The CO2 tax has been the main instrument in the country's climate policy. A bill was passed in 1990 introducing a carbon tax and value-added tax on energy, and lowered the existing energy tax as part of an overall fiscal reform[40]. The original tax was at SKR 250 per tonne of CO2, levied on oil, coal, natural gas, LPG, and gasoline, while fuel use for electricity production was exempted. Since then, the carbon tax rate has been increased, while the energy tax and other taxes on industry were lowered. As part of a larger green tax reform, which is to continue during the period 2001-2010, the CO2 tax was increased by 25 % in January 2001. This increase was designed so as to keep the tax burden for industry and transport sectors to remain unchanged. In the 2002 Budget Bill, the Parliament has decided that the CO2 tax will be raised further by 15 % in 2002. However, the mining, manufacturing, agriculture, forestry and aquaculture sectors will be exempt from this increase, while the effect on petrol and diesel will be neutralised by matching cuts in the energy tax[41].

An evaluation of the CO2 tax by the Swedish Environment Protection Agency in 1995 concluded that it has helped to reduce CO2 emissions: CO2 emissions from the district heating, industrial and housing sectors were 19% lower in 1994 compared to the 1987 levels. Another study showed that CO2 tax helped reduce CO2 emissions in the order of 3-4 % for the period 1991-93[42].

One bottleneck in the discussion on the proposed emissions trading scheme is that the revenue from permit auctioning is expected to provide less than that from the current CO2 tax, which has been proposed to be phased out and completely replaced by trading in 2005. However, it is still too early to conclude that this will be the case. This potential revenue impact of the transition to domestic emissions trading currently faces the reluctance of Ministry of Finance, as the present CO2 tax revenue is higher at about Euro 2 billion, or almost 2 % of the national tax revenue,[43] than the expected revenue form auctioning of permits. The harmonisation of this gap is one of the important tasks for the parliamentary delegation.

Domestic emissions trading – coverage, participants and initial permit allocation: Although the Swedish system is proposed to be integrated into the anticipated EU scheme, there are some notable differences[44]. As currently proposed, the Swedish emissions trading scheme would have a wider coverage of about 80% of 1998 domestic GHG emissions, including both the transport and domestic sectors in addition to industry and electricity sectors covered under the EU scheme[45]. The transport sector is proposed to be captured through upstream allocation of quotas to fuel importers and distributors. It is recommended that industries not currently taxed for CO2 emissions should be left out of the domestic trading scheme to ensure their competitiveness, which is in contradiction to the EU-wide scheme proposed. Quotas are to be auctioned off according to the Polluter Pays Principle, whereby the highest bidder will receive the permits. However, such an auction might contradict existing Swedish law, which states that a firm which has been given the right to emit in the

40. OECD (2001b), Environmentally Related Taxation in OECD Countries: Issues and Strategies.

41. As per information provided by the Swedish Ministry of Finance in December 2001.

42. Branzini A, Goldemberg J, Speck C (2000) A future for carbon taxes. *Ecological Economics*: 32 (3) 395-412 (2000), as quoted in Rousaki, K. (2001), Market Mechanisms for Greenhouse Gas Emissions Reduction.

43. "Swedenvironment" No5, November 2001, Ministry of Environment and Swedish environmental Protection Agency.

44. Information on the Swedish emission trading scheme is from Hasselknippe, H. and Hoibye, G. (2001).

45. If the Swedish scheme were to include only the sectors proposed for the EU scheme, this would translate instead to coverage of only about 30 % of Swedish CO2 emissions.

past has legal ownership over its emissions, and the Swedish state cannot auction out these emissions without financially compensating the firm. This will need to be reconciled by the current inquiry by the parliamentary delegation and in the final Bill, most likely opting for grandfathering for some sectors.

3.6 Canada

Overall climate change strategy: Canada's Kyoto targets for the first Kyoto commitment period is a 6% reduction in their GHG emissions from the 1990 level. This target is 26% below recent business-as-usual projections for 2010, a gap that would have to be met by some combination of domestic emissions reductions, domestic sinks enhancements, and net use of the Kyoto mechanisms. Domestic emissions trading (DET) is being considered as part of a larger package of measures that could help Canada achieve its target if the Kyoto Protocol is ratified.[46]

Earlier considerations on a domestic emissions trading scheme: To date, exploration of GHG emissions trading approaches in Canada has consisted of analyses, voluntary pilot programmes implemented as public-private partnerships, and private sector trades. Between 1998 and 2000, as part of Canada's National climate Change Process (NCCP),[47] two multi-stakeholder committees were convened to explore issues related to domestic emissions trading: the Tradeable Permits Working Group (TPWG) and the Credits for Early Action Table. The TPWG explored the potential contribution that a domestic emissions trading scheme could make towards achieving Canada's emission reduction target. In a 2000 Options Report,[48] the TPWG focused on the analysis of two design options: (i) a downstream design - targeting large final emitters (electricity generation, industry, and possibly large transportation), capturing approximately 400 to 500 firms representing about 35% of Canada's total GHG emissions; and (ii) a hybrid design with a major upstream or midstream component – aiming for broadly based coverage (emissions from fossil fuel combustion mainly captured "upstream," plus non-combustion CO2 and most other final emitter GHGs captured under the proceeding option i), which would include about 100 to 500 entities and cover about 75% of total GHG emissions.

The TPWG suggested that permits could be allocated by a combination of auctioning and gratis allocation. The revenue from auctioning would be recycled to address potential equity concerns (e.g. through reductions in personal and corporate income taxes). A portion of permits could be allocated free of charge and on a continuous basis to compensate sectors that directly compete with industries in countries without GHG reduction commitments. Firms with comparatively more emissions-intensive capital than competitive alternative technologies would also be given permits at no cost, but on a transitional basis.

Further work on domestic emissions trading: With support form the Analysis and Modelling Group (AMG),[49] two new inter-governmental working groups are currently undertaking further

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46. This section on Canada draws on Haites, E. & Mullins, F (2001), *Linking Domestic Industry GHG Emission Trading Systems*; and National Round Table on the Environment and the Economy (2001), *Canada- Progress on GHG Emissions Trading*.

47. An inter-governmental/stakeholder process established to advise on Canada's implementation strategy on climate change. (See http://www.nccp.ca)

48. Tradeable Permits Working Group, April 2000. Using Tradeable Emissions Permits to Achieve Domestic Greenhouse Gas Objectives: Options Report. (http://nccp.ca/html/tables/pdf/options/TPWG_en_bw.pdf)

49. The AMG is an inter-governmental group established as part of the NCCP. When it was first established, the AMG's main objective was evaluating the overall economic and

analyses on issues related to domestic emissions trading: the Domestic Emissions Trading Working Group (DETWG) and the Emissions Allocation and Burden Sharing Working Group (EABSWG).[50] The working groups are to report in early 2002. The EABSWG is considering how the burden to reduce emissions could be allocated by regions or sectors. The DETWG is considering aspects of a mandatory cap-and-trade DET system including: (i) *sectoral coverage*; options focus on narrower coverage (electricity generation and large industries), and on "Broad as Practical" coverage - a hybrid involving fuel distributors to capture combustion CO2, and other GHG emissions at the final emitter level; (ii) *allocation of permits*; options include alternative forms of gratis allocation and auctioning of permits (with revenue recycling); (iii)use of *credits created through voluntary emission reductions* or removals in uncovered sectors; (iv) *transition to a mandatory DET system*, including possible use of pilots, interim voluntary trading regimes targeted at sectors that will be subject to mandatory coverage; (v) *link between DET and international emissions trading*, assuming that the Kyoto Protocol will enter into force; (vi) *roles of the federal, provincial and territorial governments* in the implementation of a DET system; and (vii) *competitiveness impacts of a DET system,* related to sectoral and regional coverage and allocation design elements.

Earlier pilot trading programmes and a new trading initiative in Ontario: Two private-public emissions credit trading pilot programmes have been implemented in Canada. The Ontario Pilot Emissions Reduction Trading (PERT) Project,[51] a voluntary industry-led multi-stakeholder initiative, was established in 1996 and applied to GHGs and to other air pollutants emitted in the Windsor-Québec City corridor. Members received credits and recognition from the Ontario Ministry of Environment for emissions reduced over and above what was required by regulation. The GHG Emission Reduction Trading (GERT) Pilot[52] was initiated in 1998 as a voluntary joint initiative between the federal government, provinces, industry, labour, and environmental groups. It reviews emissions reduction projects to ensure that credits generated for trades reflect surplus (over and above what is required by law), measurable, and verifiable emissions reductions. GERT received projects for evaluation until December 2001, and will complete all project reviews and technical reports by June 2002.

In October 2001, the government of Ontario announced Canada's first significant regulated emissions trading system. It started as of January 2002, and is initially limited to electricity generators in the province. While it currently covers only emissions of pollutants responsible for acid rain and smog, the government has suggested that the system could in the future extend to other industrial sectors and pollutants including GHGs.[53]

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environmental implications of Canada complying with its Kyoto commitments. However, the National Air Issues Co-ordinating Committee (an inter-governmental group that manages the NCCP) has supported a continuing role for the AMG, such as analyses of issues related to Kyoto ratification. (See http://www.nccp.ca/NCCP/pdf/media/AMG_Wkpln.pdf)

50. http://www.nccp.ca/NCCP/whatsnew/index_e.html

51. http://www.pert.org/pert.html

52. http://www.gert.org/

53. Ontario, Ministry of the Environment, "Ontario's Clean Air Plan for Industry," 24 October 2001. http://www.ene.gov.on.ca/envision/coalgen/clean_air.htm

3.7 The Netherlands

Overall climate change strategy and policy instruments: The Dutch response to its Kyoto target of 6% reduction from 1990 levels, which corresponds to a 21% reduction, or about 50 million tons CO2 equivalents, from the business-as-usual scenario, mainly consists of regulations, voluntary agreements and taxes. The government plans[54] to meet half of the reduction target through domestic actions, while the other half will be procured through the Kyoto flexible mechanisms. Through its Emission Reduction Units Procurement Tender (ERUPT), the Netherlands was the first country to call for tender for Emissions Reduction Unit originating in Central and Eastern Europe, making use of the JI mechanism. ERUPT is now complemented by Certified Emissions Reduction Units Procurement Tender (CERUPT), which allows industry to sell credit through CDM projects in developing countries to the government.

The half of reduction target to be met domestically will be though the "Basic Package," which mainly consists of (i) voluntary agreements on energy conservation, a "benchmark covenant for energy efficiency" for a period of 2000-2012, whereby less energy intensive firms are asked to undertake long-term energy conservation measures for a certain impact, i.e. 15% of internal rate of return, (ii) increase in the energy taxes that were introduced 1995, and (iii) renewable energy targets. The Basic Package is hoped to cut emissions by 25 million tons CO2 equivalent.

These measures in the Basic Package will be evaluated in 2002. If it is decided that these measures are not sufficient, then the "Reserve Package" will be introduced. The Reserve Package is expected to include a further raise in energy tax, a raise in the duty on motor fuels, and underground storage of CO2. Domestic emissions trading is included in the so-called "Innovation Package" aimed at developing new technology and policy tools. A National Committee for studying a potential domestic emissions trading scheme was established in August 2000. The Committee is expected to report on a possible domestic cap-and-trade scheme in October 2001. The Dutch industry sees their negotiated agreements on energy efficiency as their primary responsibility, while also open to the concept of emissions trading if it is combined in a sensible way. There is a view among the industry that these "voluntary" agreements are not so voluntary in practice. Also, the increasing interest in domestic emission trading among the Dutch industry may lead to a possible outcome of a domestic emissions trading system that builds onto the voluntary agreements already in place. One likely scenario for domestic emissions trading in the Netherlands would be a mixed system based on VAs for the "exposed" sectors (i.e. exposed to international trade), while the "sheltered" sector would be required to participate on a mandatory basis.

3.8 France

Overall climate change strategy and policy instruments: The Kyoto target that France needs to meet is stabilisation of GHG emissions at 1990 levels for 2008-2012. This translates into a reduction of 12 % from the business-as-usual scenario, where increases are expected in transport, residential and service sectors. The heart of the climate change policy package of France was a new energy tax, a general tax on polluting activities (TGAP), announced in January 2000, covering industry, construction and other sectors. It was to be launched as of January 2001, but it was declared unconstitutional in December 2000, and thus was not adopted. What was proposed was a tax at the rate of 23-30 Euro per ton of CO2 equivalent. It was to be raised towards the Kyoto period to around 76 Euro per tonne by 2010.

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54. Ministry of Housing, Spatial Planning and the Environment (1999), *The Netherlands' Climate Change Policy Implementation Plan – Part 1, Measures in the Netherlands.*

Prior to this, France had already adopted voluntary approaches aimed at curbing GHG emissions from industry.[55] The largest company in the French aluminium industry, responsible for 70% of primary and secondary aluminium production in the country, signed an agreement on GHG emissions in 1996 with the Ministry of the Environment. Under this agreement, the firm commits to reduce its energy consumption per unit of output by 10% in terms of CO2 emissions by the year 2000 from the1990 level. Four further negotiated agreements similar to this have been signed with the glass, the plaster, the cement and the steel industries. They set collective targets and are signed with the respective branch organisations. Whether the targets under these agreements have been met or not is uncertain, but should be investigated.

Voluntary negotiated agreements (VNAs) built into the new proposed tax scheme, TGAP[56] were considered. Under the VNA, energy-intensive industries, especially those that use energy as a raw material and for electricity generation, were to be exempted from TGAP in exchange for committing to certain emissions reductions. To complement these new proposed measures for addressing GHG emissions, an industry group, Enterprises pour l'Environnement, proposed a baseline-and-credit trading scheme built into the proposed TGAP and VNA scheme, which was widely supported by the government. The industries exempted from the tax through the VNAs would agree to either unit or absolute targets for emission reduction. When a firm over-achieves the negotiated target under the VNA, it would receive emission reduction credits, which could be kept or sold on the market. If a firm does not meet the target, then it would have to buy additional emission credits to cover the excess emissions. Conceptually, the element of baseline-and-credit trading built upon negotiated agreements is very similar to the credit trading in the "relative sector" through the Climate Change Levy Agreements in the UK. To qualify for the VNA, a firm would have to be energy-intensive, and exempted from any current or previous energy tax schemes. It was expected that the VNA scheme would cover about 80% of the industrial sources of GHG emissions.

This proposed domestic emission credit trading programme was to be integrated into the UE scheme in 2005, then in 2008 with the international emission trading scheme as well as other Kyoto flexible mechanisms. It is not certain at this time if the TGAP would be considered again or the proposal on emission credit trading would be taken up in the future.

3.9 Meeting the GHG reduction target without domestic emissions trading

As many countries already have in place regulations, energy taxes and voluntary agreements for addressing domestic GHG emissions reduction, it can be difficult, or unnecessary, to introduce domestic emissions trading if the existing policy instruments are deemed to be sufficiently effective. This is the classic situation typically faced by a new instrument. As of today, it seems that the following OECD member countries have opted for not introducing domestic emissions trading in the existing policy packages or have not yet reached final decisions, although the situation may change: Germany, Belgium, and Switzerland.

Germany: Voluntary agreements and taxes have played an important role in Germany's climate change policy. Its Kyoto target is a 21% reduction from 1990 levels by 2008-2012, corresponding to 20.5% reduction from the business-as-usual scenario. Due to the economic slowdown experienced after the unification with former East Germany the growth in GHG emissions

55. OECD (1999b), *Voluntary Approaches for Environmental Policy: An Assessment*, OECD, Paris.

56. Information on the French trading scheme is from Hasselknippe, H. and Hoibye, G. (2001), Ellerman (1999) and Concerted Action on Tradable Emission Permits (CATEP).

has been well within the Kyoto target;[57] in fact, the total German CO2 emissions fell by about 13% between 1990 and 1998. Therefore, the government has set its own target of 25% reduction in CO2 emissions by 2005 from 1990 levels.

In 1995, the Federal Association of German Industry declared a statement to voluntarily reduce their CO2 emissions or their energy consumption by 20% by 2005 compared to 1987 levels.[58] This involved a publication of the "Declaration by Germany Industry and Trade on Global Warming Prevention," supplemented by establishment of an independent monitoring system. By October 1999, five major German trade and industry associations, and 14 processing industry associations had accepted this voluntary commitment. In response, the federal government decided not to introduce further legislative measures aimed at GHG emission reduction. As a result of these as well as the structural changes experienced by the German economy as mentioned earlier, the German industry over-achieved its target, by reducing its CO2 emissions by 27% between 1990 to 1998. However, the extent to which the VAs contributed to the achievement of the environmental goal cannot be determined. In addition, in April 1999, Germany implemented its ecological tax reform package, with gradual increases over time up to 2003 in tax rates for fossil fuels and electricity tax rates. This package is expected to not only meet the national GHG emission reduction target, but also promote renewable energy, while at the same time, easing tax burden on labour.[59]

A discussion group on the possibility of a domestic GHG emissions trading in Germany supervised by the German Ministry of Environment has been set up. A special focus is laid on the compatibility of emission trading and existing voluntary agreements between the industry and the government. A position paper outlining potential design elements of a German domestic emissions scheme was delivered to a consultation group organised by the European commission in autumn of 2001. However, it is impossible to predict whether there will be a majority in favour of domestic emission trading within Germany, in light of the current strong industry opposition. The latter is also opposed to the proposed EU-wide emission trading, contending that VAs and other measures in place should be sufficient. Moreover, the current national GHG emission reduction targets are also facing opposition from the Economics Ministry.

Belgium: The Kyoto commitment for Belgium is a reduction of 7.5% from 1990 levels by 2008-2012, while, to date there has been an increase of 16% since 1990. The country's National Climate Plan relies on CO2 taxes at the national level, and voluntary agreements at regional level to achieve identical (i.e. absolute) emission reductions. The tax levels have been set to meet half of Belgium's Kyoto target, as the other half is expected to be procured through the anticipated flexible mechanisms, a similar approach to that adopted by the Netherlands. The position of the Federal Council for Sustainable Development is said to be very much pro-CO2 tax, but against domestic emissions trading, while only Electrabel, an electric utility, supports the latter.[60] Also, given the size of the economy, and thus the size and diversity among its industry, Belgium may not benefit from domestic trading. Thus, the current decision is to let the Belgium industry directly participate in the EU-wide scheme without having a domestic trading scheme.

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57. While this is the common view, recent scientific evidence has shown that the largest part of this reduction has resulted from active climate protection measures, according to the Federal Government of Germany.

58. The base year was later changed to 1990. OECD (1999b).

59. In its introductory years the tax shift resulted in a 0.8% cut in total social security contributions and the objective is to reduce social security contribution by a further 1% by 2003, aimed at triggering the 'double dividend."

60. Source: a meeting of Concerted Action on Tradable Emissions Permits (CATEP), 14 May, 2001.

Switzerland: The government intends to meet the Kyoto target of an 8% reduction from the 1990 levels by 2008-2012 primarily through voluntary agreements.[61] Switzerland has had a number of voluntary measures for energy efficiency in place under "Energy 2000" and subsequent "SwissEnergy" Action Plans.[62] Under the latter, the government has set a target to reduce consumption of fossil fuels by 10% between 2000 and 2010. The federal law on CO2 emissions came into force in May 2000, under which total CO2 emissions are to be reduced by 10% of the 1990 level by 2010 through voluntary measures. A new directive was introduced in 2001 to provide a further framework for these voluntary agreements. "Engagements formel" would set emission reduction targets agreed upon with individual companies or associations of at least five companies which together emit more than 10,000 tonnes of CO2 per year. Those that commit to the voluntary targets are to be granted exception from a CO2 tax, which would otherwise be introduced after 2004. The CO2 tax would be levied at a maximum rate of CHF 210 per ton of CO2.[63] However, a recent referendum rejected those plans. Introduction of a formal domestic emissions trading scheme is not part of Switzerland's climate change policy, although the directive allows companies or groups of companies with voluntary targets to exchange emission rights.[64]

4. Combining domestic GHG emission trading with other policy instruments

TPs are a relatively new policy tool compared to other economic instruments, traditional command-and-control measures, and voluntary approaches in general, and in particular for domestic climate change policy in most countries. As reviewed above, taxes and voluntary approaches (VAs) have already been in use aimed at GHG emission reduction in many OECD countries. Therefore, it is important to consider the issues related to design and implementation of a domestic GHG emission trading scheme in relation to taxes and VAs, within the overall climate change policy at the national level. Domestic GHG emissions trading may be introduced to be implemented parallel to, or to replace already existing taxes or VAs for achieving a common goal. Also, GHG emission trading could be designed integrated into existing or new taxes and/or VA programmes as a coherent package. As described earlier, a number of the domestic GHG emissions trading schemes, such as those for the UK and France, have been designed or envisaged with close linkages to taxes and/or voluntary agreements inherent in their architecture.

In this section, the main characteristics of taxes and VAs and their use for curbing GHG emissions will be briefly reviewed. Then, the issues of compatibility of domestic emissions trading, taxes and VAs, and which instruments are suitable under what circumstances and in what combinations will be explored.

4.1. Taxes and TPs: theoretical assessments

Similarities: In theory, both taxes and TPs function through sending direct price signals to firms and consumers on environmental resource scarcities and the cost of emissions, thus influencing their production and consumption decisions. Therefore, taxes and TPs share many common

61. OECD (1999b).

62. IEA (2001):http://www.idea.org/pubs/newslet/eneeff/ch.pdf. (Updated November 2001)

63. Majocchi, A., (2000), Greening Tax Mixes in OECD Countries: A Preliminary Assessment, OECD Paris; and Swiss Agency for the Environment, Forests and Landscape: http://www.buwal.ch/klima/f/klimapolitik.htm.

64. E. Haites and F. Mullins (2001).

characteristics.[65] As is the case for TPs, taxes are also a cost-effective policy instrument that can minimise the overall costs through equalisation of marginal abatement costs across firms for achieving a given environmental objective. Taxes also provide flexibility to firms whether to abate pollution and pay less in taxes or to keep the level of pollution and pay more in taxes. Either way, under both taxes and TPs, firms pay both for abatement costs and for remaining emissions.[66] Also, taxes act as incentives for technological innovation and further reduction in polluting emissions. Moreover, both taxes and TPs allow the flexibility for preferential treatment of certain firms or sectors for competitiveness reasons; through free initial permit allocation (while others buy through auction) and through lower tax rates or tax exemptions, as we have seen for energy-intensive sectors exposed to international competition.

Differences: While taxes and TPs can be equally cost-effective, they function through different paths. Taxes differ from TPs in that that can set the cost of emission abatement, but not the quantity, while TPs can set the level of emission abatement, but not the cost which is reflected in the permit price. This means that in situations where the assimilative capacity of the environment requires strict control over the level of emission abatement while the cost of emission abatement is not expected to be excessive (i.e. where the social cost of environmental damage is expected to increase more steeply than that emission abatement costs), then a TP scheme is more suitable. In converse, for situations where emission abatement cost is expected to increase more rapidly than the social cost of environmental damage, then taxes may be more suitable. Another main difference between taxes and TPs is the knowledge of the payments, i.e. taxes rates are know in advance while permit prices are unknown. This could potentially have different implications for investment planning from a firm's point of view.

Both taxes and TPs allow the flexibility for preferential treatment of certain sector for strategic (competitiveness) reasons, but with different consequences.[67] While tax exemptions provides no incentives to abate emissions, preferential treatment of certain firms or sectors through free initial permit allocation still provides incentive to reduce emissions, as these freely allocated permits can be sold for revenue if emissions are reduced to leave excess permits. This also means that competitiveness concerns for certain sectors can be addressed though free initial permit allocation without compromising dynamic efficiency of TPs, while providing tax exemptions would no longer provide incentives for technological innovations.

The recent OECD guidelines[68] on designing and implementing tradable permits draw out the criteria for choosing between TPs and taxes related to incentives structure, organisational and transaction costs, and redistribution, including the following:

(i) *Sensitivity of firms and consumers to price vs. quantity signals*: – As different regulated sectors have different competitive structures, they react to different variables. Taxes are more appropriate if agents are sensitive to prices, while if the industry is oligopolistic in structure and product demand elasticity is low, quantitative limit on emission would be more suitable.

(ii) *Existing institutional framework*: – It is important to assess the capacity of existing authorities that would implement and manage tax or TP schemes. If organisational and

65. OECD (2001b).

66. This would be the case only if permits are auctioned. Furthermore, it assumes no revenue recycling of auction proceeds.

67. Schreiner, P. (1999).

68. OECD (2001a).

transaction cost of establishing a new administration for a new trading scheme is deemed excessive, for example, introducing new taxes through existing tax authority may be easier. Meanwhile, for an "upstream" TP scheme which target suppliers of fossil fuels, existing institutional frameworks for taxes might be used to help administer it, as long as the point of imposition for TPs are the same as those used in the existing tax regime. Also, if the existing monitoring mechanism is not sufficient (or lacking) and the additional cost of compliance monitoring and verification of emission reduction is anticipated to be very high (e.g. for numerous small emission sources such as SMEs or mobile sources, or for a GHG for which measuring technology is expensive), it may outweigh the benefits of trading and thus energy or carbon taxes may be more appropriate.[69] Simplistically put, from the standpoint of transaction and monitoring costs, emission trading would be more suitable for large point sources.[70]

(iii) Distributive impacts: - While both taxes and TPs are compatible with different distributive options, the choice between the two have certain implications. In most cases, energy or carbon taxes pre-exist before introduction of domestic GHG emissions trading. Therefore, if trading with grandfathered permits is to replace taxes for a given sector, it would mean a loss of tax revenue to the government, while the concerned industry would not only have reduced tax burden, but also be assigned newly created assets. Naturally this would be an option favoured by industry, and currently found in most of the domestic GHG emission trading schemes proposed. Even in case of auctioned permits, whether the revenue they generate is expected to be more or less than the current tax revenue can influence the preference or choice between taxes and trading. This has been a contentious issue in discussion of the domestic trading schemes in and Norway where they are proposed potentially to replace existing energy taxes.

These aspects above imply that, for climate change policy at the national level, taxes and TPs can potentially complement each other by addressing GHG emissions from different sectors with different market structures (i.e. as they may be sensitive to different variables) for maximum coverage and results. This is the rationale for the cases where domestic emission trading is introduced to supplement existing taxes in different sectors, as seen in the case of Denmark (emission trading for electricity producers and taxes for other sectors). However, where emission trading with initially grandfathered permits is introduced to replace pre-existing energy or CO2 taxes, it represents a significant resource loss to the government and gain to the industry. If initial permit allocation is by auctioning, *ex ante* estimations of the level of revenue from auctioning could guide the final choice of instrument, depending on the desired distributive outcome. However, as seen from the case of Norway, reaching a political consensus on what the "desired" distributive outcome is can be troublesome.

If we are to consider compatibility with the flexible mechanisms being considered under the Kyoto Protocol, a domestic TP scheme may fit more easily than taxes. Assuming that a national government allows firms to acquire and use credits for emission reductions abroad (or Kyoto mechanism units) along with domestically-issued permits to meet the requirement to match emissions with permits, the existence of a domestic emissions scheme will create a demand for import of Kyoto mechanism units. Further, in these circumstances, a domestic emissions trading scheme would be

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69. However, there are also differing views on this point. For a tax system to work properly, it also requires accurate and consistent emissions data from sources. Therefore, monitoring requirements should be of comparable rigour for both emission trading and taxes.

70. Although there are counter-examples that demonstrate feasibility of TP schemes for numerous non-point sources. See, for example, the Austrian ECOPOINT programme for trucks in Chapter 5 by Charles Raux.

expected to keep the domestic "price of emissions" in line with the price of Kyoto mechanism units – a condition for cost-effectiveness in achieving a Kyoto target. Under an emissions tax approach without a domestic trading scheme, on the other hand, the government would presumably have to arrange for direct import of Kyoto mechanism units if such an import were required to meet the country's overall Kyoto obligation. In this case, the domestic price of emissions (corresponding to the tax rate) would not likely be kept closely in line with the price of Kyoto mechanism units in the international market.[71]

4.2. Voluntary approaches and TPs.

Voluntary approaches (VAs) have been developed in response to the need for more flexible ways to achieve a given environmental objective while accounting for the concerns about competitiveness of industries. Thus, they have traditionally been favoured by industry for their flexibility, and also as a way of avoiding formal regulations and taxes. To date, there is no standard terminology to designate the broad category of voluntary approaches. For the purpose of this section, VAs referred to those that are agreements entered into between government and firms or industry (e.g. negotiated agreements on certain environmental targets) rather than truly "voluntary" private initiatives (e.g. voluntary codes and company environmental management programmes) which are, strictly speaking, not a public policy instrument. A recent OECD review of VAs[72] shows that they have often been used in a policy mix to supplement regulations and taxes, as VAs can, under the right conditions, improve the flexibility and acceptability of the policy package. For example, a tax scheme can include voluntary agreement provisions under which firms would be granted tax exemptions in return for taking certain actions towards meeting an environmental goal. The earlier section provided some examples of energy or CO2 taxes being exempted or reduced on the conditions of VAs. While the use of VAs without regulatory backing or economic instruments in combination has been found to be economically inefficient, the overall performance of this kind of policy mix still remains uncertain, as few examples are available so far in practice, and for those that do exist, *ex post* evaluations are yet to be carried out.

Many OECD countries have in fact relied on voluntary approaches as a first step in their efforts to reduce CO2 emissions in energy-intensive sectors. For example, industries enter into agreements with regulators for energy efficiency standards or emission reduction targets, but without specifying the means for reaching these targets. As illustrated in the cases for GHGs, VAs have often been used in new policy areas not yet covered by existing regulations. More recently, several domestic GHG emissions trading schemes have been designed with close linkages to voluntary agreements. For example, the proposal for the UK is one model whereby entering into voluntary negotiated agreements for emission reduction allows firms to participate in domestic credit trading. The same agreements are at the same time linked to the Climate Change Levy schemes as a condition for exemptions. While the novelty of integrating emissions trading, taxes and voluntary agreements in this manner is worth noting, it is too early to determine its effectiveness.

For the combination of voluntary agreements and TPs to be effective, some conditions would need to be met, including the need for rigorous monitoring of firms' performance against the agreed targets, just as much for TP schemes. Also, as participation is by definition "voluntary, " how to capture emissions from those firms that do not enter into these agreements must be worked out. With examples like the UK's Climate Change Levy Agreements, which set the conditions for being granted tax discount, this would not be an issue. However, cases like that of Switzerland, where emission

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71. See Bjart J Holtsmark, (1999), "A comparison of taxes and tradable permits in national climate policy", Working Paper 1999:8, Center for International Climate and Environmental Research (CICERO), Oslo.

72. OECD (1999b).

reduction is agreed to on a purely "voluntary" basis, environmental effectiveness cannot be guaranteed.

4.3. A word of caution on TPs in policy mixes

Combining TPs with other instruments can be a cost-effective policy package for addressing a given environmental goal, if they supplement each other in a predictable way to achieve a desired outcome, and if a single policy instrument could not achieve the same outcome more effectively. However, a combination of multiple instruments is not without potential pitfalls.

First, packaging multiple policy tools could mean potential problems during implementation phase, undermining potential efficiency of them individually. Smith[73] argues that a mix of instruments applied to the same problems could undermine the efficiency of the main economic instrument(s) if they are overlapping and not supplementary, and even wasteful if using more than one instrument leads to higher administrative costs to be born by the government or the regulated emitters. Schreiner[74] maintains that, for the proposed Norwegian emission scheme designed for the widest coverage, it is better to extend that system than to add additional instruments. Ellerman[75] also join in this cautious view. He points out that the success of the US SO2 programme was based on the fact that "simplicity encourages accountability, and strict accountability enables flexibility." In fact, the UK scheme, combining TPs, taxes, and VAs, has sometimes been viewed as being overly complicated. The main complication arises from the "gateway" that links two markets, one for those with absolute targets and another for "relative" participants stemming from negotiated agreements (CCAs), which, in turn, is related to Climate Change Levy.

Second, mixing domestic emissions trading and other instruments could potentially pose a problem of equity between the group of firms covered by the trading scheme and those that are outside of it. Ellerman notes that, even if the combination of instruments is adequate to ensure meeting the Kyoto targets, it is unlikely that the effect of each instrument on the respective sectors would be equal, as differing instruments will almost surely create differing marginal costs. Also, if major point source emitters are untouched by any scheme, e.g. electricity generators or the energy-intensive industry sectors, this could potentially lead to large differentials in compliance costs and an inefficient pattern of abatement efforts.

4.4. Introducing domestic GHG emissions trading: interface with taxes and VAs in practice

Despite these cautious views on policy mixes, one needs to keep in mind that greenhouse gases are quite different from many other environmental problems. The GHG emissions profile of a given country can be complex, comprising emissions of different GHGs from numerous heterogeneous sources operating under different industrial and market structures. Thus, it can be argued that this diversity could potentially be better served by a mix of several instruments, each acting on different variables for each target group, to achieve maximum overall results. Naturally, the first step in designing a domestic GHG emissions trading scheme would be understanding the characteristics of the emissions sources in diverse sectors, and the conditions under which their

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73. Smith, S. (1999), *The Compatibility of Tradable Permits with Other Environmental Policy Instruments*, in OECD (1999a).

74. Schreier, P. (1999), *Obstacles to the Implementation of Tradable Permits: the Case of Norway*, in OECD (1999a).

75. Ellerman (2000).

polluting activities are carried out. In the presence of pre-existing taxes and voluntary agreements aimed at GHG emission reduction, the effectiveness of other instruments, would need to be assessed first in order to determine whether to maintain, modify, or eventually replace them (e.g. as proposed for CO2 taxes in Sweden), before considering how best to combine or integrate them with emission trading. This would be the staring point for determining the sector coverage of a domestic trading scheme.

In practice, the environmental effectiveness of energy and carbon taxes in place in most OECD countries have been limited, due to pervasive exemptions often granted to most energy-intensive sectors, including electricity generation.[76] In most cases, when governments introduced taxes on fossil fuels, the electricity sector has been exempted because it was not possible to tax electricity imports on the same bases.[77] Instead, they applied a general tax on electricity consumption, which is not as efficient to encourage CO2 emission reduction. This is because directly taxing and adjusting relative tax rates for different fuel inputs for electricity generation can lead to desired fuel switching to less carbon-intensive ones, while the direct impact of general tax on electricity would only be on demand and consumption. Thus, in practice, most energy and carbon taxes in majority of OECD countries are almost exclusively levied on households and transport sector.

Given the current lack of coverage by taxes for electricity sector, one logical way of introducing emission trading as a supplementary measure to taxes would be to fill this gap. Capturing CO2 emissions from electricity generation sector, which make up large shares of many countries' CO2 emissions portfolio, is crucial for their climate policies. Meanwhile, most of the existing energy and carbon taxes are levied on households and transport sectors. Also, incidentally, due to the transaction and monitoring cost concerns discussed earlier, households and transport sector would be less suitable for TPs than for taxes. Thus, a common-sense conclusion would be that these sectors should be covered by taxes rather than emission trading. This, of course, is a generalisation, as each country's GHG emission profile is different than another, yet a valid one. The Danish emission trading scheme specifically targeted at the electricity sector is one such example. However, this approach, in the context of the electricity market liberalisation, could result in CO2-leakages when electricity is traded across borders between countries with power generation of different carbon-intensities.

The results from a policy simulation carried out by the OECD[78] imply that taxes alone may not be able to sufficiently reduce GHG emissions to meet the Kyoto target for Annex I countries, which is a 5% reduction from 1990 levels to 2008-2012. Under existing policies, OECD countries are likely to increase CO2 emissions by a further one-third to 2020, far from the Kyoto target. The simulation modelled removal of subsidies on energy production, and annual increases of energy taxes to 2020, eventually reaching high tax rates of 50% for coal, 40% on crude oil and 30% on natural gas of pre-tax prices in 2020. Results showed that CO2 emissions would be reduced by 25% in OECD regions in 2020 compared to a business-as-usual scenario. However, to 2010, there would still be an increase in CO2 emissions by 3% from the 1990 level, falling short of the Kyoto target. As tax increases of this magnitude, even combined with removal of subsidy, would still be insufficient for meeting the Kyoto target, it points in the direction of considering other supplemental measures such as domestic emission trading.

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76. OECD (2001b).

77. OECD (1997) *Economic/Fiscal Instruments: Taxation* , Annex I Expert Group on the UNFCCC, Working Paper No. 4, OECD, Paris.

78. OECD (2001c), OECD Environmental Outlook, OECD, Paris.

This poses a question, in converse; could domestic trading alone lead to GHG emission reductions that meet the Kyoto targets in some countries? In other words, could domestic emissions trading be introduced as a substitute for taxes and regulations, to eventually replace them? In principle, it could, as the total cap would be set equal to the desired total domestic emissions reduction. The proposed Norwegian emissions trading scheme is one such example with the widest possible sector coverage of nearly 90 % of Norway's 1997 GHG emissions, intended to replace the current CO2 taxes. Proponents of such universal coverage of domestic emissions trading schemes emphasise the equalisation of marginal abatement costs across sectors as one of the most important considerations in the choice of policy tool(s), as different actors under different policy tools are bound to have uneven abatement costs. However, as discussed above, such a shift from taxes to TPs would lead to substantial resource shifts where allowances are allocated for free, and uncertainty in level of revenue if they are to be auctioned.

5. Preliminary conclusions and areas for further work

OECD countries are considering use of GHG domestic emissions trading to different degrees. In some cases, no plans for domestic emission trading exists because the country plans to rely mainly only on taxes, VAs and/or direct regulations (e.g. Switzerland VA only, and Belgium with taxes). In other cases, planning has advanced on introducing domestic emissions trading but in combination with new or pre-existing taxes, VAs, and regulations. In a few cases (e.g. Sweden and Norway), domestic trading will be introduced, possibly to replace existing taxes. So far, there seems to be no case where TPs will be the only domestic tool for meeting the Kyoto targets, although the Norwegian case is close, as there will always be emission sources that can better be addressed by other instruments.

Some preliminary list of issues to be considered when determining how to strategically introduce (or not) GHG emissions trading scheme within an overall climate policy of a country would include; GHG emission profile of the country, market and cost structure of electricity production and other carbon-intensive sectors, and existing policy instruments and their historical performance. These form the basis of domestic climate strategies, in estimating expected emission reductions to be delivered by each of the policy instrument proposed, and ensuring that they deliver the overall national reduction target. These expected benefits should also be set against costs, both emission reduction costs to be born by the regulated sectors as well as administrative costs born by the government.

The next steps in considering the choice between domestic emissions trading and taxes, in particular, for a given target group should focus on relative economic efficiency of the tools, which would involve finding out; (i) the sensitivity of regulated sectors and consumers to price vs. quantity signals, as these would help determine the optimal coverage of each instrument; (ii) existing institutional framework, which would influence the transitional cost of the proposed policy shift; and (iii) distributive impacts of respective instruments.

In addition to the above, socio-political elements would also need to be understood and recognised. These include industry lobby and their influence, or lack there of, in the country's policy and legislative process, as well as historical or cultural disposition towards each instrument choice (e.g. long history of effective VAs such as in Japan and Germany), as these also seem to determine the how domestic emissions trading is incorporated into the overall climate policy package.

Some of the new and emerging challenges for future considerations for policy mixes would therefore include the following points:[79]

- patters of relationship, or division of labour/coverage among different policy tools;
- analyses of complementary aspects as well as possible negative interference among different policy tools;
- linkages and interface between tools aimed at the common objective (e.g. the UK case of taxes, VAs and trading);
- linkages with other policy tools aimed at different pollutants (e.g. could we consider combining GHG emissions trading with that for SOx or NOx, for example?); and
- voluntary participation in emission trading as a way of exemption from taxes (as seen in the French proposal, the UK scheme).

79. See Chapter 8; Godard, O. (2001), *Domestic Tradable Permits: Future Directions.*

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Chapter 4

RENEWABLE ENERGY CERTIFICATES: TRADING INSTRUMENTS FOR THE PROMOTION OF RENEWABLE ENERGY

by

Richard Baron and Ysé Serret[1]
Energy & Environment Division, International Energy Agency

Executive Summary

A number of countries are introducing tradeable renewable energy certificates (TREC) to support the development of alternative energy sources in more competitive electricity markets. These systems either substitute or supplement earlier mechanisms that have proven either too costly, or not efficient as means to promote the technological progress and market penetration of renewable electricity (RE). As the need to reduce greenhouse gases emissions – in particular energy-related CO_2 – becomes a priority in energy policy, governments turn to instruments that appear to be more efficient to promote the development of renewables, including trading mechanisms.

This paper offers a survey and a discussion of government-led efforts combining a renewable energy quota with tradeable renewable energy certificate systems – renewable energy certificates are also used (and traded) to support "green" electricity pricing regimes, not addressed in this paper. The principle of such systems is simple: power companies are mandated to produce a certain share of total output from RE sources and must redeem certificates to the regulator testifying that they have done so. Certificates are issued once a certain quantity of RE has been put on the electricity grid. Electricity and RE certificates may be traded separately.

We cover the following elements of TREC: the definition of the objective; the nature of liable entities; the eligible renewable energy sources (RE), since countries differ on their definition of renewable or "green" energy; the level of the penalty; the accompanying measures to support RE and the question of international transactions. We do not address details related to certification, monitoring, and institutional arrangements pertaining to them.

TREC are still at a very early stage: an assessment of their efficiency is therefore not possible. On paper, they offer clear theoretical advantages – those of tradeable permits when compared with command-and-control and targeted subsidies. But they are less predictable when it comes to the premium given to renewable sources – hence the mixed feelings about TREC among RE generators that so far benefited from fixed tariffs. The implementation details reveal that some adjustments were necessary to take into account the pre-existing support schemes for renewable sources of electricity.

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These accompanying measures will have some impacts on the economics of TREC systems although the exact magnitude of these effects is difficult to measure.

In addition, the efficiency of TREC should be measured against the underlying renewable energy policy goals. The development of a portfolio of options with various costs and market niches may not necessarily be achieved with a "blind" market system, hence a number of adjustments and complements to the systems currently envisioned.

As is common in trading instruments, the cost of achieving specific RE generation objectives would be reduced if international transactions were allowed. The recent EU Directive on the promotion of renewable electricity creates the building blocks for an international regime, although it does not require countries to choose TREC as the tool for the promotion of renewables, and the countries' targets are indicative. This paper highlights some of the pros and cons of an international system, in light of existing domestic systems.

Because renewable power sources contribute to lowering greenhouse gas emissions (GHG), we consider the interaction between TREC systems and greenhouse gas emission trading, another tradeable permit regime that could apply to power generation in the near future. While ambitious CO_2 emission reductions play in favour of an increased renewable electricity capacity, economic analysis shows that TREC can in some cases isolate renewable energy from the incentives brought by GHG trading. We also envision the possibility of allowing RE certificates to be traded in GHG emission trading systems. This option raises one significant problem: the uncertainty of GHG emission levels once the GHG trading regime is open to TREC. TREC systems are not based on absolute caps: objectives are expressed as *percentages* of total electricity output. There may be good reasons, for the proponents of both renewable electricity and GHG mitigation, to keep the two trading systems separate.

1. Introduction: why tradeable renewable energy certificates?

Since the early days of the first oil shocks and the threat that they brought in energy policy-makers' minds about the rapid depletion of oil resources, governments have struggled to promote the development of new and renewable energy sources. In addition to their renewable nature, these energy sources offer many advantages:

- they are substitute to depletable – and often imported – energy sources. They contribute to improve the energy security of countries that would otherwise depend more on imported energy;
- their impact on the local and global environment is small in comparison with conventional energy sources;
- even if higher in current conditions, their cost is rather stable compared to fossil fuels – even though availability is sometimes questioned;
- as an emerging industry, it is argued that RE creates more employment than conventional sources and contributes to regional economic development.

However, when considered on the basis of an engineering-cost calculation, most renewable energy sources remain relatively costly, even if sizeable reductions in cost have been observed over the last few years. However, some technologies are competitive in certain off-grid, niche and power markets. A portfolio analysis also tends to point to higher competitiveness than otherwise assumed.

Most OECD countries have undertaken reforms of their electricity sectors, seeking to improve their economic efficiency through more open and transparent markets.[2] Market reform alone will not lead to the automatic internalisation of the external costs of power generation. Furthermore, if such reforms are successful, they should ultimately lead to lower electricity prices, increased consumption and generation of electricity – and related environmental impacts. This has become a growing issue as climate change concerns and the Kyoto Protocol under the UNFCCC, in particular now, call for a drastic departure from current energy practices. While other options exist to reduce energy-related CO_2 emissions, renewable sources are key to the achievement of a more climate-friendly energy world. But their contribution to electricity demand to date has been limited (16% of total output in 1999, 14% of which comes from hydro).[3] Electricity market reform processes are therefore more a threat than an opportunity for the competitiveness of RE.[4] In addition, existing support schemes for renewables are being challenged as too costly, or as not providing a level playing field for renewables[5] – or for power generators, as their markets become open to international competition and not all are required to support RE to the same extent.

While the high cost might have been acceptable to date, a significant expansion of RE technologies calls for more cost-effective systems. The possibility offered by newly-open markets to market renewable electricity at a premium led to the idea that RE sources could compete for a share of electricity delivered to the grid, to be fixed on a voluntary basis – the Green Labelling system of the Netherlands – or by the government.

Tradeable instruments, well known to environmental economists, have thus found a new field of application as policy support schemes for renewable – and other clean energy sources – with the establishment of markets for renewable energy certificates. This paper provides details on tradeable renewable energy certificates systems combined with mandatory renewable energy goals (TREC). It studies in particular its international dimension and its implication for GHG policy in the context of international GHG emission trading.

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2. See IEA (2001): Electricity market reform: California and after. *A fact sheet from the International Energy Agency*, http://www.iea.org for an overview of electricity market reform in industrialised countries.

3. IEA (2001): Energy Balances of OECD Countries, 1998-1999. IEA/OECD, Paris.

4. Even if some companies are seeking to gain market shares by advertising – and certifying – the "green" nature of the purchased power. Green pricing systems are developing rapidly in some countries, but not yet to the point where RE makes a significant dent into the overall electricity supply.

5. The otherwise successful Non-Fossil Fuel Obligation of the UK, by setting the share of each generation source in its tenders, set artificial limits on options that could supply electricity at lower cost than other sources promoted under the Obligation. But this is also a reflection of the multiple policy goals pursued with the development of renewable energy.

2. What are tradeable renewable energy certificates

2.1 The objective(s) of TREC systems

A priori, the goal of TREC is to encourage the development of renewable or cleaner energy sources through a market mechanism. There are essentially two forms of policy tools that have been used for that purpose to date:[6]

- feed-in prices: the government or regulator guarantees a fixed price for renewable electricity (this tool was applied in Denmark, Germany, Spain and France, to a lesser extent);
- tendering procedures: the government calls for a certain quantity of renewable energy to be delivered at a given date and funds the least-cost projects (the UK's Non-Fossil Fuel Obligation, and France's Eole Programme are two examples of tendering procedures – both systems are now being phased out).

Menanteau *et al.* (2001) have analysed the pros and cons of these two "polar" options. They show that countries that implemented feed-in prices have secured a faster growth in renewable energy capacity (mostly wind), while tendering procedures have had a more limited impact on the overall penetration of renewables. In terms of technological process and research and development efforts, feed-in prices have proven more effective as they provided a more stable investment and research environment. On the other hand, they have turned out to carry a higher price for tax-payers and consumers than tendering procedures. The latter were much more effective to bring down the price of generated renewable energy through a fiercer competition, but did not develop RE technologies as much as originally intended.[7] In fact, the majority of wind projects under the UK's Non-Fossil Fuel Obligation (NFFO) programme have been equipped by Danish wind technology, developed thanks to the country's more stable feed-in tariff approach. Neither system seems entirely satisfactory in a longer-term perspective.

In the case of tradeable renewable energy certificates combined with quotas, the government assigns a multi-year schedule for minimum production of renewable electricity to the electricity companies – i.e. a percentage of annual sales. Tradeable certificates corresponding to electricity from RE sources are issued to RE generators who then sell them to liable entities, either bilaterally or through a market. The electricity can be sold separately on the wholesale electricity market.

Hence, the price asked for a TREC does not reflect the total generation cost of the underlying renewable electricity, but only the extra-cost of generating electricity from RE sources up to level mandated by the obligation, when compared with the wholesale electricity price. In a perfect market, the price of TREC represents the difference between the wholesale electricity price and the marginal development cost of RE sources within the system.[8]

Under a more competitive market power environment, liable entities should go for the cheapest available certificates. This in turn encourages RE producers to constantly advance the technology to reduce production cost and to secure a larger market share in the TREC market (or higher profit margins). In so doing, TREC systems should promote technological progress and market

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6. Menanteau *et al.* (2001), Hoogland and Schaeffer (1999).

7. Hoogland and Schaeffer (1999) also argue that the change in mix of different RE sources from one year's tender to the next makes it difficult for participants to assess the market for RE in the long term.

8. Menanteau *et al.* (2001)

penetration of RE sources – at least up to the point set by the quota. Ideally, the price of traded certificates should draw towards 0, an indication that the policy has made RE entirely competitive with more traditional sources of power...

2.2 Defining the trading system

In general, tradeable permits systems require that liable entities (sources) be subject to a quantitative goal and that the corresponding permits be tradeable. In the case of TREC systems:

- A certain quantity of renewable electricity has to be generated / delivered – rather than a "cap", this is a minimum, or a "floor". In most cases, the precise quantity is defined *ex post* as a share of total electricity output / sales over the period.
- Renewable energy certificates are issued as a proof that a certain amount of renewable energy has been generated, and can be traded by generators and / or liable entities.

TREC systems are therefore similar to transferable permits based on so-called averaging:[9] on average over one year, x% of the sold electricity must be from renewable energy source. Because liable entities are free to choose TREC from any generator, they can opt for the cheapest available option, which helps assure that the set goal is met at minimum overall cost.

Because liable entities may not have access to such sources themselves, the regulator – in many cases, the operator of the electricity grid – issues certificates to electricity producers for each unit of power that is produced from eligible renewable energy sources.[10] TREC are therefore denominated in megawatt-hour (MWh). They are the proof that the indicated volume of electricity has been produced from an eligible RE source.[11] The RE generator sells the issued certificates to a liable entity. A registry is usually set up to track the ownership of certificates, alongside other information (date of issuance, plant, etc.)

In order to comply with the mandatory requirement, liable entities must redeem renewable energy certificates amounting to the pre-defined percentage of their total output. Targets are usually fixed over a one-year period. Assuming a 3% RE objective is assigned to electricity retailers, if such a company sold 10,000 MWh in a year, it must redeem 300 MWh-worth of certificates to the regulator.

2.3 TREC, the environment and other renewable energy policy goals

The certificates embody the environmental benefits obtained from producing power from renewable sources, when compared with other conventional sources that are in use in any given country. As such, they fit in the category of tradeable permits used for environmental policy. Among the environmental attributes of renewable energy certificates are:

- avoided local emissions (SO_x, NO_x, particulates, volatile organic compounds, etc.);

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9. See pp. 17-18 in OECD (2001): *Domestic Transferable Permits for Environmental Management – Design and Implementation*, OECD, Paris.

10. An obviously more costly option would be to require *each and every* generator to *produce* x% of its total output from renewable sources. The tradeable nature of these systems avoids such costly arrangements and encourages RE sources that are the cheapest, regardless of their ownership.

11. See pp. 24-25 in OECD (2001) on the definition of permits in tradeable permit systems.

- avoided greenhouse gas emissions (CO_2 primarily, N_2O, CH_4).

Less known benefits of RE are increased security of energy supply through lower dependency on imported resources, lower exposure to price variations of these resources and therefore a potentially improved trade balance – although RE sources have not to date had significant impacts on these fronts. In addition, the emerging renewable energy industry has created jobs in those countries that have fostered such development.[12] In some cases also, RE plants are also more acceptable to local communities and raise fewer siting problems, than conventional power plants.[13]

The strict environmental benefits of renewable energy could equally be reflected through the full internalisation of environmental externalities in the price of other energy forms. TREC appear to be a form of second-best – the political logic being: if you cannot tax the "problem", subsidise the "solution". Ecological taxation and other environmental policies have progressed towards that aim, but a lot remains to be done.[14] There are also other, non-environmental, market failures that have hampered the penetration of renewable energy. Taxing fossil energy – or capping CO_2 emissions – would not necessarily remove these barriers either.

There is also a sense that other attributes of RE – in particular the need to find reliable alternative sources of supply for the longer term – must be promoted for their own sake. On the other hand, governments that have introduced TREC systems usually stress the immediate gains that these will bring in terms of reduced CO_2 emissions to meet the – relatively short-term – Kyoto Protocol greenhouse gas objectives, even if these reductions come at a relatively high cost. In all, the main goal of TREC tends to be the general support to RE sources, but also part of a broader package of policy measures taken to respond to local and global environmental goals.

2.4 Related but different instruments: "green" pricing mechanisms

In this paper, we distinguish TREC systems from measures that are also based on the issuance of certificates for RE, but are not based on a government objective for renewable energy. A large number of electricity retail companies in IEA Member countries have introduced pricing options that allow end-users to opt for electricity based on renewable energy, provided that they pay a premium on the price of their electricity.

There is a wide range of options available for these "green pricing" mechanisms. Some companies propose that only a share of the delivered energy be produced from RE while others propose that 100% be based on these technologies. End-users may also have the possibility to choose the particular form of energy that they would like to sponsor through their supplementary fee. In order to guarantee that the delivered electricity is indeed renewable – or, rather, that this quantity has

12. For a discussion of renewable energy as a contribution to sustainable development, see IEA (2001): *Toward a Sustainable Energy Future* – "Renewables: Strategies for Market Acceleration". p.131-150. IEA/OECD, Paris.

13. Even if not all communities welcome the installation of wind farms, for reasons related to landscape, noise, threats to birds, etc.

14. The OECD developed an on-line database of environmental taxation at http://www.oecd.org/env/policies/taxes/index.htm. See IEA (2001): *Dealing with Climate Change* – Policies and Measures in IEA Member Countries. International Energy Agency, Paris, for a database covering the full range of policy options adopted to reduce energy-related GHG emissions.

been produced from a RE source and that the premium paid by the consumer was used as payment for this electricity – electricity companies rely on certification procedures and labelling organisations. For instance, three different forms of clean electricity labels are currently available in Sweden. However, in some countries companies that are subject to a renewable energy obligation also offer "green pricing" schemes to their end-users. Governments have generally sought to keep the two systems separate to avoid having certain end-users pay for RE that would have been produced anyway under the obligation. The electricity is therefore sold on top of the RE obligation.

3. A survey of TREC systems

This section offers an overview of existing TREC systems and, in some cases, of those in discussion, although information on the latter is not always reliable. The existing information on the official TREC systems is summarised in the tables below.

Table 1 presents the main features of government TREC systems including:

- The definition of the renewable energy obligation – *i.e.,* the target – and its legal status;
- The liable entities, companies responsible for fulfilling the set objective (Electricity producers, retailers, consumers etc.);
- Eligible electricity sources in the system, *i.e.,* the definition of "renewable" or "green" electricity. Some countries include co-generation of electricity and heat. Others exclude energy produced from incinerated waste;
- The penalty level in case of non-compliance, the possibility to save unused certificates for use in the following period – banking – and the possibility to make up an existing shortfall in the future target period – borrowing.

Table 2 provides further details on these systems and complementary RE support measures:

- The specification of the certificate, including denomination, information provided by the certificate regarding its origin, and underlying energy source, and expiration date;
- Other forms of incentives that applied and/or still apply to the same energy sources;
- The international dimension of the system. Is the system open to acquisition of certificates from other similar systems in other countries, and under which conditions?

The section below the following tables offers complementary information and comments on the systems presented here.

Table 1. Main features of government TREC systems

Country	Target	Liable entities	Eligible electricity sources	Penalty, banking and borrowing
Australia	Mandatory Renewable Energy Target aims to raise the contribution of RE by 2%. Will require an additional 9,500 GWh by 2010. 2001: 0.24% (equivalent to 300 GWh) Percentage of renewable electricity specified by the regulator annually (Office of the Renewable Energy Regulator) (Legislation, 8 December 2000; Effective 1 April 2001 to 1 January 2010)	Wholesalers Applies to all wholesale purchases of electricity on grids of more than 100 MW of installed capacity (excluding standby plants).	All renewable sources (including large hydro) and bio-gas, sewage gas, co-firing, fuel cells, solar water-heaters and pump-storage net of fossil fuel inputs to electricity. A REC is created if generation above a 1997 power generation base line.	• Penalty: above 10%, a AUS$ 40/MWh penalty applies to total shortfall. Repaid if shortfall is made up in 3 years. • Banking is unlimited. • Borrowing allowed to contribute to 10% of target.
Austria	8% of the electricity sold to final consumers should be produced by small-scale hydropower plants. (Legislation, 2000 ; effective 1 October 2001, starting date in the implementation legislation of the Provinces 1 January 2002) Target for other renewables: 1% in 2001; 2% in 2003; 3% in 2005 and 4% in 2007.	The obligation rests on registered suppliers, based on their deliveries to end users, otherwise on the end users.	Small-scale hydropower plants with a maximum capacity of up to 10 MW.	• Penalty: level defined by the Provinces. • Banking: certificates not older than 2 years.
Belgium (regional level)	• Flanders Share of output: 0.96% in 2001 1.41% in 2002 2.05% in 2003 3% in 2004 5% in 2010 Can be adjusted if higher percentages are observed (Legislation, 5 June 2000)	Electricity retailers	> 1 MW of capacity Solar, wind, hydro, biomass, bio-gas, geothermal, tidal and tidal wave energy. Offshore installations in Belgium also eligible. Waste incineration is excluded.	• Penalty: €0.05 / kWh for 2001-2003 and €0.12 / kWh thereafter. Revenues recycled in the Renewable Energy Fund Banking: valid for year of production and 5 thereafter.
	• Wallonia Share of output: 2.9% by 1/1/2002 5% by 1/10/2004 12% by 1/10/2009 (Legislation, July 2001; start date: 1 April 2002)	Electricity retailers	> 1 MW of capacity Hydro, wind (incl. offshore), wood, agricultural products, biomass, "high quality" co-generation: 3kWh lead to issuance of 1 certified kWh. Co-generation can contribute up to 0.7% of 2002 target; 1.4% in 2004; 4% by 2009.	• Penalty: €0.075 / kWh in 2002 and €0.1 from 1 October 2003.
Denmark (proposal)	Consumers required to buy 20% of all their electricity from RES by 2003. (Postponed; originally scheduled to start 1 January 2003)	Electricity consumers (industrial and residential). Can mandate retailers to achieve objectives on their behalf.	Included: wind, central CHP on biomass, bio-gas, incinerated waste with minimum organic matter. Excluded: large hydro, existing biomass and waste incineration plants.	• Penalty: tax of DKK 0.27 per kWh (€0.035) – The certificates have also a minimum price of DKK 0.10 per kWh. • Borrowing allowed.

Table 1. Main features of government TREC systems (continued)

Country	Target	Liable entities	Eligible electricity sources	Penalty, banking and borrowing
Italy	2002: 2% of electricity produced the past year. Obligation over three years (Legislation on 1 April 1999)	Importers and producers of electricity (more than 100 GWh/year) – the only country to apply the obligation to producers, apparently. Electricity exports are exempted.	New renewable energy (after 1 April 1999). Includes new large hydro plants (except pumping storage facilities) and organic waste plants, under strict pollution controls.	• No penalty. No minimum and maximum price. The government would issue additional certificates if cost is deemed to high. • Banking: certificates are valid for one year. • Borrowing allowed.
Netherlands	Voluntary Approach (Legislation 1998, operational 1 July 2001)	Participants in the Green Certificates system: Electricity generators, traders/suppliers	Included: hydro under 15 MW, wind, solar and biomass plant which uses no plastics by way of additional fuel. Plants partly running on biomass by way of fuel have to fill in the correct percentages of biomass-generated electricity in the system. Excluded: large hydro and waste incineration.	• No penalty: voluntary scheme • Banking: certificates valid for a one-year period. • Borrowing not allowed.
Sweden (proposal)	Should start 1 January 2003. Percentage of electricity consumption: 6.4% by 2003 7.6% by 2004 9.5% by 2005 11.4% by 2006 12.8% by 2007 13.9% by 2008 14.6% by 2009 15.3% by 2010	Electricity users or suppliers	Wind, solar, geothermal, certain biofuels, wave energy, existing small-hydro (below 1.5 MW), hydro plants commissioned after the entry into force of the Certificate law, and plants which were not in operation after 1 July 2001, increases in installed capacity at existing hydro plants after 1 July 2002.	• Penalty for non-compliance: 150% of the weighted average price observed in the 12 months preceding 1 April, the date at which certificates must be surrendered for compliance, with a ceiling at 200 SEK/MWh until 2007. The ceiling price would be revised for 2008 onward. • Certificates can be banked and have unlimited lifetime.
United Kingdom (proposal)	Renewables obligation certificate (ROC): percentage of output. 3% between October 1, 2001 and March 31, 2003 (4.3% in 2004, 4.9% in 2005, 5.5% in 2006, 6.7% in 2007, 7.9% in 2008, 9.1% in 2009, 9.7% in 2010) and 10.4% by March 31, 2011. Remain at least at that level until March 31, 2027. (Proposal, Statutory Consultation, August 2001)	Electricity retailers	Only stations commissioned after January 1, 1990, except for micro hydro and co-firing. Landfill gas, sewage gas, some energy from waste, hydro (<20 MW) and new hydro (>20 MW, if commissioned after Order), onshore and offshore wind, co-firing until 2011, other biomass, energy crops, geothermal, tidal, wave, photo-voltaic.	• Penalty: buyout price: 3p / kWh, adjusted by the Retail Price Index. Proceeds recycled to suppliers that met their targets. • Banking for up to 25% of the following year's obligation. • Borrowing is not allowed.

Table 1. Main features of government TREC systems (continued)

Country	Target	Liable entities	Eligible electricity sources	Penalty, banking and borrowing
United States (State	• New Jersey The legislation requires all retail electric suppliers to provide 0.5% of their energy from Class I renewable resources by January 1, 2001, 1% by January 1, 2006 and an additional 0.5% per year until it reaches 4% by 2012. (The Renewable Portfolio Standard (RPS) began at 2.5% in 2000, a requirement that could be met by Class I or Class II Renewables. However, all subsequent additions must be met with Class I renewables.)	Electricity suppliers	Wind, Solar, Biomass, Hydro, Geothermal, Fuel Cells, Waste, wave, tidal.	• Possible penalties for non-compliance: suspension or revocation of the electric power's license, financial penalties, disallowance of recovery of costs in rates, and prohibition on accepting new customers.Borrowing allowed.
	• Texas Renewables Portfolio Standard: The standard calls for 2,000 MW of new renewables to be installed in Texas by 2009. 400 MW 01/01/2002 ; 400 MW 01/01/2003 ; 850 MW 01/01/2004 ; 850 MW 01/01/2005 ; 1,400 MW 01/01/2006 ; 1,400 MW - 01/01/2007 ; 2,000 MW 01/01/2008 ; 2,000 MW ; 01/01/2009 – 2019. A conversion factor is used to translate these capacity objectives into production goals (MWh). (Enacted 1999; Effective July 1, 2001)	Utilities	Solar, Wind, Biomass, Hydro, Geothermal, Tidal, Landfill Gas.	• Penalties for non-compliance: the penalty is the lesser of $50 per MWh or 200% of the average market value of credits for that compliance period. • Banking: valid for three years. • Borrowing: during the first two compliance periods (2002 and 2003), each retailer may incur a deficit of up to 5% of its final REC requirement.
	• Wisconsin Renewables Portfolio Standard: 0.5% 12/31/2001 0.85% 12/31/2003 1.2% 12/31/2005 1.55% 12/31/2007 1.9% 12/31/2010	Electric service providers	Solar, Wind, Biomass, Hydro (less than 60 MW), Renewable Transportation Fuels, Geothermal, Fuel Cells.	• Penalties for non-compliance up to $500,000 • Banking: credits can be used in subsequent years

Table 2. Additional details on government TREC systems

Country	Specification of certificates	Earlier and other support for renewables	International trading
Australia	Unit:1 MWh. Identification of power station (registry contains information on energy source) Year of generation Current owner of the REC No expiration date	The "Measures For a Better Environment" package adopted in May 1999 include: subsidies for up to 50% of the capital cost of installing renewable power facilities in new off-grid applications; a rebate (up to AUS$7,500) for the cost of installing roof-top PV power systems on houses and community-use buildings and a programme to support alternative fuels. The Renewable Energy Commercialisation Programme and the Renewable Energy Equity Fund provide grants to foster the development of a renewable energy.	Not mentioned in the Law.
Austria	Units of 100 kWh or multiples	Feed-in tariffs in favour of small-hydro in some Provinces.	Not excluded but already have to deal with nine different implementing frameworks at the provincial level.
Belgium (regional level)	• Flanders 1 MWh. Specifies other forms of government support (e.g. free distribution of power, etc.). REC with other forms of support only accepted upon conditions.	Feed-in premium paid by distributors or producers to RE and clean energy producers. Level depends on the energy type. Tax incentives and direct investment support (e.g. for PV).	Excluded for the moment. Once allowed, the target will consist of two parts: the agreed Flemish one – to be met by domestic TRECs – and an additional target that can be met by imported TRECs.
	• Wallonia No information yet.	A production subsidy (€75/MWh) is available for renewable energy brought on line after 31 December 2003, for 10 years. Certificates for this energy revert to the government. Prior to that date, renewables are subsidised under an existing federal scheme at a rate of BFr2/kWh (5€¢/kWh) for wind and hydropower and BFr1/kWh (2.5€¢/kWh) for other renewables.	Certificates from other Belgian regions will be accepted; imports from other countries will not be considered until 2003. Sources will have to produce 10% less CO_2 emissions per unit of energy to be considered "green".
Denmark (proposal)	10,000 kWh.	A fixed feed-in tariff is paid to RE producers, in addition to revenues from the RE certificates. Earlier scheme included a consumer paid subsidy for wind, and state subsidy for hydro, bio-gas, and biomass (plus a fixed electricity price). Investment subsidies are also available for CHP and PV. Tax exemptions apply to auto-producers of renewable energy.	Open to international trading provided that imported certificates are from an equivalent quota system (including transparency about existing support systems and the existence of a mandatory renewable energy quota system). However, imported certificates will only have one-fifth the value of domestic ones.

Table 2. Additional details on government TREC systems (continued)

Country	Specification of certificates	Earlier and other support for renewables	International trading
Italy	100,000 kWh Certificates are issued for only more than 50 MWh (rounded to 100 MWh) to minimise certification cost.	In the past, used guaranteed fixed price during first 8 years of operation (including co-generation). Support systems will be implemented in addition to the RE certificate system, e.g. to foster technologies whose cost is still higher than the expected certificate price. This includes "green pricing" – the possibility to charge a premium for renewable electricity to end-users that are willing to pay the extra amount.	Possibilities to integrate the scheme with an international system as long as the certificates comply with domestic parameters. For example, only plants that started operating after 1 April 1999 and export electricity to Italy are recognised as eligible.
Netherlands	There are four "Green Certificates" denominations: 1, 10, 100 or 1,000 MWh	Certificates surrendered to the tax department exempt renewable electricity consumers from the Regulatory Energy Tax (an eco-tax paid by electricity and fossil fuel users). A transition scheme has been introduced to deal with electricity having been introduced prior to 1 July 2001.	As of 1 January 2002, imported renewable electricity is eligible for RE certificates if the corresponding amount of electricity has been imported into the Netherlands. Exporting countries must have de-regulated electricity markets to the same extent as the Netherlands.
Sweden (proposal)	1 MWh Identification number used as key to various registers.	The current support scheme comes to an end by 31 December 2002. It contains: 1) Investment aid to: (a) biomass-based CHP, 25 % of eligible investment costs; (b) wind, 15 % (10 % after 1 Jan 2002 according to Budget Bill); (c) Small-scale hydro (<1,5 MW) 15 %. 2) Operational aid to: (a) wind 0,09 SEK/kWh (1 USD = 10,50 SEK today); (b) small-scale hydro 0,09 SEK/kWh 3) Tax exemption for wind 0,181 SEK/kWh. The Green Certificates System will replace existing schemes.	Bilateral agreements can be considered. Open to international trading provided some conditions under discussion.
United Kingdom (proposal)	1 MWh Units specify the generating station, renewable source used, period of generation.	Generators of renewable electricity are exempted from the Climate Change Levy paid by non-residential electricity customers at the rate of 0.43p/kWh (issuance of a Levy Exemption Certificate). Special funding is foreseen for offshore wind and biomass – buy-out level too low to support these options. The ROC replaces the Non-Fossil Fuel Obligation, a tendering process for renewable energy projects. Renewable electricity sold to end-users under green tariff "green pricing" systems should not count towards the RO, but should continue to be encouraged.	Electricity generated from renewable sources outside of the UK (incl. territorial waters and Continental shelf) will not be eligible for now. Should another EU country undertake a similar obligation, the Obligation would be amended to recognise certificates of supply to customers in Great Britain from that country. Exported electricity not eligible to ROC.

Table 2. Additional details on government TREC systems (continued)

Country	Specification of certificates	Earlier and other support for renewables	International trading
United States (State level)	• New Jersey	Sales tax exemption: full exemption from the state's 6% sales tax for all solar and wind equipment. State Rebate Programme as follows: Small Systems (<10kW) ---------- $5.00/Watt Medium Systems (>10-100kW) ------ $4.00/Watt Larger Systems (>100 kW) ---------- $3.00/Watt Maximum incentive as a ---------- 60% percentage of eligible system cost	Not applicable.
	• Texas A Renewable Energy Credit (REC) represents 1 MWh.	State franchise tax exemption: A company may deduct the total cost of a wind, solar or other qualifying system from its taxable capital or deduct 10% of the system's cost from its income. Texas also offers a franchise tax exemption for manufacturers and installers of wind and photo-voltaic systems.	Not applicable.
	• Wisconsin	Solar and Wind energy equipment exemption: taxpayers exempted from any value added by a qualified renewable energy source for property tax purposes Renewable Energy Assistance Programme(REAP): grant funds for renewable energy systems incorporated into construction projects. The WisconSUN Demonstration Grants: net metering[15] is available to all customer classes and to customers of any utility type – municipal, co-operative, and investor-owned.	Not applicable.

3.1 Further information on surveyed TREC systems

Australia

The Renewable Energy (Electricity) Act 2000, that supports the implementation of the Mandatory Renewable Energy Target (MRET), was passed by the Australian Federal Parliament in December 2000. Under this measure, tradeable Renewable Energy Certificates (RECs) will be used to demonstrate compliance with the objective.

The renewable power percentage target is announced annually (on March 31 of that year at the latest) as a percentage of total electricity sales. The percentage determines the number of certificates that liable entities must surrender each year to comply. Electricity wholesalers must redeem certificates on February 14 the following year.

15. Net metering allow end-users to sell electricity that they generate back to the grid. The quantity sold to the grid is deducted from the end-user's electricity bill.

A registry has been created to generate the certificates from generators or market players with extra power from renewable energy sources.

Each REC displays information on the plant at which the energy was produced. Some brokers are offering the possibility to select RECs based on the underlying energy technology.

Austria

The implementing legislation of the Energy Liberalisation Act, amended in 2000, provides for operators of hydropower plants with a maximum capacity of up to 10 MW (small-scale hydropower plants) to issue small-scale hydropower certificates.

Small-scale hydropower certificates shall refer to units of 100 kWh or multiples thereof. The operator of the grid into which electricity is supplied issues certificates. The grid operator shall keep a record of these certificates.

The certificate scheme has been set up with the following characteristics: quota equal to 8% for suppliers registered in Austria, based on their deliveries to end users, otherwise the obligation rests on the end users. The requirement to provide certificates applies to generators; a "balance fee" similar to one for renewable electricity excluding hydropower is applied.

Following the Energy Liberalisation Act Framework (2000), each of the nine Austrian provinces has to elaborate an implementation act. The Austrian programme will be monitored through an electronic database run by the regulator. There will be no physical paper certificates. In the longer term, RE certificates could be exchanged on the Austrian Electricity exchange platform (in 2002 at the earliest).

The framework legislation for the Small-Scale Hydropower Certificates scheme became effective 1 October 2001 when the full liberalisation of the electricity market came into force. Implementation at the level of Provinces should start 1 January 2002.

Under the Energy Liberalisation Act, distribution system operators are also obliged to purchase electricity from renewables (excluding hydropower) for a certain share of their electricity sales derived from these renewables. The amount of energy purchased from RE plants shall be equivalent to the following percentages of the amount of electricity delivered to the final customers:[16] at least 1% as of 1 October 2001; 2% as of 1 October 2003; 3% as of 1 October 2005; 4% as of 1 October 2007. Additional costs for the network operator are compensated. Non-compliance is to be sanctioned by a penalty.

The implementing legislation requires operators of recognised RE power plants to issue certificates stating the amounts of RE energy supplied. In instances where the energy is not purchased by the grid operator, the certificate must be endorsed by this grid operator in order to be valid.

Belgium

The Federal Law on the organisation of the electricity market creates the possibility for market instruments to supply a minimum of renewable electricity at a minimum price. More recently, the Federal Plan for Sustainable Development indicates that Belgium will introduce a RE certificates

16. Those connected to the distribution grid in the previous year.

system. The three regions have already taken measures to that aim, although details of the systems seem to vary already.

Flanders introduced the first renewable electricity certificate system in the country (approved by the Flemish Parliament on June 5, 2000. The European Commission approved the proposed system on 18 July 2001). Flanders considers this policy as a means to develop *domestic* resources and therefore obliges entities to meet RE objectives through the purchase of Flemish certificates only. It does not exclude the possibility to import RE certificates from other Belgian regions (Brussels and Wallonia) and countries, but the latter in particular can only be used to meet *additional* renewable energy requirements, on top of the Flemish ones.

Wallonia adopted a similar obligation, although it varies at least in two significant elements (Renewable Energy Report, 2001). First, the penalties for non-compliance differ slightly (see table 1). Second, electricity from "high-quality" co-generation is eligible in Wallonia (1 for 3 MWh of generated electricity) but not in Flanders.

Both systems co-exist with the federal "Green franc" system, which provides a subsidy of €0.025/kWh for renewable, bio-gas and biomass electricity sources. The system is to be reviewed in 2003. Wind and hydro facilities of 10 MW or less can also receive a €0.025/kWh subsidy. Photo-voltaic installations are also subsidised for the entire duration of their operation[17].

Denmark

The Danish law promoting RE energy consumption entered into force in 2000, but the introduction of the RE quota system has been postponed in 2001. Once implemented, renewable energy obligations apply to all electricity consumers, but they can turn to their electricity suppliers to meet this obligation for them.[18] The certificates can also be used for "green pricing", whereby consumers ask to purchase renewable energy *beyond* the mandatory quota.

The proposed Danish scheme establishes a framework for the certificates' price fluctuation. The certificates have a minimum price of DKK 0.10 per kWh as well as a maximum price. Consumers that do not fulfil their quotas would pay a tax of DKK 0.27 per kWh.

Turbine owners signing contracts for new installations starting 1/1/2000 and ending on 31/12/2002 would receive a fixed minimum price (DKK 0.33) and a RE certificate with a dictated minimum of DKK 0.1 and maximum DKK 0.27 for the first ten years. At a maximum, RE generators would obtain the equivalent of the fixed price applied earlier (DKK 0.6 per kWh).

The borrowing of RE certificates – the use of certificates that would be generated in the future - to fulfil the quota is partly authorised, a measure that could reduce the risk of market disruption due to unexpectedly low regime in a given year. The quota fulfilment is calculated on 1 April in the subsequent year. Hence, RE certificates from the first quarter of the coming year can be used for compliance.

17. International Energy Agency (2001): *Belgium 2001 Review – Energy Policies of IEA Countries*. IEA/OECD, Paris.

18. The complexity of maintaining a registry of certificates holdings for all electricity end-users is said to have been a barrier to the acceptability of the system by the industry.

Certificates would not be issued to all existing RE sources. Wind plants installed before 2000 will continue to receive a fixed feed-in price and a subsidy, and will access the REC system after ten years of operation (Eurelectric, 2000).

Italy

Italy undertook to reform its electricity market in 1999 and introduced a renewable energy obligation at the same time. This was motivated by the need to reform the previous mechanism to support renewable energy that had turned out to be overly costly and encouraged mostly the construction of co-generation plants (Eurelectric, 2000).

Italy choose to put the obligation on producers and importers of electricity in order to minimise administration cost, whereas all other countries have adopted a system on electricity distribution companies.

Netherlands

The Electricity Law of 1998 and the proposed Gas Act of 2000 include the possibility for the government to implement a system of RE certificates.

In January 1998, the Dutch energy supplying companies, united in EnergieNed, have voluntarily introduced the Green Label system to establish a market for renewable electricity. The certificate system was to achieve a target of 1.7 billion kWh electricity produced by renewable energy sources in 2000.

A new Green Certificates scheme started on July 2001. The system operates on a voluntary basis. No minimum share of renewables in energy consumption has been set. The idea is that voluntary demand will be enough to ensure a market for RE certificates. Since July 2001, 14 different parties have traded Green Certificates for a total amount of 600 MWh[19].

The Dutch system allows for imports of RE certificates from other countries provided that: an independent authority in the exporting country certifies the renewable origin of the electricity, and the electricity is also consumed in the Netherlands[20].

Sweden

Following the recommendation from the government committee in March 2000 and the proposal presented by a national investigation in October 2001, Sweden is planning a RE certificates scheme to encourage electricity production from renewable energy sources. This 'quota' system is planned to replace the existing support for investment and operations of renewable electricity plants by 2003, and measures are being considered to facilitate the transition between systems. The proposed obligation would be on end consumers. The RE quota should be operational from 1 January 2003.

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19. As per information from Mr. Vorrink at Groencertificatenbeheer (October 2001).

20. In principle, there is no way to distinguish between renewable electricity and electricity from other sources. In this case, the exporting entity must show evidence that enough transport capacity has been contracted on the interconnector between the Netherlands and the country of origin.

Details about the proposals from the Governmental Committee were presented in October 2001. Liable entities are end-users or their suppliers if they so choose. Electricity-intensive companies will be temporarily exempted, as their fiscal status is being assessed by a Tax Reduction Commission. A floor price for certificates will run from 2003 to 2008, going from 60 SEK/MWh to 0 by the end of this period. Wind farms which started operation before 1 January 2003 will be granted support until they achieve 25 000 full-load hours of operation, from 150 SEK/MWh in 2003 to 30 SEK/MWh in 2007, which should assure reasonable pay-back times for such investment.

The Swedish Parliament should make a decision on the Quota system in June 2002.

United Kingdom

The United Kingdom, England and Wales, on one hand, and Scotland on the other, are currently elaborating their respective systems for certificate trading. In the UK, a proposal for a Renewables Obligation has been open for discussion by interested Parties, the outcome of which was released in March 2001. A Statutory Consultation, including a draft Order, was issued in August 2000. Scotland is expected to issue a similar proposal shortly. The RO will come into force on the first day of the month following the Order. The details of the Renewables Obligation proposal (England and Wales) are shown in Table 1.

The UK had adopted an ambitious system to support renewable energy sources through 5 consecutive tendering processes: the Non-Fossil Fuel Obligation. These plants are subject to special conditions in order to be eligible to the RO.

Proposals were received asking for the RO to be "banded": different buyout prices would be set for different sources of renewable energy. The government saw this as too rigid an approach, and going against the market-led basis of the Obligation. Co-firing stations – burning both biomass and fossil fuels – are also eligible for the portion of electricity generated from biomass. Certificates from co-firing stations must account for less than 25% of the RO for any individual supplier. From 2006 on, the fuel used in co-firing stations must include 75% of biomass to remain eligible.

The New Electricity Trading Arrangements (NETA) that came into operation in March 2001 had an impact on market prices and triggered a 20-25% drop in whole sale electricity price. Renewable generators like wind farms have been particularly hit as NETA penalises producers which cannot predict their power output.

United States

In the United States, about ten different States have adopted or are in the process of adopting Renewables Portfolio Standards (RPS). RPS are similar to TREC: they require that a certain percentage of a utility's overall or new generating capacity or energy sales must be derived from renewable resources. A number of RPS do not use renewable electricity certificates to track and monitor compliance, but instead track power purchase contracts, at least initially. Only Texas at present has a REC scheme that involves a REC registry, though Wisconsin also has a modest certificate system. The Texas system allows trade, but only renewables within Texas can qualify under their RPS. Massachusetts[21], Connecticut[22] and Maine[23] will likely use a REC system when the system is put in

21. Massachusetts introduced outlines for a renewable portfolio standard as part of its electric utility restructuring legislation enacted in 1997. The details of the programme are being developed.

place in New England - expected in 2002[24]. In New Jersey, the interim draft rule for implementing the RPS provides for renewable electricity certificate trading. Other States that have adopted RPS include Arizona[25], Hawaii[26], Illinois[27], Nevada[28], New Mexico[29] and Pennsylvania[30].

New Jersey

New Jersey's restructuring legislation requires all retail electric suppliers to provide 0.5% of their energy from Class I renewable resources[31] by January 1, 2001, 1% by January 1, 2006 and an additional 0.5% per year until it reaches 4% by 2012. The renewable portfolio standard (RPS) began at 2.5% in 2000, a requirement that could be met by Class I or Class II Renewables. However, all subsequent additions must be met with Class I renewables.

In the New Jersey Board of Public Utilities' interim draft rule for implementing the RPS, electric power suppliers may satisfy the requirements by participating in a renewable energy trading programme. The programme also addresses reporting requirements, third-party certification, record keeping, and penalties for non-compliance.

Texas

In 1999, the Public Utility Commission of Texas issued the Renewable Energy Mandate Rule. This standard establishes the state's renewable portfolio standard, a renewable energy credits trading programme, and defines the renewable energy purchase requirements for competitive retailers in Texas. The standard calls for 2,000 MW of new renewables to be installed in Texas by 2009, in addition to the 880 MW of existing renewables generation. Qualifying renewable energy sources include solar, wind, geothermal, hydroelectric, wave or tidal energy, or biomass or biomass-based waste products, including landfill gas. Qualifying systems are those installed after September 1999. The RPS applies to all retail energy providers including municipal and co-operative utilities.

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22. Connecticut's 1998 electric utility restructuring law created a renewable portfolio standard that was revised in 1999.

23. The State of Maine Public Utility Commission adopted a Renewable Resource Portfolio Requirement rule on September 28, 1999 and became effective November 4, 1999.

24. As per information from Ryan Wiser at Lawrence Berkeley National Laboratory.

25. Arizona's Environmental Portfolio Standard (EPS) was formally approved in May of 2000 and became operational on March 30, 2001.

26. The legislation enacted in June 2001 in Hawaii establishes a renewables portfolio standard goal for each electric utility company.

27. A legislation signed in June 2001 in Illinois states as an explicit goal that at least 5% of the State's energy production and use be derived from renewable forms of energy by 2010 and at least 15% from renewable forms of energy by 2020.

28. As part of its 1997 restructuring legislation, the Nevada legislature established a renewable portfolio standard.

29. In 2000, the New Mexico issued a legislation calling for retail electric suppliers to meet 5% of their standard offer service with renewables.

30. Pennsylvania's Renewable Portfolio Standard was enacted in 2000.

31. Class I renewables include wind, solar, fuel cells, ocean energy, landfill methane and biomass. Class II renewables include hydro and waste-to-energy facilities that meet the highest environmental standards.

The Public Utility Commission of Texas is establishing a Renewable Energy Credits Trading Programme to start July 1, 2001, to continue through 2019. A Renewable Energy Credit (REC) represents one megawatt-hour (MWh) of qualified renewable energy that is generated and metered in Texas. A Capacity Conversion Factor (CCF) will be used to convert MW goals into MWh requirements for each retailer in the competitive market. The CCF will be administratively set and equal to 35% for the first two compliance years, thereafter based on the actual performance of the resources in the credits trading program.

Each retailer in Texas will be allocated a share of the mandate based on that retailer's pro rata share of state-wide retail energy sales. The programme administrator will maintain a REC account for programme participants to track the production, sale, transfer, purchase, and retirement of RECs. Credits can be banked for 3 years, and all renewable additions have a minimum of 10 years of credits to recover over-market costs. A penalty system has been established for providers that do not meet the RPS requirements. The penalty is the lesser of $50 per MWh or 200% of the average cost of credits traded during the year.

Wisconsin

The Wisconsin renewable portfolio standard became effective October 27, 1999 making Wisconsin the first State to have a RPS in advance of retail competition. The schedule of the percentages required and compliance dates are as indicated in Table 1.

Qualifying renewables include fuel cells that use a renewable fuel, tidal or wave action, solar thermal electric or photo-voltaic energy, wind power, geothermal technology, biomass, and hydro power (less than 60 MW). A credit-trading programme has been established such that electric service providers may sell to other electric providers renewable credits for any renewable energy in excess of the percentage specified for a given year. Credits can also be used in subsequent years. Violation of the RPS or misleading certification of renewable resources can lead to penalties up to $500,000.

3.2 A variety of implementation options for a similar goal

There are many similarities in the design of the new TREC systems especially in the basic definition of the RE objectives, even if percentage levels differ. The national TREC systems also show differences that reflect:

- the domestic policy goals underlying the introduction of these systems;
- the history of policy support measures for renewable energy and other environmental objectives in the country.

The TREC systems must accommodate two different objectives: the least-cost promotion of renewable energy sources, but also the promotion of a diversity of sources with varying competitive advantages. This is why certain technologies continue to receive support in addition to the TREC system. More details are provided in the section "Other support to eligible renewable energy sources".

Liable entities

Most systems make electricity retailers responsible for meeting the objective assigned in the TREC systems. These companies market electricity to end-users but do not, in most cases, produce electricity; they will be responsible for redeeming certificates to comply with their obligation. Because

they will be paying to acquire these certificates from RE generators, they will also be the primary players on the domestic TREC markets.

There are notable exceptions to this choice. In the case of Denmark, the obligation is on all electricity end-users although for practical reasons they can ask their electricity retailers to fulfil this obligation on their behalf. In the Green Certificates Scheme currently under discussion in Sweden, the electricity consumers could be designated as liable entities as well. In Italy, the obligation for compliance is placed on the producers. The Italian scheme puts the requirement on a relatively small number of large players: electricity generators and importers. This choice was made to minimise the cost of issuing and administering the systems. It isn't clear whether this implies a more – or less – concentrated market, with greater risk of market manipulation and less liquidity.

Eligible electricity sources

The TREC systems surveyed in this paper cover a wide range of renewable electricity sources, as well as other sources considered as environmentally friendly. When available, the following technologies are usually included:

- wind, onshore and offshore;
- tidal and wave energy;
- geothermal power;
- wood, agricultural products and energy crops;
- solar, PV and, in some cases, water heaters;
- "small" hydroelectric turbines, with a varying threshold size (20 MW in the UK, 15 MW in the Netherlands and 1,5 MW in Sweden);
- sewage gas, bio-gas and biomass.

Resources that are not systematically eligible include:

- large hydro-electric turbines (typically over 10 MW). While some countries exclude it, others consider new large hydro resources as eligible;
- incinerated waste. Italy accepts organic waste plants, under strict pollution control;
- combined heat-and-power, which is not a renewable energy technology but is considered a very efficient energy source and as such is included in the Walloon TREC system, albeit with a discount – but not in the neighbouring Flemish system – and a cap on its contribution to the annual target;
- co-firing, *i.e.*, power generation from both fossil and biomass fuels;
- pumping storage facilities, whereby hydro resources are pumped in a dam, using another energy source, and used when most profitable to do so. Italy excludes this technology explicitly, whereas it is eligible source in Australia, but *net* of fossil fuel input to the electricity used for pumping.

Certificates generally carry information on the underlying energy source. It is not clear whether liable entities would seek to distinguish certificates on that basis and therefore rely more on certain types of certificates to comply than others. Because certificates are issued after generation of the underlying power, each certificate released on the market is valid for compliance. All should therefore be fungible when it comes to complying with the set goals.

The nature of the fuel and the energy technology are not the only criteria for eligibility. First, there is usually a threshold size under which facilities will not be considered as eligible (1 MW in the two Belgian systems), probably to minimise the administration cost of the system. Second, the TREC system applies, in some cases, exclusively to *new* facilities – and targets are defined accordingly. In

Australia, a certificate is issued only for power generation above a 1997 production baseline: certificates apply only to additional RE produced since that year.

Last, some countries will not issue certificates to facilities that have benefited from other forms of support in the past, and to facilities beyond a certain lifetime, to avoid the distribution of windfall profits to these plants.

Penalties, banking and borrowing

Legal entities face penalties when they do not redeem enough certificates to cover their annual RE targets. There is no homogeneity in the penalty levels reported to date. The revenues are recycled to suppliers that meet their targets in the United Kingdom proposal and in the Renewable Energy Fund in Belgium (Flanders). These penalties also play the role of "safety valves", as they represent a ceiling price for traded certificates. This somewhat reduces the risk of a high cost attached to quantity-based policy instruments, in absence of perfect information on marginal cost curves for RE generation. The Italian system is an exception and has no maximum price. The government would issue additional certificates if cost is deemed too high. Some TREC systems also set a minimum price for certificates (Denmark and Sweden) where a decreasing price guarantee (floor) is to be introduced during the initial five years.

In some countries, liable entities also have the option to borrow certificates from their next annual commitment, *i.e.,* they commit to make up the shortfall in the future, but this option is limited to avoid the risk of postponing compliance with targets indefinitely. After discussion, the United Kingdom decided to exclude borrowing from its proposal. Borrowing is allowed for up to 10% of the liable entities' objective in Australia. While borrowing does create a risk of non-compliance, Menanteau *et al.* (2001) also highlight that borrowing can help to reduce price volatility in such markets.[32]

Liable entities that have over-reached their RE targets have the possibility to "bank" the surplus certificates for use in the following period. Banking is another means to avoid price volatility. Instead of selling unused TRECs at a time where supply is in excess and the price is declining, a generator could set these aside and wait for a more profitable price environment – and contribute to a lesser price increase then! Banking is limited to one year and 25% of the obligation in the UK, 5 years in Flanders and unlimited in Australia.

Other support to eligible renewable energy sources

Most TREC systems were established on top of, or next to, existing support schemes for renewable energy sources: feed-in prices for power generated from these sources – with rates adjusted according to the technology –, tendering procedures for RE facilities, and other forms of subsidies, including reduced tax rates and tax exemptions.

Some countries have maintained existing support policies – *e.g.* the Green Franc system in Belgium, to be reviewed in 2003. In some cases, these policies are targeted to RE sources that are unlikely to be competitive under the TREC regime, such as offshore wind in the case of the UK. Some governments have made the decisions to exclude sources that received other forms of support from

32. The comparison of tendering procedures and feed-in tariffs has clearly shown the importance of a stable price environment for the development of renewable energy sources (Menanteau, 2001).

the TREC system, while Denmark has created a mix of feed-in tariffs and a price band for certificates. This is a means to secure a certain level of revenue for generators, in the absence of a clear and stable price for TREC.[33]

In the Netherlands, each Green Certificate represents a certificate of authenticity on the basis of which the Inland Revenue may grant the supplier exemption from Regulatory Energy Tax, an environmental tax on energy use. Similarly, generators of RE electricity in the UK are exempted from the Climate Change Levy.

Certain TREC systems also contain sub-targets and requirements for certain energy sources. For instance in Wallonia, the contribution of CHP is limited to a certain share of the overall target, in order to assure that more advanced RE sources are also encouraged.

If left alone, the TREC system encourages the most cost-effective sources, leaving out certain technology options that may have a large potential but are not yet competitive with other renewable technologies (the case of offshore wind and biomass energy in the UK). Systems are therefore supplemented by other forms of government support. Absent a proper internalisation of external effects that RE technologies help mitigate, it may make sense to support such technologies to avoid locking them out of the market. But the remaining subsidies should be scrutinised as closely as the ones that were replaced by TREC systems for economic reasons[34]. In that sense, the goal of improved efficiency is somewhat undermined, even if, in a dynamic sense, there is value in assuring that technologies that are currently expensive can grow and increase the potential of sustainable energy sources in the future.

Two main elements emerge as justifications for the additional support to RE in countries that have adopted TREC systems:

- A form of burden-sharing, whereby the government, through lower tax rates, tax exemptions and an occasional premium on the RE electricity price, avoids passing all the additional cost of the TREC obligation to liable entities;
- Certain forms of RE – the contribution of which are generally small – are unlikely to be competitive if left to compete under the TREC system and may warrant direct support help reduce their costs to more acceptable levels in the long run.

In the end, RE policy does not generally seek to develop a single, least-cost technology, but rather to secure a portfolio of present and future technologies. A TREC regime that would exclude – "lock-out" – technologies with a longer-term potential would not be productive in that sense. With this broad policy goal, complementary support seems justified.

3.3 In summary: a step into the unknown?

Very few TREC systems have started operating in full, and there is still limited information about the price dynamics in different countries. Some features start emerging nevertheless:

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33. It is projected that Denmark will meet the agreed target of 20% from RE sources two years in advance, which should bring the price of certificates down to the floor price (Kjaer, 2001).

34. Schaeffer *et al.* (1999) analyse the effects of combining several support systems, including TREC.

- Targets are usually set as percentages of total electricity output over one-year, the total required output is not known until the end of the target period;
- Certificates are issued once the corresponding RE power has been produced;
- Some core technologies (wind, solar, some biomass, wave, small hydro, geothermal) are eligible in all systems;
- Promising technologies that remain relatively expensive are still supported by governments, outside the TREC systems.

These TREC systems depart from the textbook illustration of tradeable permits for environmental policy, largely because what they seek to promote is not exactly the internalisation of an externality – or, even, the reduction in cost to achieve a certain environmental goal – but technological development at least-cost. In a sense, TREC would mean the "kiss of death" for certain technologies that are not yet, but could become, cost-competitive in a decade and bring sizeable environmental gains as well. Supporting these technologies outside the TREC system may not be rational in static terms, but is coherent with efforts toward a more sustainable energy future.

In all, TREC systems capture a large share of RE sources and therefore remain faithful to their original market development goal. Further, the certification process provided by TREC systems make it possible to market other RE sources that may be less competitive, but are attractive for consumers. These could be sold at a premium to end-users willing to carry the extra cost – and do so outside the RE obligation.

A couple of issues should be closely monitored as these markets evolve:

- Will RE generators and the RE sector more generally be able to handle the transition from what has been, in some countries, a fairly protected market towards a less predictable one, especially when it comes to the payment of the RE premium?
- Will liable entities – but also electricity consumers – distinguish between sources of RE when they acquire certificates?

In light of the differences in the systems – target stringency, other support policies for RE, eligible RE sources and penalties for non-compliance – the prices of TREC will differ from one country to the next. This creates opportunities for further cost reductions, provided these systems can be linked.

The experience gained over the coming years will be valuable and can be expected to cast a new light on these issues. The Australian Renewable Energy Certificates created as of 1st April 2001, Belgian regional RE Certificates programmes under development and the United Kingdom Renewables Obligation scheme which comes into force in January 2002 will be valuable test cases.

4. From domestic to international TREC systems

Tradeable permit systems, as they seek to reduce the cost of achieving a given goal, gain from being open to a diverse set of participants with different marginal costs across the system. This principle should also apply to TREC systems, even if the underlying objective – promoting the development of renewable energy sources – makes it a special case. The recently agreed EU Directive

on the "Promotion of electricity from renewable energy sources"[35] has brought the issue of international trading of TREC to the forefront.

This Directive provides a set of *indicative* RE targets to be reached by Member States in 2010, with an overall goal of 22.1% of RE electricity. This overall target is declined as follows for the fifteen Member States:

Table 3. Target shares of renewables in electricity generation by 2010 for EU countries (%)

Country	Target share	Country	Target Share
Austria	78	Italy	25
Belgium	6	Luxembourg	5.7
Denmark	29	Netherlands	9
Finland	31	Portugal	39
France	21	Spain	29.4
Germany	12.5	Sweden	60
Greece	20.1	United Kingdom	10
Ireland	13.2	EU	22.1

Source: Enerpresse, No 7719, 7 December 2000.

Options to implement these objectives are left open to individual governments – feed-in tariff, quotas, TREC, etc. At the international level however, one of the key element of the Directive is the obligation to issue guarantee certificates for the generated renewable energy, which opens up the possibility to trade these certificates across countries. Moreover, in an amendment from the European Parliament, it is proposed that "Electricity generated from renewable sources traded across national borders shall, however, only count for the achievement of the aforementioned targets in the Member State where it is consumed"[36]. The Directive specifies that this arrangement requires that a Union-wide certification of all electricity, as well as labelling be put in place. Indeed, consumed electricity cannot be tracked to its generating plant unless a certificate is sold by the generator to the electricity retailer. Hence, the Directive creates the building blocks of what could become an EU-wide TREC system[37]. The search for a least-cost strategy seems to encourage the elaboration of such a system – even if the countries' targets are not mandatory[38].

As the above tables show, a number of countries have considered the possibility of relying on imported TREC to comply with domestic objectives. Their viewpoint on this question highlights some of the barriers standing in the way of an international TREC system. The next section lays out some of the benefits of an international TREC system and the following discusses barriers to such progress.

35. Voted by the European Parliament on 3 July 2001, formally adopted by the EU Council of Ministers on 27 September 2001[Directive 2001/77/EC].

36. Amendment 32, (2.b), Official Journal of the European Communities (2001): 4. Promotion of electricity from renewable energy sources. C 223/302, 8.8.2001

37. An international association, RECS, is also actively promoting international trading in green certificates. www.recs.org

38. At a symposium held at the IEA earlier this year, stakeholders in renewable energy and TREC systems supported the idea of a global TREC system, and stated that "the possibility of international trade of certificates, beyond that already contemplated in Europe, may be attractive in certain circumstances, and deserves additional investigation" IEA/CERT/RE/WP/(01)7.

4.1 Arguments in favour of linking domestic TREC systems

Since TREC systems seek to exploit the least-cost RE resources available, one expects that a broader geographical basis will allow more efficiency gains, as marginal costs are likely to differ across countries with domestic regimes. For instance, Germany has tapped a large portion of its wind potential, whereas the UK's is still undeveloped.[39] Because of their industrial and agricultural activities, certain countries have important biomass resources. Small hydro resources are also unequally distributed among countries.

In addition, a larger number of participants with a variety of energy sources would contribute to minimise the volatility of TREC prices,[40] increase the liquidity of the market as a larger number of generators could respond to the international price signal. It would also reduce the risks of market manipulation that arises when only a few participants own a large share of any given market. Some of the market volatility may come from inadequate wind regimes in Northern Europe, although such phenomena may not affect significantly the volume of electricity from wind resources on an annual basis.

4.2 Barriers to international trades

Some countries have expressed concerns about allowing foreign certificates to be used for compliance in their own systems. Because governments seek to develop their domestic RE resource basis, they may not welcome the possibility that their systems is used to promote the development or RE sources outside their territory, at the expense of their tax-payers, to the benefit of foreign power generators.

Here are additional factors that explain why policy-makers may be reluctant to allow the imports of TREC:

- An imported TREC reduces the local benefits originally sought by the setting of a domestic RE target. Indeed, the imported TRECs are a proof of reduced externalities elsewhere – and not necessarily with a social benefit equivalent to that which would have occurred, had the RE been generated in the importing country.[41] Of course, this is also a matter of perception: an end-user may be purchasing power from a utility that complies with its TREC obligation and not directly benefit from the amenities of RE power, if it is produced in another location in the country...
- If TREC systems are to be instrumental in reducing a country's GHG emissions, fostering foreign development of RE, while profitable to the global environment, does not reduce the importing country's GHG emissions. Further arrangements would need to be elaborated to assure that a country could import the tradeable GHG emission permit together with the imported TREC. This issue is developed in the following section.

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39. Menanteau *et al.* (2001)

40. Menanteau *et al.* (2001)

41. "Given that many of the benefits derived from renewable generation [...] are local in nature [...], the question can be asked: Why would someone in another region or country want to contribute to the development of renewable generating facilities that benefit someone else?" Hamrin (2001).

    On a smaller scale, but for similar reasons, green electricity marketers in the United States note that it is difficult to sell a certificate in New York State when the underlying generating unit is located in Oregon.

Another barrier to international transactions is the very different starting point of countries with respect to renewable energy support policy. There is little homogeneity in the level of subsidies that RE technologies have received in the past and in the measures that complement TREC in some instances.

A subsidised source could generate certificates that would be cheaper than certificates in a country where such subsidy does not exist. Opening the two countries' systems would lead to a distortion of competition. The subsidy in the exporting country would benefit to the liable entities in the importing country, at the expense of the budget in the exporting country. As Schaeffer et al put it: "... the taxpayers of country A [exporting certificates] are subsidising not only the price of the certificates in their country but in all countries participating in the [TREC] system".[42]

Aware of this potential problem, some countries have indicated that they need a guarantee that imported TREC do not benefit from other forms of support at home.

Some other possible differences in TREC systems may complicate international transactions:

- Eligible sources. This is not an insuperable obstacle, as certificates usually carry information about the underlying energy source. Liable entities seeking to import certificates from another country would need to specify the list of eligible sources in its domestic system. The installation date of the facility is also a criterion for most of these systems and starting dates do differ. Again, such information could be included in certificates;
- Penalty levels (buyout rates). In theory, an international regime in which penalty rates differ from one country to the next should encourage non-compliance in the country with the lowest penalty level. A liable entity could transfer TREC to another such entity in another country at a price higher than the penalty, pay the penalty and make a profit from the transaction;
- Absence of mandatory RE requirements. Some generators in country A may offer RE certificates that correspond to sources that are eligible in country B, but without being subject to a domestic RE goal. These sources would have been in operation with or without the incentive provided by the certificate system and the transaction represents a straight windfall profit for existing generation. To avoid this problem, the United Kingdom foresees the possibility to open its domestic system to foreign sources *only if* they are subject to a similar objective at home. With such an approach, only certificates that are surplus to the domestic requirement are sellable to another country.

Goossens (2001) looks into the possibility to trade certificates from any European country with a TREC system into another. Her analysis confirms that discrepancies in the adopted regimes could prevent certain bilateral transactions: the denomination of TRECs (100 MWh is the minimum size in Italy), the threshold of eligible hydro-resources (10 MW, 20 MW, unlimited), and possible market distortions from additional support systems that only apply in some countries.

4.3 In summary

We have laid out a list of arguments in favour of an international system but also likely obstacles that show that such endeavour may not be straightforward. Already, governments that have introduced TREC system expect a fair access to other countries' TREC markets before they agree to open their own:

42. Schaeffer et al (1999) provide a useful illustration of this issue, page 50.

- an equivalent obligation on the generation sector;
- the absence of subsidies that would threaten to distort competition;
- similar eligibility requirements, not to mention the assurance that issued certificates are accurate.[43]

On that basis, governments may see value in broadening their systems, because this could ensure a more efficient and competitive trading regime and a lower volatility in TREC prices – one of the main disadvantages of TREC systems compared to fixed feed-in tariffs. What is needed is a political decision that the environmental and other attributes of certificates are as valuable regardless of the country or region where they occur. This may be more easily achievable after countries have developed their own RE basis to a satisfactory level, and other support systems have been phased out or limited to a level that would greatly reduce potential market distortions.

5. Interactions between TREC and GHG emission trading

Because renewable energy sources do not produce net CO_2 emissions, tradeable renewable energy certificates system, in promoting more renewable energy production, contribute to lower CO_2 emissions from what would have happened otherwise. This is one of the primary objectives of TREC systems in the countries covered here. The traded certificates will carry a certain value corresponding to this clear environmental benefit.[44] TREC systems, at domestic level – and possibly at international level – will operate alongside the broader greenhouse gas emission trading system introduced under the Kyoto Protocol. GHG emission trading is also being implemented in the UK and in Denmark, and studied closely by governments elsewhere. These GHG trading systems will also put a price on avoided GHG emissions, and in particular on avoided energy-related CO_2 emissions.

5.1 How would the two markets interact?

Morthorst (2001) analysed the interaction between TREC and GHG emission reduction objectives, especially when these are implemented with GHG trading, asking the question: How would these two – separate – markets affect each other? His results show that GHG reduction objectives would affect TREC prices differently, depending on how GHG reductions are implemented. Three options are considered: no trading; trading based on grandfathering; trading based on auctioned GHG permits. In the absence of a trading system – or, presumably, an equivalent carbon tax – the electricity price would not accurately reflect the marginal cost of reducing GHG emissions in the country. As a result, the difference between the spot electricity price and marginal cost of renewable electricity – the price of the certificate – is higher than the economically efficient level. Certificates generated in that context are unlikely to compete on the international market, which results in an inefficient allocation of effort.

At the opposite end of the spectrum, if power generators had to bid for GHG permits, the electricity price would fully reflect the value of avoided CO_2 and would reduce the price of TREC. The certificate price would only represent the extra cost of meeting the RE obligation, *i.e.*, the non-CO_2 and non-electricity value of renewable energy generation. A GHG permit system based on grand-fathering would not fare as well as a system based on bidding, but would be superior to the first system.

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43. The RECS organisation has focussed its first efforts on drafting the basic commitment for the so-called issuing bodies, *i.e.*, organisations in charge of issuing the TREC and redeeming the TREC that it has issued. http://www.recs.org

44. And indeed in the Netherlands and the UK, the production of TRECs lead to an exemption from the Energy Regulatory Tax (an eco-tax) and the Climate Change Levy, respectively.

In brief, Morthorst (2001) argues that in order for an international TREC system to best achieve its objective – the least-cost promotion of renewable energy – in presence of a CO_2 constraint, it is best for GHG trading to be implemented with a bidding system for GHG permits.

Amundsen and Mortensen (2001) studied the specific case of Denmark, where a CO_2 trading regime and a TREC system will co-exist starting 2003. They find that under a scenario where electricity can be traded freely from and to Denmark, a harsher constraint on CO_2 emissions in Denmark would leave the profit of RE generators unaffected, as both the wholesale electricity price and the TREC price would remain unaffected. This assumes that the international electricity market determines the electricity price and the Danish obligation on RE defines the TREC price. This conclusion is surprising: common sense would argue that the more demanding CO_2 emission reductions are, the more competitive renewable energy would become. The existence of a TREC system somewhat changes this intuition in the narrow setting of a small power market with an electricity price set by international conditions.

5.2 Can renewable energy certificates be traded for GHG emission permits?

It has also been argued that a TREC could lead to the issuance of CO_2 or greenhouse gas (GHG) emission permits, with a conversion factor to be agreed. The opposite is less obvious, as a GHG permit does not necessarily come with the same local environmental benefits as ones embodied in TRECs[45]. In this section, we choose to focus on the conversion of TRECs into GHG permits.

A fundamental difference between the two systems could prove difficult to overcome: the nature of the permits in both trading regimes. Tradeable renewable energy certificates are based on averaging: a percentage of total electricity output must be matched with TRECs. The total number of certificates can vary from the originally intended target. In most cases, GHG tradeable permits are based on absolute quantity caps. More simply: a company can comply with its TREC obligation but still augment its fossil fuel-based generation and its CO_2 emissions. We discuss this issue in more detail below. We first start with a description of how TREC could be transformed into GHG permits in a domestic setting.

Converting renewable energy certificates into GHG permits

A TREC represents a proof that renewable energy has been used to generate, say, 1 MWh of power. The new RE plant displaces generation from another plant, which represents the avoided carbon from RE-based electricity. Finding which plants or power generation technologies are being displaced by RE is not necessarily complicated, but would be cumbersome, as it would require doing so every time a TREC is issued. A TREC issued at the peak of the electricity load curve would therefore not carry the same avoided carbon as a TREC issued "off-peak".

A less accurate but considerably easier approach is to determine the average CO_2 content of one MWh in the country. It is a simple ratio of total CO_2 emissions from power generation divided by total electricity output (roughly 1 tonne of CO_2 per MWh in Australia, 0.44 tonne in the UK, etc.)[46].

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45. See Calder and Hough (2001) for a detailed discussion of this question in the UK context, where both an emission trading system and a renewable obligation are being introduced.

46. Calder and Hough (2001), Sonneborn (2001)

If we assume a conversion factor of 0.5 tCO_2 /MWh, a liable entity with 4 unused TRECs of 1MWh each could sell 2 tCO_2 to a liable entity in the GHG trading system. The corresponding TREC would need to be cancelled in the system's registry. These are rather trivial operations.

An illustration

In the negotiation of the United Kingdom's emission trading systems, lengthy discussions have taken place on the possibility to reconcile two different emission trading systems:

- One based on an absolute cap: liable entities cannot emit more than x t CO_2 per year;
- One based on a CO_2 intensity (or "averaging"): liable entities are not allowed to emit more than y tCO_2 per unit of output (the so-called "unit" sector).

An entity with a CO_2 intensity below its cap could generate an unexpectedly large number of CO_2 credits, if its underlying production levels are also higher than expected. If these credits were sold to an entity in the absolute cap sector, it would inflate the allowable emission level in that part of the system.

This potential for such problem has led to the introduction of a gateway between the two systems. Entities in the unit sector would only be allowed to sell permits to the entities with absolute caps once the regulator has confirmed that the unit sector's emissions are at an acceptable (low) level.

The fact that TREC systems are based on "averaging" and not "absolute" caps suggests that the same problem could arise between TREC and GHG permits. Electricity output could increase and along with it, related CO_2 emissions, even though the overall percentage share of renewable energy is maintained.

We now illustrate how the automatic conversion of TRECs could lead to an unintended increase in CO_2 emissions, taking a single country and two generators as an example. The assumption is that these generators are subject to both a CO_2 cap, and a renewable energy obligation.[47] The TREC obligation is for 10% of total output to be generated from RE sources. The CO_2 caps of the two generators and other information are shown in Table 4.a.

Table 4a. Power generators under a CO_2 cap and a RE obligation – Before target period

	Generator A	Generator B
Carbon-based electricity output	100 MWh	100 MWh
New RE output	0 MWh	0 MWh
Total electricity output	***100 MWh***	***100 MWh***
CO2 intensity of non-renewable generation	0.6 tCO2 per MWh	0.4 tCO2 per MWh
CO2 emissions	60 tCO2	40 tCO2
CO2 allowances (i.e. cap)	50 tCO2	36 tCO2
CO2 / TREC conversion rate in the country	***0.5 tCO2 per MWh***	

Generator A installs enough wind capacity to produce 20 MWh. It thus receives 20 one-MWh renewable energy certificates. It decides to substitute 10 MWh of its previous carbon-intensive generation with half of the newly-generated wind energy. Its total output at the end of the year is 110

47. For the sake of the illustration, we assume that power generators are liable both under the TREC system – as is the case in Italy – and under the GHG permit system.

MWh. It must therefore redeem TRECs for 11 MWh to comply with its objective. That leaves 9 MWh-worth of TRECs for sale or banking. Table 4.b illustrates generator A's new situation.

Generator B produces 10 MWh of biomass-based electricity, displaces 10 MWh of its previous carbon-based generation – hence complying with both its RE obligation and its CO_2 cap. Its status at the end of the target period is shown in Table 4.b.

Generator A is short of 4 t CO_2 to comply with its CO_2 cap but carries a surplus of 9 MWh-worth of TRECs. These certificates can be converted into the CO_2 trading system, as they have not been used for compliance with the RE objective – i.e. surplus TRECs are not counted twice in this example. Based on the conversion factor adopted at country level – which reflects the average carbon content of generation *ex ante* – it generates 9 times 0.5 = 4.5 tCO_2, enough to offset its excess emissions. Both generators have complied with the RE obligation and their CO_2 cap.

On the whole however, the original CO_2 goal assigned to the generation sector was 86 tCO_2 while its total emissions are now 90 tCO_2. The country as a whole must therefore acquire 4 tCO_2 from the international sector to make up for the excess emissions. This illustration shows that:

- the TREC obligation did contribute to lower CO_2 emissions – with the initial generating capacity, a similar volume of electricity would have released (66 + 40 = 106 tCO_2), to be compared with 90 tCO_2 here;
- but the automatic conversion of surplus TRECs into CO_2 permits allowed the target to be exceeded by 4 tCO_2.

Table 4b. Power generators under a CO_2 cap and a RE obligation – Target period

	Generator A	Generator B
Carbon-based electricity output	90 MWh	90 MWh
New RE output (1)	20 MWh	10 MWh
Total electricity output	***110 MWh***	***100 MWh***
TRECs: redeemed (10% of total output) (2)	11 MWh	10 MWh
TRECs: surplus (1) – (2)	9 MWh	0 MWh
CO_2 intensity of non-renewable generation	0.6 tCO_2 per MWh	0.4 tCO_2 per MWh
CO_2 emissions	54 tCO_2	36 tCO_2
CO_2 allowances	50 tCO_2	36 tCO_2
Compliance? CO_2 allowances – CO_2 emissions =	***- 4 tCO_2***	***0 tCO_2***
CO_2 allowances from surplus TRECs	9 * 0.5 = 4.5	
Adjusted CO_2 allowances – CO_2 emissions =	***+ 0.5 tCO_2***	

In all, the fact that RE obligations are expressed in average or intensity terms brings some uncertainty as to its *ex ante* contribution to avoided CO_2 emissions or, more precisely, to its contribution to the production of GHG permits in excess of the distributed allowances. The above illustration shows how the automatic conversion of TRECs into CO_2 emission allowances could inflate the number of such allowances and undermine a country's compliance.

We find other causes for caution in the possibility of trading TRECs at international level and converting them to CO_2 permits where it would be most profitable to do so. Let us assume that countries have determined their own conversion rates on the basis of their average – or marginal – electricity production technology. These rates would vary, as some countries rely extensively on fossil fuels (Australia) while others have a much lower carbon content of electricity (*e.g.*, France, Sweden). Let us assume a generator with 10 MWh of TREC, operating in a country where 1 MWh of renewable electricity displaces 0.2 tonne of CO_2.

The generator could either sell these TRECs to an electricity retailer that needs them for compliance, or transfer them into the domestic GHG trading system where it would obtain 2 tonnes of

CO_2 worth of permits. Or it could trade them into another country where a higher conversion rate applies, convert them into CO_2 at that rate and obtain a higher number of GHG permits. Although these 10 MWh of RE have only displaced 2 tonnes of CO_2 in their country of origin, they have been sold for more, allowing a higher level of CO_2 emissions overall. A rule that would force imported TRECs to be converted with their country of origin's conversion factor could be easily by-passed. The imported TREC could be used to comply with the RE obligation, which would free a local TREC that could then be converted at the more favourable "exchange rate".

If we push this further, governments willing to allow international trading of TRECs and the conversion from TRECs into GHG permits would need to agree on a universal conversion factor. Yet our above example has shown that even this technically-challenging solution would not guarantee that the allowed GHG emissions from power generation would not be exceeded.

5.3 Summary: interactions and compatibility

A broader renewable energy basis contributes to lower CO_2 emissions and TREC systems would therefore affect a market for GHG permits, even if RE are not usually considered as the primary option to achieve the GHG emission goals set under the Kyoto Protocol.[48] The co-existence of the two systems could lead to market distortions in the TREC system, *e.g.* if the marginal cost of avoided CO_2 emissions is not properly reflected in the market price of electricity. However, other analysis shows that the existence of an international electricity market may lead to a disconnection between the stringency of the CO_2 targets and the price level in the TREC regime.

Renewable energy avoids CO_2 emissions, but should TRECs be allowed to be imported into the GHG emission trading system, on the basis of a simple conversion factor? We have illustrated that this could, in some cases, undermine the environmental effectiveness of the GHG trading system and could further complicate the trading of TREC at international level.

6. Further research

A thorough assessment of tradeable renewable energy certificate systems must wait until actual trading has started and issues related to compliance have been studied from practical experience. In particular, the management of price volatility could become a crucial question for investors in renewable energy and affect the efficiency of TREC systems as a tool to promote renewable electricity sources. The simultaneous regulatory reform of the electricity industry and new market settings are also likely to affect RE generation.[49]

Linkages between domestic markets and interaction between GHG emission and TREC trading warrant further investigation. In particular, different types of entities would be liable in CO_2

48. In its latest proposal for the Renewable Obligation, the Department of Trade and Industry evaluates the cost of the obligation per avoided tonne of carbon to be more than GBP 300 (US$ 120 per tonne of CO_2).

49. A recent report by the British regulator Ofgem to the UK Department of Trade and Industry shows that RE generation – most of all its competitiveness – has been negatively affected by the New Electricity Trading Arrangements in the first two months following their launch (Ofgem, 2001). Wind energy, in particular, suffers from its lower level of predictability. There are nevertheless possibilities to "consolidate" such risk, although these have not yet materialised in the first two months.

and TREC systems – energy distributors in most cases for TREC, generators for CO_2. This could affect the market dynamics in these trading regimes but also electricity prices in a de-regulated market.

GLOSSARY

Co-generation	Combined generation of electricity and heat. Standard power plants release heat into the atmosphere. Co-generation plants use the heat for other industrial or domestic uses.
Distribution companies	Companies that own the transmission or distribution networks for conveying electricity from the generators to end use consumers.
Final Consumer	Customers buying electricity for their own use.
GHG	Greenhouse gases from human activities causing global climate change. CO_2 from fossil fuel energy is the primary gas among these.
Liable entity	Company that is responsible for meeting the assigned renewable energy objective. With the exception of Italy, liable entities are electricity retailers. Italy has put the burden on power generators and importers of electricity.
MWh	Unit of electricity output, representing one hour of generation by a plant of one MW of capacity.
Producer	Legal or natural person or a profit-making company generating electricity
RE	Renewable energy. In this paper, this abbreviation covers all forms of energy that are eligible in TREC systems – not all of which are strictly renewable, in some instances.
RPS	Renewables Portfolio Standard. A RPS requires electricity generators or suppliers to generate or supply a set percentage of their electricity from renewable energy sources. Percentage requirements, definition of renewable energy, and other specifications may very. Portfolio Standards most commonly refer to electric sales measured in megawatt-hours (MWh), as opposed to electric capacity measured in megawatts (MW).
TREC	Tradeable renewable energy certificate. Generic term used in this study to define the tradeable certificate corresponding to a renewable energy obligation, or the underlying system.
Utilities	Enterprises engaged in the generation, transmission and distribution of electricity to end-use customers.

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Chapter 5

THE USE OF TRANSFERABLE PERMITS IN THE TRANSPORT SECTOR

by

Charles Raux
Laboratoire d'Economie des Transports (CNRS), Lyon, France .

Executive Summary

This report focuses on the potential use of domestic transferable permit (TPs) systems in the transport sector, in order to address the issue of mobility needs management and especially the reductions of airborne pollution and greenhouse gas emissions. Firstly the context of the transport sector is briefly reviewed, the main arguments for or against the use of TPs in the sector are analysed and relevant areas are identified. Secondly four case studies of past, present or possible future permits systems are presented and evaluated. The main conclusions are: TPs applied to mobile sources are technically feasible at acceptable financial costs for protecting sensitive geographic areas. TPs schemes applied to automakers for unit vehicle emissions are also viable. Clarity, simplicity in target and pragmatism in scheme design help for their success. Regarding the broader GHG issue end-user TPs would currently involve significant administrative costs when compared with fuel tax system. Given the social resistance encountered by increase in fuel taxes in several countries, end-user TPs with free allocation may intrinsically have potential greater effectiveness and acceptance and should be thoroughly evaluated case-by-case as an alternative.

1. Introduction

"There is a clear understanding that we cannot provide for the world's continuing and growing needs for mobility by simply extending today's means of transportation."[1] Clearly today mobility is part of the whole sustainable development issue: that is to say how can we meet the needs of today without diminishing the capacity of future generations to meet theirs?

Maintaining a sustainable mobility means to continuously assess and drive efficiently its different aspects for now and the future: that is to say economic (e.g. achieve a better balance between demand and supply), financial (e.g. finance transport infrastructure and operation), environmental (e.g. avoid harmful health effects) and social aspects (e.g. maintain spatial and social cohesion).

There are already numerous studies about the way transport activity could be made more sustainable. ECMT (1997) has conducted a detailed review of the available policy options to reduce overall CO_2 emissions of the transport sector. With its Environmentally Sustainable Transport (EST)

1. WBCSD, 2001.

program, OECD has recently achieved a thorough analysis of possible scenarios to make transport environmentally sustainable for 2030 and elaborated related packages of measures (OECD, 2000*b*). The business side (World Business Council for Sustainable Development) is also undertaking a sustainable mobility initiative (WBCSD, 2001).

Within the wide range of policy instruments Transferable Permits (TPs) are currently attracting growing interest in OECD Member countries. In both theory and practice, the main strength of applying TPs has been for larger point sources for air and water pollution, while smaller or non-point sources have been better addressed through taxes. Given that transport sector is increasingly becoming the major source of air pollution as well as greenhouse gases, the possibility of applying TPs to non-point sources is worth exploring.

One strong conclusion of the studies on transport emissions reduction strategies is that these strategies should involve multiple technological, economic and societal aspects, thus meaning a *package* of instruments within a proper implementation. With this respect, the focus of this report is much more limited and is on the potential use of domestic transferable permit systems in the transport sector. It aims at answering the question of potentials of TPs to address the issue of mobility needs management and especially the reductions of airborne pollution and greenhouse gas emissions stemming from transport. The analysis will be conducted both on a general level and on the basis of a few case studies.

The first section reviews the context of the transport sector and, in particular, the environmental problems posed by transport activities. The main arguments a priori for or against the use of transferable permits are analysed and areas where they might be used within the transport sector are identified.

The second section presents four case studies of past, present or possible future systems and analyses their potential.

The third and final section concludes by drawing common lessons from these case studies and by identifying the avenues which remain to be explored.

2. The transportation sector context

The transportation sector, understood as embracing all infrastructure and services that allow for the movement of people and goods, is key to the economic and social functioning of modern societies.

These societies have only been able to grow because of accelerating economic and social interaction and exchange. The transportation system has grown apace with political transformations (the creation of unified nation States, the abolition of internal customs barriers, safer highway networks) and has benefited from technical and economic innovations: creation of highway, rail, air and river infrastructure, development and wide use of internal combustion engines. Transport has in fact been an essential vector of economic and social development, and will surely be called upon to continue as such.

In the course of achieving these general objectives, the transportation system is now subject to a number of major constraints, given the current state of its technologies: these constraints relate to concerns over energy dependence, the global environment, the local environment and quality of life, the shortage of available space in densely populated areas, and the ensuing congestion, and finally the scarcity of public funding. This paper will confine itself to the environmental aspects, while bearing in mind the other constraints that also impact on any policy for controlling the transportation system.

2.1 The global environment and the greenhouse effect

The global environmental constraint is becoming increasingly severe, and concern is rising, in particular, over the potential for global climate change caused by human activities. At the centre of these concerns is the greenhouse effect, with its associated gases (carbon dioxide CO_2, methane CH_4, nitrous oxide N_2O, and also tropospheric ozone[2] O_3 via its precursors, notably nitrogen oxides, carbon monoxide, CH4 and non-methane hydrocarbon) a great part of them resulting from the combustion of fossil fuels. To these we must add chlorofluorocarbons (CFC) released by air conditioners, particularly during the automobile wrecking process: these pollutants can persist for more than two centuries in the upper atmosphere. While the nature and scope of their effects are still the subject of debate, the stakes are such that, in accordance with the precautionary principle, ways should be found to reduce emissions of these gases, as well as the consumption of fossil fuels in the transportation sector.

Following the 1992 Rio conference (UNFCCC), which adopted the principle of an international agreement to reduce greenhouse gases emissions at least by half, industrialised countries (known as the "Annex I Countries" under the protocol) agreed in 1997 at Kyoto on a protocol for limiting emissions over a horizon extending to 2010, but with a more limited target of average –5% compared with 1990. Every Annex I country committed itself to the limitation target.

In most industrialised countries the transport sector is one of the most significant GHG emitter and particularly of CO_2, the main greenhouse gas associated to transport activity (for instance 34% of CO_2 in France[3] in 1999, 30% of CO_2 in the US in 1997). However the biggest threat is that the rate of growth in CO_2 emissions from the transport sector is projected to outstrip that for the other sectors in most of these countries, so the share of CO_2 emissions from the transport sector is expected to be even greater in the future. According to the OECD transport is the second highest growth sectors in terms of GHG emissions in the OECD area (following industrial processes), with the share of CO_2 emissions increasing from about 25% in 1995 to 30% in 2020 (OECD, 2000a). Within the European Union, it is expected in a recent study (Bates *et al*, 2001) that the growth from 1990 to 2010 in the CO_2 emissions from transport in the EU would be 35% in the baseline trend and 25% taking into account the full effect of the voluntary agreement with the car makers[4]. This result should be related to the Kyoto objective of –8% for the EU.

According to the OECD (1995), forecast traffic growth (in vehicle kilometres travelled, VKT) is such that current strategies for reducing unit vehicle emissions will be inadequate to reduce overall emissions. Only with more intensive application of a combination of technical solutions for reducing emissions, enhancing the energy efficiency of engines and slowing the growth of VKT will it be possible to reduce greenhouse emissions over the coming 30 to 40 years.

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2. However tropospheric ozone is not recognised by the UNFCCC as a greenhouse gas, since significant uncertainties remain in the understanding of its contribution to the climate change.

3. CITEPA, 2000

4. Known as the "ACEA-EU agreement" it is a voluntary agreement between EU and European (ACEA), and then Japanese (JAMA) and Korean (KAMA) car manufacturers to reduce the average CO_2 emissions of all new cars.

2.2 The local environment and its impact on daily life

The local environmental constraint relates to localised emissions of atmospheric pollutants, as well as to transportation noise and safety concerns.

Local and regional airborne pollutants

The principal atmospheric pollutants produced by automobile traffic (OECD, 1995, 2000*b*) are:

- Carbon monoxide (CO) which, as a haemoglobin oxidant, hinders oxygen transport in the body; the short term limits set by the WHO are often exceeded (in one-half of the world's cities, according to the United Nations).
- Volatile organic compounds (VOC), including benzene, 1:3 butadiene, formaldehyde, acetaldehyde and polynuclear aromatic hydrocarbons; these result from the incomplete combustion of fuels and from evaporation losses during the fuel distribution, storage and transport cycle. They are for the most part highly carcinogenic. According to the USEPA (the USA's Environmental Protection Agency), mobile sources may account for 54 percent to 58 percent of cancers associated with toxic air pollutants in the USA.
- Nitrogen oxides (NO_x), which are the second cause of acid rain after sulphur dioxide and are among the precursors of tropospheric ozone or smog, together with VOC; they provoke respiratory problems, particularly in children (coughs, rhinitis, sore throats) and increase the sensitivity of asthma sufferers to urban dust and pollen.
- Sulphur dioxide (SO_2), released primarily by diesel fuel, is a contributor to acid rain and also to respiratory irritation.
- Particulate matter (PM), which include dust, dirt, soot, smoke, and liquid droplets. They are emitted directly, for instance resulting from the incomplete combustion of diesel fuel, or formed by condensation or transformation of emitted gases such as SO_2, NO_x, and VOC into tiny droplets. Fine particles (PM_{10} of a diameter less than 10 microns) are the most readily inhalable. These substances also produce respiratory problems, and are probably carcinogenic. Gasoline releases fewer particles, but they are more highly toxic.
- Lead as a gasoline additive; this has harmful effects on health, even in extremely low concentrations (behavioural problems, difficulty in concentrating, low IQ); it is widely dispersed through exhaust emissions, but is now gradually disappearing in most OECD countries[5].
- Other pollutants, such as asbestos, use of which is to be reduced in any case (brake pads, clutches, automatic transmissions), toxic heavy metals, and dioxins, which are present in infinitesimal quantities but pose a grave risk when accumulated.

It is important to appreciate the complexity of the mixtures of chemical compounds that, under the effect of solar radiation for example, produce photochemical fogs or smog (tropospheric ozone, a harmful irritant, in contrast to stratospheric ozone, which provides protection against ultraviolet radiation).

This atmospheric pollution is worse in urbanised areas with features such as unfavourable topography (e.g. mountainous basins) or meteorology (e.g. thermal inversions) which prevent dispersion of pollutants. The improvements made in fuel quality and vehicle emission standards in most OECD countries tend to lower the trend of total emissions of these kind of pollutants (for instance in the USA[6] or in the European Union[7]). However the continuous and fast growth of VKT

5. Cf. the discussion of the gasoline lead phase-out program in the United States, below.

6. See the ZEV case study below

tends to offset the optimist projections in these countries while the rapidly growing use of old and less compliant vehicles in developing countries (e.g. in Latin America) is a major source of high-level air pollution in urbanised areas of the developing world (Onursal and Gautam, 1997). The OECD projections to 2030 confirms the continuation of the decline of traffic-related local pollution in the OECD area (OECD, 2000b) while in the rest of the world this pollution should increase.

A technology dilemma?

While the generalisation of catalytic converter is an adequate answer to the local/regional pollution problem, this is not the case for CO_2 emissions. Of course reductions in unit vehicle emissions or consumption can still be achieved while battery electric vehicles are developed for introduction to the market within the next 10 years to address this issue of local/regional pollution[8]. However this last solution is criticised from the point of view of both CO_2 emissions, depending on the way electricity is produced, and waste recycling or stocking, whether considering batteries or nuclear power.

This dilemma could resolve by considering the longer term perspective of CO_2 emissions reduction. As underlined in the EST study (OECD, 2000b) if the CO_2 target is met, mostly by reduction of overall fuel consumption, the other air pollutants emission targets would also be met. This is why a focus given to CO_2 target would probably not offset the achievement of other pollutants targets.

It could be said that in the longer term, e.g. more than 30 years, completely new and perhaps truly "clean" technology will be available. However the current situation requires immediate actions to be undertaken, whether addressing the local/regional pollution issue in urbanised areas in both industrialised and developing countries, or the Rio objectives on GHG emissions which are much more ambitious than the Kyoto ones. As already pointed in the previously quoted studies, technology alone will not be sufficient.

Safety, physical disruption and noise

Apart from these environmental aspects, the need to maintain the quality of life also implies dealing with the safety hazards posed by vehicle traffic, not only for motorists but also for pedestrians and cyclists. It also implies minimising the disruption caused by transportation infrastructure works; this increases further the cost of projects for burying or covering roads. Vehicle hazards translate statistically into numbers of accidents, deaths and injuries, and remain a major problem in most countries, linked to the continuous growth of road traffic.

Finally, one of the worst nuisances felt by city dwellers is the noise that penetrates their homes. For instance this has become a subject of major concern in the European Union. Yet the disparity of exposure is great: vehicle traffic is more disturbing in large cities, while motorcycles and heavy trucks are a greater concern in suburban or sparsely populated areas. This problem also has obvious social implications: for example, in France the number of people subject to an average noise level of more than 65 dB is four times as high among the lowest income groups than among the wealthiest. Generally speaking, the conjunction of urban crowding and automobile traffic growth tends to increase people's exposure to noise. Despite efforts to date to equip urban highways with acoustic barriers, technical solutions remain very limited.

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7. See the Auto-oil programme

8. See the ZEV case study below

2.3 Selecting a control strategy and instruments

In seeking an optimal, or at least acceptable, balance for the community as a whole, a trade-off must be made between the advantages offered by the development and use of transportation systems, on one hand, and the environmental and social costs which follow from, on the other hand. This trade-off should be such as to maximise the total net surplus for the community, *i.e.* advantages less costs.

Strategies for controlling the transportation system must take account of certain specific features of transportation, in comparison to conventional economic goods. For the transportation system user (motorist, pedestrian, public transit passenger, freight carrier) transportation is an intermediate good: it allows a citizen, for example, to pursue his work in a given place, or a wholesale business to deliver a good to a retailer for sale. The demand for transportation is a derivative of the demand for conducting activities in determined places, or the demand to make goods available in determined sites. It follows that transportation, since it consists of a service for moving goods and people between two given places, is a unique good in spatial terms: a given movement can rarely be substituted by another, because it will not have the same utility.

This intermediary good is itself not storable, and not transportable, which explains why its production is subject to specific peaks in space and time. This production involves the following agents:

- The public authorities who regulate the transport activity, supply the infrastructure, and subsidise the transport operation, sometimes through public-private partnership.
- Automobile makers who provide the vehicles that use the roads.
- Oil producers, importers and refiners, who supply the fuel used in the vehicles.
- Road hauliers, companies using road for transporting freight for their own account, and other transportation operators providing transport service, including urban public transport, railways, coach lines, taxis and airlines operators.
- Households, through their own automotive consumption (car purchase, maintenance and use).
- As well, transport infrastructure and service users inject their own travelling time as an input in the production of transportation.

Available controlling instruments involve not only the transportation system in its strict sense, but also land use management. They may be broken down into the following categories:

- Regulations governing land use (lot occupancy coefficients, zoning), roadway use (traffic regulation, sharing the road between different modes, speed limits), fuel standards and vehicles (emission and safety standards).
- Taxation, essentially through fuel taxes, but also through land use taxes (real estate and home occupancy taxes).
- Transferable permits as a hybrid instrument between regulation and taxation.
- Voluntary agreements negotiated with business industries, such that they limit their emission or consumption of depletable resources.
- Infrastructure user fees, through road tolls and railway fares.
- Land use planning policies, through exercise of expropriation rights and official powers to decide the location of public facilities.
- Financing the supply of infrastructure and transportation services (by the means of the general budget, specific taxes, or user fees, sometimes through a public-private partnership), and subsidisation of new technology developments (new engines, new fuels, etc.)
- Organisation of transport markets (organisation and regulation of markets, concessions and licenses).

2.4 Pros and cons of using transferable permits in the transport sector

Two main criteria can be used for judging the relevance of permits systems[9]: these are, on one hand, the ability to impose a constraint or a right defined in a quantitative manner within a specified space-time, and on the other hand the ability that agents have to transfer all or a portion of these quantitative obligations.

First, permit systems are of particular interest because of the relevance of certain of their attributes[10] to control of the transport sector:

- In cases where a given environmental performance must be achieved in a context of uncertainty over agents' price response functions; in such cases a permit system is more likely to achieve a quantitative objective than taxation.
- In cases where agents are more sensitive to quantitative signals than to price signals (notably if the elasticity of demand to prices is low in the short or medium term).
- In terms of flexible implementation of control policy, permit systems (like road tolls) can be used to target local and regional problems arising from transport activities.
- In political terms, systems in which permits are allocated free of charge are seen by agents as a means of avoiding an additional tax: in some cases, e.g. high fuel duties in Europe, this can enhance the acceptability of the new instrument.
- A quota system is the only type of system which allows distributive impacts to be treated explicitly and separately from the issue of the economically efficient allocation of efforts to reduce environmental damage. Given the fundamental role that transport plays in the right to mobility, these distributive impacts merit close attention.

Since the most difficult step is firstly to set up permits in the transport sector, most of the discussion on pros and cons is about permits system per se. Making them transferable is a second step which would essentially add transaction costs, as mentioned below.

Arguments for

Besides the previous general aspects, there are other arguments in favour of the use of permit systems in the transport sector.

In many instances it is possible to set precise and measurable targets, as in the case of local or global emissions of air pollutants (greenhouse gases) or congestion. In all cases, it is the sum of individual outputs of agents which produces the overall output[11]. In contrast, this does not apply to noise which does not increase linearly in accordance with the number of individual emissions.

It is also possible in many of the cases to establish space-time equivalents of aggregate nuisances, as in the case of local or regional air pollution for which permits can be traded within the corresponding geographical area; this is also the case for global air pollution such as greenhouse gas emissions, for which the corresponding geographical area is obviously the planet. Such equivalents

9. Cf. OECD, 2001

10. See OECD, 2001

11. Primary gases in the case of air pollutants. Possible secondary chemical reactions, such as ozone formation, are not taken into account.

can not be applied as readily to congestion, which is a phenomenon that is usually restricted to specific routes and times of the day. It would nonetheless be possible, in the case of conglomerations or urban areas subject to widespread road congestion, to design quota systems applicable to trips made by road within the area[12].

Lastly, the appearance of threshold effects may require a quantity-based approach to be adopted. This is clearly the case for greenhouse gas emissions, but also applies to local emissions of air pollutants which must not be allowed to exceed specific thresholds beyond which they become a health hazard.

Arguments against

Among the arguments against the use of permit systems in the transport system, the foremost is the question of the cost of administering such systems which by definition target a large number of mobile sources. From this standpoint, if the target nuisance can be linked, with an acceptable degree of approximation, to fuel consumption (for reducing greenhouse gases emissions, for example), then increasing existing duties on fuel would naturally be the cheapest solution. In other cases such as congestion, the use of automatic vehicle identification technology to collect electronic road tolls from moving vehicles, a practice that is currently growing rapidly, would seem to be the most appropriate solution. Indeed this technology can provide a basis for reducing the administrative costs of certain types of permit system (see the case study below on Ecopoints in Austria).

Moreover, making permits transferable would involve well-known transaction costs (Stavins, 1995). Resolving this issue of administrative and transaction costs is a key element in the introduction of any transferable permit system in the transport sector. We shall see in the case studies examined later in this report how this issue can be dealt with under different conditions.

It is by no means clear, a priori, which of the two instruments would be the least socially acceptable, *i.e.* rationing of the activity either by outputs (permits) or by prices (a much higher tax than currently to meet the objective of reducing emissions): such a rationing would target indirectly, at least in the short term, the number and length of automobile trips. Since the automobile plays a fundamental role in mobility, as there is frequently no transport alternative, that would make explicit restriction on freedom of movement, a fundamental right that is universally recognised in declarations of human rights such as that of the United Nations.

Lastly, compared with the sole use of regulations, transferable permits would introduce the notion of a commercial transaction on the *rationing* of the universal right to freedom of movement, as mentioned above. In many cultural contexts this might represent an additional handicap.

2.5 The potential for transferable permit programs in the transportation sector

The global of transportation activity is the result of a combination of factors relating to land use (location of activities and consequences for distances travelled), the supply of infrastructure and services (price and quality of service for different modes of transport), technical features of transport vehicles (energy source, unit consumption, emissions) and the intensity of vehicle use (mobility as a function of economic and social trends). These factors all offer potential fields of action for controlling nuisances in the transport sector.

12. See, for example, Marlot (1998) or Viégas (1999).

The first potential field concerns land use, and in particular the battle against urban sprawl and the dispersal of activities, which lead to longer travel distances: these longer distances are most often travelled by private automobile, since in such scattered settings there is no public transit alternative. This field is currently managed through the regulatory approach, but it could be the subject of quantitative controls in the form of rights yet to be devised.

A second field concerns the supply of infrastructure, which is today controlled primarily by the availability of financing through general budgets, but is increasingly the subject of user fees. The use of permits would not seem appropriate in this field, a priori, but we will suggest hereafter some points for debate on the co-ordination between permits and financing.

A third area concerns, on one hand, vehicle technology, through the regulation of unit emissions of local or global (greenhouse) air pollutants, and on the other hand, the technical characteristics of fuels. This is where we find the most advanced use of permits, as we shall see below, applied either to automakers or to fuel refiners (quality standards).

A fourth field concerns total fuel consumption by vehicles, as a function both of the intensity of use and of unit consumption, where the natural instrument is a tax. However, as we shall see, the observed social limitations on fuel taxes suggest the wisdom of exploring permits systems that would target the intensity of carbon consumption inherent in fuel consumption.

Finally, there is a fifth field, related to the previous one, which also involves controlling vehicle use but is focused more precisely on VKT (vehicle kilometres travelled), as a medium-term solution for controlling noise emissions or congestion. The most common approach to noise control is still through regulation or funding for acoustic protection around transportation infrastructure. With respect to congestion, control is still achieved essentially through the waiting line, despite the ongoing and long-time debate about congestion pricing. One relevant use of permits could therefore involve setting quotas on VKT or trips within a given urban area that would be allocated to motorists and could be transferred among them[13].

In addition to the well-known problems of designing and implementing transferable permits systems in general[14], however, their use in transportation systems involves some specific considerations. These relate, on one hand, to the great number of externality sources, and on the other hand, to the burden of a regulatory system that, as described briefly above, is already quite complex.

The great number of mobile emission sources that automobiles represent constitutes an obvious obstacle to decentralising permit systems in the transportation sector, since administrative costs would appear, a priori, to be prohibitive. This explains why most proposals to decentralise permits have stopped at the level of automobile makers, and have been targeted at unit vehicle emissions. We shall return to this question later.

The pre-existence of a complex control system for the transport sector implies that the design of transferable permits will have to consider how they can be inserted into the system. As we shall see in the following case studies, these systems can be used as a supplement to existing controls.

Given the limited scope of this study, it has been impossible to explore all the potential fields identified above. We have selected four case studies that illustrate the diversity of possible applications.

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13. See Verhoef et al (1996), Marlot (1998).

14. See OECD, 2001.

3. The case studies

The first case study is the US lead phase-down in gasoline and illustrates the case of fuel quality standards. This is a program of lead rights tradeable between refineries that was implemented from 1982 to 1988 to ease the accelerated phase-down of lead in gasoline until a complete ban came into effect in the USA in 1996.

The second case study is the Ecopoints system for heavy goods vehicles (HGV) in transit (Austria) and points to the issue of vehicle intensity of use. This is a non-tradeable permits system which was introduced in Austria in 1992 to prevent an excessive rise in HGV emissions throughout the country's Alpine region. Although the permits are not transferable, the experience gained in this experiment serves to highlight the potential advantages and drawbacks of permit systems applied to mobile sources such as HGV.

The third case study is California's Zero Emission Vehicles (ZEV) program and illustrates the case of unit vehicle emissions. This is an on-going scheme of tradeable permits aimed at reducing the level of local pollution stemming from car use, by increasing the proportion of ZEVs and LEVs (Low Emission Vehicles) sold by automobile manufacturers in California.

The fourth case is a prospective study in France for introducing transferable permits within the transport sector in order to control the GHG emissions of this sector. It was conducted by a working party of experts and representatives of transport operators and users. The study reviewed possibilities for introducing both "upstream" and "downstream" tradeable permits and covers the wide range of implementation levels, from unit vehicle emissions to fuel whole consumption

3.1 The Lead Phase-down Program (USA)

The objective of this program was to eliminate use of lead as a gasoline additive in the United States. This system of granting refineries rights to add lead to gasoline was part of a family of permits based on the concept of averaging. The program was in place from 1979 to 1996, when lead was effectively banned. It was also accompanied by a program of rights transfers that was in operation from 1982 to 1987: the trading of rights between refineries was authorised from 1982 to 1986, and the banking of rights was authorised from 1985 to 1987.

Context

Lead is one of the substances added to gasoline to increase the octane rating and reduce engine knock. These properties were discovered by engineers at General Motors in 1921, and the use of lead spread quickly. The toxic effect on human health of lead in high doses had been suspected, if not fully understood, since Roman antiquity at least. But it was only after the enormous expansion of lead use, as automobile travel became generalised, that the first scientific proof of its toxicity, even in low doses, surfaced during the 1960s and 1970s: lead then came to be seen as a threat to public health (Lewis, 1985).

The combustion of leaded gasoline disperses lead particles into the atmosphere. Lead is a severely toxic pollutant that causes neurological dysfunction that can have grave consequences, particularly on the foetus and on children. In adults, low doses provoke hypertension and increased risk of heart disease. This, then, was the primary motivation of the control programs that were introduced in many countries to cut back or completely eliminate lead in gasoline.

The second motivation for phasing out lead in gasoline had to do with the use of catalytic converters, which are rendered inoperable by the presence of lead in the fuel. Catalytic converters were introduced by all U.S. automakers, beginning in 1975, to reduce other emissions associated with fuel combustion (hydrocarbon compounds, carbon monoxide, oxides of nitrogen). This requirement also explains the declining consumption of leaded gasoline, as the automobile fleet was gradually renewed.

The use of lead as an automotive gasoline additive has now been banned in most OECD countries. On the other hand, lead continues to be widely used in Africa, in Central and South America, in Asia and in Eastern Europe. In many of these regions there is no lead-free gasoline available (Kaysi *et al*. 2000). The essential rationale behind continued use of lead in some countries is that lead is still the least expensive means for increasing the octane rating of fuel[15].

Program description[16]

The regulatory program for reducing lead in gasoline was initiated by the U.S. Environmental Protection Agency in December 1973. The EPA is a federal agency responsible for defining and enforcing environmental regulations and standards pursuant to federal environmental legislation, and in particular the Clean Air Act.

Flexibility was built into the lead phase-down program in a series of three successive stages: averaging, trading and banking.

First stage: averaging over a quarterly compliance period.

At the end of 1973, the EPA proposed new regulations for a five-year reduction in the average lead content of all gasoline products (leaded and unleaded) for every refinery, measured on a quarterly basis: this average was to be cut from the current level of about 2 grams per gallon to a maximum of 0.6 grams per gallon in 1978. Enforcement of the new regulation was held up for several years by lawsuits, and the 0.5 grams-per-gallon rule came into effect only on October 1, 1979. This was the first stage in the introduction of a degree of flexibility in the standard, which was applied not to each gallon but as an average to total output over a three-month period.

Second stage: tightening of the standard and introduction of the rights trading option

New scientific revelations about the toxicity of lead allowed the EPA to tighten the standard in 1982, to a maximum of 1.1 grams per gallon, but calculated this time solely on the total of leaded gasoline produced by the refinery. As well, the very small refineries were granted a slightly less rigorous rule, from October 1, 1979 to July 1, 1983.

To facilitate this adjustment, a program was introduced for trading rights to add determined amounts of lead to gasoline.

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15. The alternatives consist of replacing lead by oxygenate additives such as ethanol or methanol, which are less expensive, or modifying refineries to use other procedures for increasing the octane rating (platforming, alkylation, isomerisation).

16. The principal reference works are: Hahn and Hester 1989, Kerr and Newell 2001, Kerr and Maré 1998, Nussbaum 1992.

This rights trading program reflected a concern to give refiners (in particular the smaller ones) some flexibility in adapting to the new rules.

The rights allocated to each refinery were calculated in terms of the quantity of leaded gasoline produced by the refinery and the authorised amount of lead per gallon (as determined by the EPA). For example, a refinery producing 100 million gallons of leaded gasoline during any quarter of 1983 or 1984, when the standard was 1.1 g per gallon, received rights to 110 million grams for the quarter. If the refinery added less than the authorised amount of lead, it could sell its lead rights up to an amount equal to the difference between the actual amount added and the norm. If on the other hand the refinery wanted to add more lead than the rule allowed, it would have to purchase rights prorated to the excess.

The balance between rights held or acquired and rights needed for each refinery was recalculated every quarter, and had to be either positive or nil.

The validity of the rights was limited to three months. Refiners could use up their rights over the course of that period (by adding more or less lead to their gasoline during the quarter) or they could sell these rights to other refiners during the same period. Any rights not used or sold would expire at the end of the quarter for which they were created.

This last rule was amended with introduction of "banking" in 1985.

Third stage: further tightening and introduction of the rights banking option

The allowable level of lead in gasoline was further reduced to 0.5 g per gallon in mid 1985, and the EPA announced that the program for trading rights between refineries would end in 1986. In 1985 the EPA also introduced the banking of rights, which could be used until the end of 1987.

In anticipation of the tightening of the rule in mid 1985, refineries added less than the authorised amount of lead during the first two quarters of 1985, and banked the balance of their rights: the smaller refiners were then able to use or buy banked rights from the larger refiners during the second half of the year, and could thereby exceed the norm.

In 1986, the standard was cut again to 0.1 g per gallon, while the ability of a refinery to bank its rights was abolished. Rights acquired by one refinery from another could still be banked, but few refiners took advantage of this possibility. The reaction of refiners was to use their previously accumulated rights to continue exceeding the stricter standard.

Fourth stage: windup of the rights transfer program

The rights trading program came to a close at the end of 1986. On the other hand, banked rights could still be used until the end of 1987. In 1988, after the rights transfer program was terminated, the 0.1 g per gallon rule was applied to each refinery individually. Lead was finally banned as an additive to gasoline for road vehicles in 1996.

In summary, during the course of the gasoline lead reduction program, which ran from 1979 to 1996, the rights transfer program was operational from 1982 to 1987.

Participants in the program

All refiners were subject to the EPA rule governing the authorised amount of leaded gasoline. On the other hand, participation in the rights trading program was optional. In 1983 and 1984, for example, it is estimated that there were between 300 and 400 refineries involved in such trading.

Managing transfers

No prior approval was required from the EPA for trading permits, but an ex post declaration had to be submitted (see below).

There were however some minor restrictions on trading:

- California refiners, which were subject to a stricter standard, could not resort to trading in order to exceed the State norm.
- Small refineries, which initially faced a less severe standard, were not allowed to sell their rights to the large refiners.

This second restriction was eliminated on July 1, 1983, after which time all refiners were subject to the same standard.

The most important restriction concerned the life of the permits, which was limited to three months after 1985. Nevertheless, there was significant trading both before and after that date.

Monitoring and auditing

Under the regulations, every refinery had to submit a quarterly declaration to the EPA on its output of gasoline and the amount of lead used. Refineries participating in the trading and banking programs also had to indicate in the declaration the amounts traded or banked, and the refineries with which they had conducted their trades.

Overall, transaction costs as identified by Kerr and Maré (1998) were not insignificant: they included the costs of optimisation, the costs of seeking a partner and researching prices, the cost of uncertainty about the validity of permits that were, in practice, traded before they were validated by the EPA at the end of each quarter, the costs of negotiation and the costs of disclosing confidential information on refinery output. The presence of these costs explains why the smaller refineries, those belonging to companies of smaller scale or with fewer refineries, tended not to participate in the trading. By contrast, refineries belonging to the larger companies were more active in trading, reflecting their degree of specialisation in producing different types of fuel.

Assessment

The market for these rights was very active, and the volume of trading between refineries rose steadily as the standard was progressively tightened. In 1983 and 1984 there were between 300 and 400 potential participants in the trading, and refineries purchased between 10 and 20 percent of all the lead rights they used (Kerr and Maré, 1998). The tightening of the standard in 1985 and the simultaneous introduction of the banking option further stimulated trading from 1986 onward. The portion of lead rights traded reached 50 percent of the total by mid 1987 (Hahn and Hester, 1989).

Overall, the small refiners were net buyers of rights from the large refiners, which adjusted more quickly to the standard. However, some of the small refiners also took advantage of the program by selling their rights.

Costs and benefits of the program

Since the trading was conducted freely and was subject only to a simple declaration, the EPA did not collect systematic data on the trading price of rights. It would appear that the price was about one cent per gram of lead before banking was authorised, and rose to between two and five cents per gram thereafter (Hahn and Hester, 1989).

Enforcement of the standard for gasoline lead additives, quite apart from the rights transfer program, required the EPA to monitor and audit the amounts of lead consumed by the refiners. The incremental cost of administering the rights transfer program were limited to ex post controls over trading between refineries and the detection of invalid rights. It could be said, then, that the administrative costs due specifically to the transfer program were very modest.

Cost savings to the refiners from the trading and banking program have been estimated at several hundred million dollars (Hahn and Hester, 1989)

Finally, Kerr and Newell (2001) have shown that the refiners with the lowest adaptation costs were more likely to adopt new technologies under the marketable permits regime, and this encouraged the spread of those technologies.

The debate over the program's cost efficiency

It has been objected that the concept of allocating rights proportionate to total leaded gasoline produced was in itself an incentive to overproduction of leaded gasoline. In response, it has been argued that, since the addition of lead is the most profitable way to increase fuel octane ratings, the total amount of lead authorised by the standard would have been used whether there was a trading program or not. That program merely rearranged the distribution and use of lead among refineries, and did not increase the total amount of lead that could be used. Moreover, any incentive to overproduction of leaded gasoline was offset by the constantly rising market share of unleaded gasoline, as vehicles with catalytic converters became more common.

The fact that refineries were free to manage their rights without prior certification, and that their only constraint was to file a declaration at the end of each quarter, was certainly an advantage in activating the market, but it also gave rise to legal problems. Since the balance between rights acquired and rights required was calculated at the time of the end-of-quarter declaration to the EPA, it was possible that rights might be sold only to be found invalid after the sale, having been improperly created, either by error or intentionally. Once they were sold -- and they could in effect be resold several times by brokers -- it was difficult if not impossible to blame the error on a party that had bought them in good faith.

Moreover, the emergence of brokerage activity as a means of facilitating trading had no formal connection to the EPA program. Brokers did not take ownership of the rights, but merely established a market between refineries. The EPA therefore has no direct knowledge of the scope of their intervention.

In the end, the program was regarded as a success: the total authorised amount of lead was not exceeded, and some refiners that would likely not have remained in business in the absence of this flexibility were in fact able to adapt successfully.

Conclusions and lessons

The success of the program can be explained by the lack of restrictions on trading and by the possibility of banking: the market thus remained very active. Administrative requirements were kept to a minimum (a simple declaration of trading and additive purchases). But these factors alone cannot explain its success: we must add the fact that the refiners were accustomed to dealing together on other markets, in particular for other additives (Hahn and Hester, 1989).

In summary, this program exhibited three basic features that may be taken as essential ingredients for ensuring the success of the transferable permits program:

- A precise definition of the permit unit (gram of lead) eliminated all ambiguity as to what was being traded or banked.
- The rules to be followed were simple and trading could be conducted freely.
- Program implementation was pragmatic, and offered several options.

A further positive factor that facilitated implementation of the program was the availability of affordable technological solutions for replacing lead in gasoline, together with the fact that the consumption of leaded gasoline was on a declining trend because of the evolution of the automotive fleet. The transferable rights program served to accelerate the decline in lead use.

We may say that, in terms of the continuing consumption of leaded gasoline in other parts of the world (see Introduction), a program of this type could be a viable option, provided the local political and regulatory context is appropriate.

3.2 The Ecopoint program in Austria

Ecopoint is an ongoing program for limiting pollution and noise from truck traffic passing in transit through Austria. It belongs to the quotas category, but in this case they are of the "cap but no trade" type, since they are not transferable. The interest in analysing this program is to show how a system of permits applied to mobile sources can be put in place to protect the environment of a given region.

Context

Austria is located at the crossroads of Central Europe's transit routes. Through it pass the north-south route between Italy and Germany and one of the major links between countries of Eastern Europe and Western Europe[17].

Austria's typically mountainous geography funnels north-south traffic along Alpine valleys, and in particular the Brenner Valley, where the ecosystem is very fragile. The morphology of these

17. The main reference used for this report is CEC (2000), apart from a few Internet resources.

valleys is such that exhaust emissions cannot readily escape, and highway noise is intensified. The concentration of nitrogen oxides is three times as high as on a plain with similar traffic volumes.

Moreover, until recently neighbouring Switzerland has imposed a maximum truck limit of 28 tons, versus 40 tons in European Union. Beginning 2001 these regulatory limits have been suppressed (free traffic for 28-34 tons, quotas until 2005 for 34-40 tons) while road fees are imposed on trucks. Waiting at borders and road fee still constitute an incentive for north-south traffic to take alternative routes through France or through Austria.

The result has been a sharp and steady increase in truck traffic transiting through Austria: such traffic rose by 46.9 percent between 1993 and 1999, for trucks of member States of the EU-12, to a total of 1,445,700 trips in 1999[18], with an increasingly adverse impact on the natural and human environment in the areas traversed. The Brenner Valley received more than 60 percent of this traffic in 1999.

At the same time, freight transiting by rail rose by 35.66 percent between 1991 and 1999 (52.7 percent between 1993 and 1999, with a sharp decline in 1993). It amounted to more than 23 million tons in 1999. More than a quarter of this freight moves between Germany and Italy.

Concern over this growing pressure on the environment led Austria to negotiate an agreement with the European Economic Community in 1992, calling for a reduction in noise and atmospheric pollution generated by heavy trucks transiting through Austria. The chosen target is NO_x emissions produced by trucks of a gross weight of more than 7.5 tons. The initial objective in the agreement was to reduce these emissions by 60 percent, as compared to the reference year 1991, over the 12 year life of the agreement, *i.e.* by 2003.

In 1995, when Austria joined the European Union, this agreement was confirmed as a derogation to the Single Market provisions. Protocol 9 of Austria's act of accession extends the regime to December 31, 2003.

Program description

The agreement between the EEC and Austria introduced a system of transit rights known as "Ecopoints" ("Ökopunkte") for heavy goods vehicles of a gross weight of more than 7.5 tons transiting through Austria, whether loaded or empty.

The trucks concerned are those of all European Union member countries and certain other countries (Switzerland, Norway, Liechtenstein and Slovenia) that have signed agreements with European Union.

Emissions are represented by a quota of points called Ecopoints, which must be used by trucks transiting through Austria. The total quota as calculated in 1991 is to be reduced every year on a straight-line basis to reach the 60 percent reduction target in 2003.

Definition of Ecopoints and period of validity

Since October 1, 1990, unit emissions of heavy vehicles are referenced in an approval document produced at the time the vehicle is manufactured. Each member State must establish a

18. 1,706,545 trips for the EU-15.

COP (Conformity of Production) document for the vehicle, declaring NO_x emissions and the number of Ecopoints required to travel through Austria.

An Ecopoint corresponds to the emission of one gram of NO_x per kilowatt-hour (kWh). For example, a vehicle normally emitting 10 grams of NO_x per kilowatt-hour will have to use up 10 Ecopoints to transit Austria.

For vehicles manufactured before October 1, 1990, or without a COP document, a flat charge of 16 Ecopoints is required. When a vehicle receives a new engine, it will be given a new COP document.

Ecopoints are valid from January 1 of each year until January 31 of the following year, *i.e.* for 13 months. This allows hauliers to avoid having to use two different sets of Ecopoints when making round trips at the end of the year.

Initial allocation

Ecopoints are distributed by the European Commission among member States according to an allocation schedule established in the regulations and periodically revised by the Commission. Countries are then responsible for redistributing their Ecopoints among their own hauliers.

The allocation schedule among States was based on their share of the traffic between the Community and Austria in 1991. In practice, Italy and Germany use two-thirds of the Ecopoints, while the third-largest user is Austria itself (15 percent).

A Community reserve has been established, holding 3.34 percent of each member State's allocation, to meet any temporary deficit of Ecopoints.

Ecopoints are allocated to member States every year by the Commission in two blocks, one before October 1 of the year preceding the year of validity of the Ecopoints, and the other after March 1 of the validity year. Countries that do not expect to use all of their allocated Ecopoints must return the unused points by October 15 of their validity year. The Commission can then redistribute these points, together with those from the Community reserve, to other countries, in accordance with recognised criteria, and at latest one month before the end of the validity year. This redistribution is decided by a committee of member State representatives.

The transfer of Ecopoints, then, does not involve any kind of market. It is done exclusively through an administrative process that involves all participating States, and is limited to unused Ecopoints.

NO_x emissions were targeted in order to encourage the use of increasingly cleaner trucks. A further objective was to reduce noise. To avoid a situation where NO_x emissions might be reduced while allowing an increase in transit traffic, a special quantitative limit on transit trips, the so-called "108 percent clause", was imposed from the outset: if the number of transit trips in any given year exceeds that of the reference year 1991 by more than 8 percent, the number of Ecopoints distributed in the following year must be cut by 20 percent beyond the linear reduction already established.

Such a situation occurred in 1999, and sparked a dispute between Austria and other member States: this was settled by the European Commission in 2000, by suspending temporarily enforcement of the clause. According to that clause, many countries (including Germany, the principal user of Ecopoints), would have reached their quota as early as the summer of 2000: carriers from those countries would no longer have been able to transit through Austria. This crisis, marked by

demonstrations in Austria against truck traffic on the Brenner motorway, led to a reassessment of the program and proposals for reform from the European Commission. In February 2001, the Commission set forth new regulations that would: suspend the "108 percent clause" on an exceptional basis for the year 2000, but confirm its enforcement for the years 2001 to 2003; impose a new reduction in Ecopoints until 2003 to reach the initial objective of a 60 percent cut; and establish a four-year rescheduling of the reduction of Ecopoints that would result from renewed enforcement of the 108 percent clause. This reduction would be shared among member States as a function of the growth rate of traffic generated by their own hauliers. However, Austria appealed to the European Court of Justice to overturn the new regulations and on February 23, 2001, the President of the Court ordered suspension of the measure rescheduling the Ecopoint reductions over four years.

Monitoring and auditing

The Ecopoints were initially issued in paper format, a system that required systematic manual controls at the border. With Austria's entry into the European Union and abolition of internal border controls within the Union, systematic border stopping were no longer required and a system had to be found that was compatible with these new provisions.

This was accomplished with the introduction, on January 1, 1988, of an electronic system for processing Ecopoints by automatically detecting heavy vehicles with the aid of an onboard device.

Consistent with the desire of member States to minimise the cost to hauliers of installing the equipment, it is a very simple device. The onboard electronic transponder[19], called an "Ecotag", identifies the haulier and the vehicle and contains details from its COP document.

The haulier must first register with the competent authorities of his own country, who have a direct link to the central system in Austria for recording information on hauliers, their vehicles (COP data) and their Ecopoint credits. The haulier must purchase an Ecotag (available in member countries and at the main Austrian border points) and then, before its first use, it must be initialised and mounted behind the vehicle's windshield by Austrian officials on duty at selected border crossings.

The Ecotag automatically signals the passage of the truck across the border[20]. It is only if the vehicle is en route to a destination in Austria (and not in transit) that the driver must push a button to signal this exemption. Information on the date, time and point of entry into Austria are recorded in the Ecotag. These data allow for inspection by mobile control units within the country. Upon leaving the country, the vehicle's passage will again be detected by the electronic station, which will read the information recorded in the Ecotag and transmit it to the central system that maintains haulier accounts. If Ecopoints are required, the central system debits the points, transmits an electronic invoice to the haulier's country and notifies the authorities of any trips not covered by sufficient Ecopoints. The system will reject Ecopoints if their validity date has expired. Hauliers can consult the balance on their Ecopoint account through national terminals located in member countries.

European regulations require that the electronic system must be used for most transit trips. This means the purchase and installation of the Ecotag device, which may seem an excessive burden for occasional transit trips. The regulations therefore authorise each member country to use the paper system up to a limit of 0.6 percent of the total quota, or some 9,000 trips for each country. In practice, more than 95 percent of Ecopoints are handled electronically.

19. 5.8 GHz DSRC transponder.

20. 169 border crossings are equipped with overhead electronic readers.

For vehicles without Ecotags transiting Austrian borders within the European Union, the driver must cancel the paper Ecopoints in special machines located at the principal points of entry to Austria. Random controls are performed on vehicles travelling within the country.

Systematic control of paper Ecopoints has been retained at Austria's frontiers with non EU countries. The driver must present his COP document and the required Ecopoints, which will be stamped. One copy is retained by the Austrian entry-point authorities, one is kept by the driver as proof for presentation during random inspections, and a third copy is handed over to the authorities upon exiting Austria. The year of validity is printed on the paper Ecopoints, and they must not be used after the expiration of their validity period.

About 4 percent of trips declared by Ecotag involve unauthorised use, and perhaps half one million fraudulent Ecopoints are used each year. On the other hand, there are no data on fraud committed by vehicles not equipped with Ecotags[21].

Regulatory framework and authority responsible for program operation

The principle of the Ecopoint program was adopted by the European Union upon Austria's accession in 1995, as a derogation from the provisions governing the Single Market, which prohibit any barrier to trade in goods between member States. Nevertheless, to the extent that the Ecopoint program applies to all member States, it does not violate the essential principles of non-discrimination and harmonisation of competition. The Ecopoint distribution formula is established by regulation of the European Commission.

The Ecopoint system is the property of the Austrian Ministry of Transport. Installation and operation of the electronic system has been contracted to a private operator.

Evaluation and comment

There is very little quantitative data available on the costs and benefits of the Ecopoint program. With regard to costs, one must consider the fact that electronic road pricing systems targeting trucks will be implemented and interoperable in the short term in several European countries: this is already the case in Switzerland since 2001 and it is planned in Austria and Germany. Thus it can be assumed that once these systems are operating the incremental cost of operating a permit system addressing truck traffic in ecologically sensitive areas will be low.

In terms of benefits, there has been a very clear technological training effect. The proportion of trucks (measured by number of trips) paying 15 Ecopoints or more dropped from 51 percent in 1993 to less than 2 percent in 1999. On the other hand, the proportion of trucks paying 7 Ecopoints or less (EURO II) skyrocketed from 0.1 percent in 1993 to more than 78 percent in 1999[22]. Thus, the average number of Ecopoints used by member country trucks has declined more swiftly than the target value established in the agreement.

Although it is impossible to say exactly what role the Ecopoints have played in this technological evolution, it is probable that the system has had a supplementary effect in accelerating the impact of the overall program for tightening European emission standards (EURO I, II and III).

21. Bearing in mind that the paper system represents only 5 percent of Ecopoints.

22. The Euro III Standard applicable in 2000 corresponds to a maximum of 5 Ecopoints.

On the other hand, the program's design has revealed a number of shortcomings. These are of four kinds: insufficient coverage of pollution sources, lack of incentive to use even cleaner trucks, an excessively comprehensive measure for addressing the objective of preserving alpine valleys, and a degree of mistargeting in environmental terms.

First limitation: inadequate coverage of pollution sources

Several categories of heavy goods vehicles escape the obligation to produce Ecopoints:

- Trucks en route to or from destinations in Austria are not affected by the Ecopoint system. The same is true for trucks travelling exclusively within Austria.

- Trucks from non-member countries[23] of the European Union and those using ECMT permits[24] are also exempt.

- The system does not apply to light-duty vehicles (less than 3.5 tons PTAC) or to buses, which produce NO_x emissions at levels similar to those of trucks.

In short, most heavy-duty vehicles travelling in Austria do not pay Ecopoints, because they are travelling to or from points in Austria. This loophole could provide an incentive to circumvent the Ecopoint system by splitting a transit trip into two trips, and simply transferring the cargo within Austrian territory.

Second limitation: lack of incentive to use cleaner trucks

Statistics on the average Ecopoints used by member State trucks show that the number declined by 44.7 percent from 1993 to 1999, or more swiftly than the target values established in the agreement.

Now that low-emission vehicles are in production and are being used, the "108 percent clause" represents a hindrance to the development of truly low-emission vehicles. In effect, using such vehicles would only make sense if additional trips beyond the 108 percent could be made for the same number of Ecopoints, which is not the case.

Third limitation: an excessively broad measure for meeting the objective of preserving alpine valleys

Although most transit traffic passes through the Alps (60 percent through the Brenner pass), the Ecopoint system applies to the entire territory of Austria. The system makes it possible for transit traffic, and hence emissions, to grow sharply in the alpine valleys as long as it is offset by reductions on the plains. The system as currently designed, then, is not accurately targeted at protecting regions that are particularly sensitive to local emissions.

23. With the notable exceptions of Switzerland, Norway, Liechtenstein and Slovenia, which also use the Ecopoint system.

24. 752 permits. Excessive use of these permits seems to be reported.

Fourth limitation: environmental mistargeting

Targeting NO_x emissions was seen as an indirect way of limiting overall nuisances, not only in terms of atmospheric pollution but also of noise. However, trucks have become steadily cleaner, and have exceeded the specific objective set for NO_x emissions.

Given that traffic volumes have been maintained and have even grown, despite the 108 percent clause (which was suspended for a time), there has been no improvement in other nuisances such as noise. In fact, the noise problem is unlikely to be abated unless something is done at the point of contact between the tire and the pavement, or unless traffic is moderated. The NO_x emissions target thus no longer plays a role in reducing the general level of nuisance from the road transport of goods.

Conclusions

The Ecopoint system has the advantage, first of all, of showing that it is technically quite feasible to apply a quota-based system of permits to mobile sources within a defined area. It thus provides one element of a response to the frequently heard objection that the administrative costs of permit systems for mobile sources will be too high.

The Ecopoint system suffers from several limitations, however, that could well present stumbling blocks for other programs of a similar nature designed to protect sensitive regions:

- To target nuisance vehicles in a given region more thoroughly would require a great many electronic detection devices: a trade-off would have to be found between the costs and benefits involved, depending on highway geography. Such as system would only be applicable in certain fairly specific geographical configurations (for example in valleys with few points of entry and exit).

- If there is to be continued incentive for further reductions in unit vehicle emissions, restrictions on total traffic volume would have to be abandoned.

One possible option for avoiding excessive traffic growth (which produces noise and diminishes road safety) might be to set stricter NO_x emission quotas while allowing them to be transferred among hauliers on a market basis: in this case, the incentive to reduce emissions could be maintained while controlling traffic growth.

However, tightening emission quotas in order to control traffic growth would create a gap between the physical basis of the quotas (pollutant emissions) and the target (traffic) that could give rise to allocational inefficiencies. An alternative might be to modify the physical base, in one of two ways: either by combining NO_x emissions and trips, or by replacing NO_x emissions by trips. The second option would have the advantage of keeping the base simple, an indispensable consideration for a properly functioning system. It would also do more to foster use of the rail as an alternative for crossing the country.

3.3 The ZEV program in California

The ZEV program (Zero Emission Vehicle) now underway seeks to speed up the introduction of electric vehicles in order to reduce local atmospheric pollution in the State of California. It includes a system of credits based on the concept of averaging. These credits can also be transferred between manufacturers (trading) and over time (banking).

This case study was chosen rather than the Corporate Average Fuel Economy (CAFE) program because the ZEV program goes further on the way of implementation of TPs at the level of automakers. CAFE[25] requires automakers to comply with the fuel economy standards (that is miles per gallon, mpg) set by the US Department of Transportation. CAFE value is computed by averaging the vehicle standards weighted by vehicle sales. Fines are set at 5.50 USD per tenth of mpg deficit. Since credits can only be banked or borrowed by the automaker up to three years in order to offset fines calculated in other years, CAFE can be considered as a non-tradeable permits system.

Context

In the 1970s it was frequent to have more than a hundred smog alerts in the Los Angeles basin over the course of a year. A major effort was then made to stiffen vehicle emissions standards, and emissions have now been reduced by 98 percent. Smog alerts have declined sharply, to the point where there were none at all in 1999.

The California Environmental Protection Agency, however, has determined that further efforts are required. Today, 95 percent of California residents live in areas where the air quality does not meet U.S. federal standards. Cars and trucks are the second greatest source of atmospheric pollution (accounting for more than 50 percent of smog precursors). Some one million vehicles are sold every year in California, which is the largest automobile market in the United States.

Since VKT are projected to jump tremendously in coming decades, progress in terms of reducing pollutant emissions[26] from gasoline-powered vehicles is likely to reach a dead end, since these vehicles will never be completely "clean". The only solution would seem to be a general resort to vehicles that produce no emissions, *i.e.* the ZEV. The ZEV is defined as having:

- no tailpipe emissions,
- no evaporative emissions,
- no emissions from fuel production and handling (*i.e.* at the refinery or at point-of-sale), and
- no onboard emission control system that might deteriorate over time.

In fact, the only technology that currency meets this standard is the electric vehicle[27].

The California Air Resources Board (CARB), a division of the California Environmental Protection Agency, is the office responsible for California's air pollution control program. The CARB has the legal power to introduce specific programs as necessary to conform with federal and State clean air legislation, as well as to supervise local air quality district programs. It has both the authority and the responsibility to issue State standards for air quality. In particular, it has the authority to regulate road vehicle emissions and other mobile sources. Finally, it is responsible for enforcing emission reduction measures adopted by the State.

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25. See http://www.epa.gov/otaq/factsht.htm, Office of Transportation and Air Quality of the US EPA.

26. CO_2, the main greenhouse gas, is not considered as a pollutant in this context.

27. Emissions from electric power stations, which represent a limited number of sources, are easier to control than those from multiple mobile sources, *i.e.* automobiles.

Program description[28]

The first major drive towards a sharp reduction in pollutant emissions from motor vehicles in California dates from the early 1990s. From the outset, this program relied on the principles of averaging-based permits, while setting a highly ambitious objective for the introduction of electric vehicles. The program has undergone several amendments that have taken some of the rigor out of the march towards the ZEV objective, while maintaining that goal despite pressure from automobile makers.

The 1994 program: annual averaging, trading and banking of credits, and introduction of ZEVs.

The CARB drew up the LEV I (Low Emission Vehicle) program in 1990. From the beginning, the California LEV program was seen as a complement to the federal exhaust emission reduction program, but one that was more rigorous in terms of its standards, in the context of the devolution of powers from the federal to the State level.

The LEV I program was designed both to enforce more rigorous emission standards and to give automakers greater flexibility in specifications for fuels, pollution control techniques and type of propulsion. It came into effect in 1994.

The program made reference to the concept of ZEV, and established four categories of LEV: the transitional LEVs (TLEVs), LEVs, ULEVs (Ultra LEVs) and the ZEVs. Each category is defined according to a maximum emissions level for hydrocarbons (NMOG, non-methane organic gases), carbon monoxide and nitrogen oxides. The NMOG index is taken as the reference indicator reflecting total emissions of atmospheric pollutants.

Instead of requiring every vehicle sold to meet a single emissions standard, automakers are allowed to spread their fleet among the four vehicle categories and to conform with the standard on the basis of a weighted average of emission rates for the entire fleet[29]. They can also earn credits if they are exceeding the required standard, and they can then sell or bank these credits, or they can purchase credits if their fleet does not meet the standard.

Finally, a constraint was issued requiring a minimum percentage of ZEVs to be delivered for sale. The initial legislation required the 7 most important automakers (American and Japanese)[30] to ensure that at least 2 percent of their vehicles delivered for sale in California be ZEVs in 1998, and this percentage was to rise to 5 percent in 2001.

Given California's leading role in this area, the initiative was taken up by other U.S. States (Massachusetts, Vermont, Maine and New York), despite several lawsuits brought by the automakers, and these States also adopted the subsequent amendments, with some changes.

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28. A useful source of information is the CARB web site, at www.arb.ca.gov/msprog/zevprog/zevprog.htm. The main references used in this section are CARB (2000a, 2000b, 2001) and Friedman et al (1998).

29. This weighted average is calculated for each automaker by breaking down the vehicle fleet into the four categories and applying the maximum allowable emission rate to the number of vehicles sold in each category.

30. Originally General Motors, Ford, Chrysler, Nissan, Honda, Toyota and Mazda. Mazda was subsequently reclassified as an intermediate volume manufacturer.

The 1996 amendment: longer deadlines and voluntary agreements

The automobile and oil industries, along with elected politicians, brought pressure to amend this legislation, resulting in the March 1996 memorandum of agreement (MOA): the CARB agreed to push back the 1998 deadline to 2003, but in return it insisted that the ZEV share of sales should be increased to 10 percent at that date. As well, through voluntary agreements negotiated between the CARB and each of the large volume manufacturers, the latter undertook to produce a certain number of demonstration vehicles between 1998 and 2000.

Mazda, one of the large volume manufacturers, in which Ford has a one-third interest, elected to purchase ZEV credits from Ford in order to meet the minimum production target for demonstration vehicles.

The 1998 LEV II program: introduction of partial ZEV (PZEV) credits.

The regulations were again amended in 1998, in recognition of the difficulties that automakers are having in producing low-cost ZEV vehicles, and the wide diversity of technologies available for further reducing emissions (in particular, hybrid electric vehicles and fuel cell vehicles). The LEV II program was also given further flexibility, in the form of partial ZEV credits which could be earned by producing "very clean" but not strictly ZEV vehicles.

Automakers can earn credits by:

- making ZEVs available for sale before the 2003 deadline: credits earned in this way are greater than unity (2 or 3 credits per ZEV prior to 2003, calculated according to the vehicle's all-electric operating range and battery charge data);
- selling vehicles that, although not strictly ZEV, are sufficiently clean to earn partial ZEV credits;
- purchasing credits from another manufacturer.

Partial ZEV credits are calculated by adding up scores under three headings:

- First of all, the vehicle must meet a minimum standard in order to earn a PZEV. It must be a Super Ultra Low Emission Vehicle (SULEV), *i.e.* a gasoline-powered vehicle certified to meet emission standards for 150,000 miles[31] and to produce no evaporative emissions: such a vehicle earns a credit of 0.2.
- Next, a credit is calculated in terms of the vehicle's zero-emission operating range (*i.e.* by recharging batteries off the mains); such a vehicle earns a credit of 0.2.
- Finally, the NMOG emissions level resulting from the fuel cycle (manufacture, transport and distribution) is calculated: if this level is less than that of electric power stations (which is the case with compressed natural gas) the vehicle earns a credit of 0.2.

31. In order to avoid the deterioration to which conventional vehicles are subject: aging and contamination of the catalytic converter, malfunctioning of the emissions control system, alterations or lack of maintenance by the user. Periodic inspection programs are inadequate to prevent such deterioration.

For example, the PZEV credit will vary from 0.2 for a gasoline-powered SULEV to one full credit for a hybrid electric SULEV (HEV) with an emission-free operating range of 100 miles.

Underlying these changes was the CARB's recognition that significant progress has been made in reducing emissions from SULEVs: because their operating range is greater than that of electric vehicles, they have the potential for greater market penetration and thus for reducing emissions more quickly. The requirement for 150,000 miles of emission-free operation is a further guarantee that SULEVs will maintain this advantage.

Nevertheless, since there can be no absolute guarantee against the deterioration of emission control mechanisms, the CARB issued restrictions on these mechanisms: the large volume manufacturers (those selling more than 35,000 cars and light-duty vehicles annually in California[32]) can meet their 2003 obligation with partial ZEV credits, but only up to 6 percent of the 10 percent ZEV mandate; the remaining 4 percent must be achieved through sales of actual ZEVs. Intermediate volume manufacturers (4,500 to 35,000 vehicles sold annually) can meet the objective entirely with partial ZEV credits, while the small volume manufacturers are not affected.

In introducing this amendment, the CARB expected that manufacturers would naturally turn to producing SULEVs as successors to ULEVs. In fact, this technological shift is less costly than a move to the electric ZEV, which does not yet enjoy the same economies of scale as the gasoline-powered vehicle. Moreover, by producing SULEVs, manufacturers can reduce the average NMOG index of their individual fleets, which provides a further incentive to making SULEVs instead of ZEVs.

To the end of 2000, only the Nissan Sentra (a SULEV), introduced in November 1999, had been certified for PZEV credits. Three other SULEVs introduced in 1999 and 2000 have yet to obtain this certification (because of other criteria, in particular the problem of evaporative emissions).

The 2000 review: deadlines confirmed and further flexibility

The 1996 agreement called for automakers to produce 3,750 ZEVs between 1998 and 2000. Thanks to the credits earned through the introduction of advanced batteries, this commitment was reduced to 1,800 electric vehicles. In 2000 there were about 2,300 electric vehicles on California roads but, despite growing consumer interest and a subsidy of US$5,000 to manufacturers for every ZEV sold, those automakers who reached their quota under this agreement have virtually ceased production. There are two essential reasons behind this "black out": cost, naturally enough, but also uncertainty in the absence of a clearly identified market and a definitive regulatory signal from the State of California.

Under the 1996 agreement, carmakers were to provide the CARB with confidential data on their production plans for meeting their 2003 obligations. According to CARB these plans showed that all automakers have the technical capacity to produce vehicles in the required quantities. Yet all of them argued that the production costs of these vehicles were still high and that the limited operating range of electric vehicles, given the foreseeable state of battery technology, would prohibit any sustainable mass market for such vehicles by that date.

Moreover, during the 2000 debate over the program, some manufacturers declared that they were unable to make full use of the opportunities offered by the PZEV option because they could not produce vehicles in sufficient quantity. This meant that these manufacturers' production of pure ZEVs will have to exceed the required 4 percent minimum.

32. This limit was raised to 60,000 at the time of the 2000 program review.

The 2003 ZEV target was again confirmed by the CARB in January 2001, following the biennial review of the program in 2000, but further amendments were introduced to reduce the cost of the program to manufacturers. The principal changes are as follows:

- the required number of "pure" ZEVs is reduced by half in the first years of the program; this number may be reduced further for manufacturers producing other kinds of very clean vehicles;

- the ZEV credit multiplier for an pre-2003 ZEV sale is increased to 4 for sale in 2001-2002 (compared to 2 to 3 before) and to 1.25 for sale in 2003-2005;

- the number of vehicles needed to obtain PZEV credits in the first years is also reduced;

- on the other hand, the 2003 mandate of 10 percent for ZEVs or equivalent credits is increased in stages to 16 percent by 2018.

The result is that manufacturers must produce between 4,650 and 15,450 electric vehicles in 2003, depending on which of the various options they select. The authorities also decided to include sport utility vehicles (SUVs), pickups and light vans, as of 2007, in calculating the required percentage of ZEVs, thereby increasing the calculation base by 50 percent.

According to the CARB, the savings to manufacturers resulting from these changes to the program, compared to its previous version, will amount to between $130 million and $400 million by 2003, depending on the option selected. The CARB has also established an $18 million subsidy fund for consumers as an incentive to purchase or lease ZEVs (up to a total of US$9,000 for a three-year lease prior to 2003, and up to US$5,000 after 2003): these subsidies are in addition to other local or federal incentives.

Content, nature and calculation of ZEV credits

Automakers who offer more ZEVs for sale than required in any model year will earn ZEV credits. The ZEV credits are expressed in grams of NMOG per mile. They are calculated by subtracting the number of ZEVs required from the number of ZEVs delivered for sale, and multiplying this balance by the average NMOG emission standards required for the manufacturer's light-vehicle fleet (cars and light-duty trucks). As well, a credit multiplier can be earned for vehicles delivered for sale prior to 2003 (see above).

A manufacturer can comply with regulations by submitting a certain number of ZEV credits, either earned previously or acquired from another manufacturer. The number of credits required is calculated in the same manner as before.

A manufacturer offering fewer than the required number of ZEVs for sale in any model year must make up the deficit before the end of the following model year, by submitting a certain number of credits. Any manufacturer that fails to offer the required number of ZEVs, or to submit the required number of credits, without making up the deficit within the specified time limit, will be subject to a fine under the Health and Safety Code. This fine applies to any manufacturer selling a new motorised vehicle not compliant with the State emission standards, and amounts to US$5,000 per non-conforming vehicle. The number of non-conforming vehicles is calculated in terms of the ZEV credit deficit.

It has been suggested that the option of paying a fine should be eliminated, and that the manufacturer should be forced to make up any credit deficit by purchasing credits from another manufacturer. The CARB has rejected this proposal, since the manufacturer could then be faced with

paying an exorbitant price to other manufacturers for such credits. A maximum price for credits is then established, equal to the amount of the fine established in the regulations.

Program management

All automakers, except for those in the "small volume" category, are subject to the ZEV mandate and are required therefore to participate in the credit program.

As noted in the introduction, the CARB, as a State agency, has full authority to establish objectives and standards and to implement the credit system, within the framework of federal and State clean-air legislation. The CARB is also responsible for certifying vehicles and verifying the credits submitted by manufacturers.

On the other hand, participating manufacturers have full freedom in negotiating credit transfers.

Assessment

Our assessment relates to estimates of operating costs and the advantages expected of the program. Next we present a summary of the positions of the various stakeholders in the debate over the program's evolution.

Operating costs of the program

The administrative cost of the program itself includes the CARB's measurement and monitoring of pollutant emissions, tracking new developments in engine and battery technologies, and verifying the credits submitted by manufacturers. Among these costs, only the last one is specific to the credit program, the others being an inherent part of any regulatory system. In the absence of more accurate data, it can be concluded that the incremental administrative costs represented by the credit program are fairly low.

Costs relating to accelerated introduction of electric vehicles

In contrast, the costs flowing from the accelerated market introduction of electric vehicles are quite another matter. The CARB (2000a) has estimated the incremental initial cost in 2003 of producing electric, hybrid or PZEV vehicles (vehicle, battery and charger) and the total life-cycle incremental cost per mile travelled[33], compared to the baseline SULEV vehicle. These calculations involve a number of assumptions about the trends in battery technologies and costs, the costs of specific vehicle components and recharge equipment, electricity prices, gasoline prices, vehicle performance, maintenance costs, and inflation and discount rates.

The all-in incremental cost of the battery-powered electric vehicle compared to the conventional vehicle in 2003 ranges from US$7,500 for a "city EV" to US$20,000 for a freeway-capable vehicle. Most of this incremental cost relates to the batteries. By comparison, the incremental cost of a hybrid vehicle is US$3,300, and for a PZEV, US$500. On a per-mile basis, the incremental cost ranges from US$0.082 for a 4-passenger PZEV to US$0.27 for a 4-passenger battery-powered EV. The extra cost of battery-powered vehicles remains significant even under different assumptions involving

33. Total production and operating costs assuming ten-year lifetime miles traveled of 117,000.

higher gasoline prices or longer battery life. Only when volume production is reached (more than 100,000 vehicles per year) will high-efficiency electric battery-powered vehicles achieve costs per mile comparable to those of hybrid vehicles or PZEVs, which will by then be regarded as standard vehicles.

During the three-year period covered by the 1996 agreement (i.e., from 1998 to 2000), electric vehicles sold in California benefited from a buy-down grant of US$5,000 paid to the manufacturers, and financed in equal proportions by the California Energy Commission and by the local air pollution control districts. Manufacturers applied the subsidy as a discount to their ZEV lease or purchase price. In addition to this subsidy, an $18 million consumer incentive program was introduced in 2000.

Expected benefits

The CARB (2000a) has estimated reductions in both direct and indirect[34] vehicle emissions in the South Coast Air Basin by 2010, depending on market penetration by battery vehicles, hybrid and gasoline PZEVs, and SULEVs. The emissions considered are hydrocarbon compounds (NMOG), oxides of nitrogen (NO_x) and toxic air contaminants.

Per-vehicle emissions of NMOG compounds from battery vehicles are 96 percent lower than those of the cleanest gasoline vehicle (PZEV SULEV). Other reductions include 88 percent for NO_x and 86 percent for toxic air contaminants (compared to a hybrid PZEV).

Total (fleet-wide) emissions are estimated for 2010, exclusively for the light vehicle fleet sold between 2003 and 2010, and using different electric vehicle penetration scenarios, with a maximum of 10 percent. The analysis shows that the advantage in terms of air quality is fairly low compared to a base scenario in which no battery vehicles are sold (maximum reduction of 1.91 tons of pollutant per day compared to total emissions of 25.45 tons per day in the base scenario). Only under a scenario where 50 percent of all vehicles on the road were ZEVs by the year 2020 would direct daily emissions be reduced by 30 percent at that time.

The benefits are thus essentially long-term ones, reflecting the time that will be needed for electric vehicles to achieve significant market penetration.

The program has sparked intensive research and development efforts by federal agencies and private business, reflected in the filing of many patents: from 1982 to 1991 the number of patents relating to electric vehicle technology averaged about seven per year, and was declining despite federal funding. Yet this rate rose to more than 50 patents in 1994, the year the LEV I program came into effect, and then to more than 80 per year between 1996 and 1998.

A further advantage lies in the diversification that the ZEV mandate will bring in terms of energy supply.

In terms of CO_2 emissions, the performance of electric vehicles depends on the way electric energy is produced. Currently, an electric vehicle powered relying on electricity produced in California emits about 250 g/mile of CO_2, compared to 300 g/mile for conventional gasoline vehicles[35].

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34. Including the production, processing, transport and distribution of fuel (including electricity).

35. Or 155 grams of CO_2 per kilometer, for the electric vehicle. In Europe, the voluntary agreement negotiated by the European Automobile Manufacturers' Association (ACEA) with the European Union calls for average emissions of 140 g CO_2/kilometer for new vehicles sold in 2008, a less ambitious objective than that initially sought by the European Commission, which was 120 g CO_2/kilometer in 2005.

This advantage is less obvious, however, in comparison to a diesel or natural gas vehicle (less than 270 g/mile) and disappears completely when compared to a latest-generation hybrid vehicle.

The positions of the different stakeholders in the debate

Stakeholders in this debate fall into three broad categories, namely the automakers, the fuel industry, and environmentalists.

Although they have not been presenting a united front, the automobile manufacturers are primarily concerned by what they see as excessive interference in their affairs and an imposed obligation to produce vehicles at a loss. They argue that by introducing partial credits (PZEV), the CARB has implicitly recognised that there is no market for ZEVs and is therefore providing a means to circumvent the regulations. According to them, the CARB should admit this and abolish the ZEV mandate. In response, the CARB insists that it will not withdraw the ZEV mandate and that PZEV credits are a way of allowing manufacturers greater flexibility and encouraging the nascent ZEV industry.

The oil industry has no interest in the ZEV mandate, which represents a threat to its business. It has therefore opposed the program strongly.

On the other hand, the natural gas industry finds the PZEV system very interesting, because it can produce fuel for SULEV vehicles.

The electricity industry is torn between disappointment over the lower required percentage of ZEVs, which means lower future demand for electricity, and the hope of increasing sales through the compression of natural gas.

Environmentalists are divided between those who think that this further flexibility will reduce opposition to the 2003 regulatory deadline, and those who think that allowing credits for gasoline vehicles, even those that are "ultraclean", will end up contributing to increase of total pollution.

Another problem relates to small and intermediate volume automakers that are majority-owned by another manufacturer. The consolidation process that has been underway in the automobile industry in recent years has tended to undermine the CARB's classification of manufacturers by sales volume (large, intermediate and small). Small volume manufacturers are not subject to the minimum ZEV sales percentage, while intermediate manufacturers can meet their obligation entirely through PZEV credits. CARB staff proposed in 2001 that the sales of 2 or more manufacturers should be aggregated if one of them has more than a 50 percent interest in the other(s). This aggregation would become effective in 2003, and its implications for classification and the resulting ZEV obligations would be taken into account from 2006.

Conclusions and comment

It is difficult to say today whether the program is a success, because its full measure cannot be taken until 2003. Till mid-2001, credit transfers have been limited to meeting automakers' voluntary commitment to produce demonstration vehicles over the period 1998-2000 (e.g. the transfer of credits between Ford and Mazda).

From the viewpoint of implementing systems of transferable permits among automakers, the California example shows that such a system is quite feasible, because incremental administrative

costs are low in comparison to a conventional regulatory program. It is on this point that we find the most proposals in the literature on CO_2 emissions certificates (Wang 1994, Albrecht 2000).

The real difficulty in the California ZEV program has been, and remains, to get the automakers to disclose the real costs of research, development and production for electric vehicles. This is, in effect, a classic case of information asymmetry between a public regulator and private economic agents. The disclosure process is not specific to the ZEV credits program, but rather inherent in any industrial regulation policy. It takes strong political will at every turn to find a politically acceptable compromise with the automakers, and that political will must be backed up by a public opinion that is sensitive to local atmosphere pollution.

If we consider the issue of aggregate pollutant and CO_2 emissions reduction, which are not the primary objective of the ZEV program in itself, some considerations can be put forward with regard to the potentials of this kind of program.

In the face of emissions resulting from automobile travel, the policy choice has been to come down heavily on per-vehicle emissions, rather than on car mobility itself. Significant progress has thus been made in terms of total emissions, but reducing emissions from gasoline vehicles has now reached its limit. Future improvements will come more slowly, and will depend on the speed with which electric alternatives suitable for the mass market can be developed. It is unclear whether progress might have been swifter if efforts had focused on reducing automobile travel, but such efforts would have run up against serious social resistance, given the fundamental role of this form of mobility in American society.

Moreover, the shift from very clean gasoline vehicles to electric vehicles is a risk that local pollution hotspots be only moved from urbanised areas to areas where electricity is produced if it is based upon charcoal or oil, despite it is easier to limit emissions from a point source. This shift also does nothing to address the issue of CO_2 emissions, given the way electricity is produced in California, but that was not the objective of the program. This suggests that, from the viewpoint of reducing unit CO_2 emissions, an all-out drive towards electric vehicles is perhaps not the best strategy. However, by sharply reducing emissions from SULEV and hybrid vehicles, the program has been indirectly responsible for significant progress in terms of per-vehicle CO_2 emissions from internal combustion vehicles.

3.4 Exploring new approaches in France

A working group of the National Transportation Council, an advisory body on transport policy in France, has been exploring potential applications of decentralised systems of transferable permits in the transportation sector.

After examining the context in which this work has been undertaken, this paper describes the approaches adopted, together with the principal points of debate.

The context

At the 1992 Earth Summit in Rio, and later at the 1997 Kyoto Conference, the European Union committed itself to an 8 percent reduction in greenhouse gases (GHG) over the period 2008-2012, compared to 1990, and subsequently issued "bubble regulations" distributing the required effort among its member States. Accordingly, France is to reduce emissions to their 1990 level over the period 2008-2012.

Among greenhouse gas-emitting activities in France, the transportation sector is seen not only as having the most significant emissions (22 percent of GHG in 1997) but as presenting the greatest threat, by far, of higher future emissions: 10.3 percent higher in 2010, compared to low growth or even decline in other sectors[36]. In the transportation sector, GHG emissions are produced primarily by the combustion of fuels in road transport (84.3 percent of the sector's emissions in 1997), and in a subsidiary manner by air transport (10.8 percent, including international refuelling tanks).

Kyoto represents a first, and modest, stage of reductions (- 5 percent world-wide), which is to be followed by further stages for which more drastic reduction targets must be defined. The question of the transport sector's responsibility for GHG emissions, compared to other sectors of activity, thus arises. Should the transportation sector be allowed to continue with growing emissions, while other sectors are called upon to make additional efforts, or should ways be sought to curb transport activity?

The potential advantages of regulation and supply-side policies have been widely explored in a report on the French program to combat climate change[37]. Other incentive mechanisms involve economic instruments, taxation on one hand, and transferable permits on the other hand, which have the advantage, compared to other mechanisms, of minimising the total cost to society of reducing emissions.

With respect to taxation, the currently proposed level of the carbon tax, an average of 0.07 Euro per litre of fuel, represents about 7 percent of the final price paid by the consumer for super-grade gasoline. In other words, current automotive fuel taxes are 7 to 10 times higher than the proposed carbon tax. The behavioural effect of a carbon tax at this level is likely to be very weak. If there were to be a more stringent "post Kyoto" tightening of objectives, the tax would have to be raised sharply: it would have to be at least three times as high, ceteris paribus, in order to achieve a reduction of 10 percent in automotive fuel consumption, given what we know about fuel price elasticities. The tax hike would have to be even higher if accelerating economic growth, such as we have had recently, were to increase mobility, and hence fuel consumption, through the income effect.

The "tax revolt" of September 2000, on the heels of sharp oil price hikes in 1999 and 2000, forged a coalition of road users, private or professional, farmers and fishermen, and forced the government to make concessions on fuel taxation. This experience shows that a drastic increase in such taxation is sure to encounter social and political obstacles.

Moreover, the speed and the magnitude of last oil price increase in 1999-2000 suggest the possibility of an equally swift and severe collapse in this price, net of taxes, as reflected recently. Under such a scenario, the price effect of a carbon tax would be completely wiped out.

In the face of these arguments, and given the difficulties in controlling the effective final price and the doubtful acceptability of imposing a further levy on a product that is already heavily taxed, a working group undertook to explore further the feasibility of introducing decentralised markets for permits in the transportation sector, within France or in Europe. The working group originated in the National Transportation Council and brought together representatives of users and the transportation industry, as well as outside experts (CNT, 2001).

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36. A specific feature of France's energy supply is the already very high share of nuclear electric generation, which produces no GHG but generates radioactive wastes. Consequently, France has hardly any technological maneuvering room for reducing GHG emissions in sectors other than transportation.

37. See " Programme National de Lutte contre le Changement Climatique 2000/2010", Interministerial Mission on the Greenhouse Effect (MIES).

Because they represent a new departure in terms of the transportation sector[38], the ideas put forth by the working group as to the operational nature of transferable permit systems in the transportation sector do not represent definitive conclusions. Rather, they suggest routes for further exploration and evaluation, primarily from a qualitative viewpoint: the final diagnosis will require more exhaustive study.

As well, the fact that some areas of the transportation industry (e.g. road hauling) are exposed to vigorous international competition means that any measures affecting them would have to be harmonised throughout the European Union, as a minimum.

New approaches explored by the working group

The group immediately found itself facing a trade-off in attempting to design a set of economic instruments for reducing emissions from multiple mobile sources. On the one hand, a more effective incentive to emission-reducing behavioural change could be obtained by applying the permits downstream in the fuel cycle, as close as possible to the final consumer, rather than increasing fuel taxes (see below the discussion on effectiveness of price-signal). On the other hand, if the intent is to minimise the reduction program's administrative costs, given the existing fuel tax system, then the permits should apply upstream in the fuel distribution process.

The working group therefore explored several different approaches:

- An "upstream" system, where the permits are applied to producers and importers of fossil fuels: such a system has the advantage of covering all points of consumption (and not just the transportation sector). It could also be combined with "downstream" application, involving businesses that are bulk consumers of energy.

- A set of "downstream" permits involving both private and public agents in passenger transportation. The agents examined, in turn, are public passenger carriers, the local transportation authorities, automakers and private motorists.

- A set of "downstream" permits involving road transportation of goods.

All of these approaches are summarised in Table 1 and Table 2, showing in each case the physical basis of the permits (or the intended target), the initial allocation, the trading mechanism, and the major advantages and drawbacks.

In most cases, except for the transport authorities and automakers, the physical basis is the carbon content in the fuel consumed, and the permits would apply to fuel by the litre (proportionately as a function of the carbon content of each type of fuel, gasoline or diesel). This arrangement allows for wide trading, extending to sectors other than transportation. Assuring broad coverage for the permits market is a way to achieve distributional optimisation of the permit price, and hence to diminish the cost of the reduction to the various agents (see below).

The quota principle (cap and trade) was adopted in each case, except that involving automakers, where averaging was selected, based on target CO_2 value in grams per kilometre.

38. Apart from proposals for emission permits for new vehicles, involving automakers (see this section devoted to the ZEV program).

Free allocation of permits was adopted in all cases, in order to make each system socially more acceptable. For road carriers of goods or passengers, the level of this allocation would be similar to that of the fuel tax exemptions in place today. For private motorists, the basis of allocation could vary in light of social equity objectives: this could involve allocations to individuals and not only motorists[39].

Of course, with downstream permits, there is the issue of the costs of introducing and managing the system, which have been described for the time being only in qualitative terms. To moderate the impact of these costs, permit transactions and verification should be closely integrated with the current fuel retailing system based on the use of bankcards. These costs would include:

- Changing the software loaded in the automatic bankcard devices on the fuel pumps, to recognise the permits system (checking the balance, debiting accounts).

- The manufacture and distribution of "smart cards", or integration of microcoding software into existing smart bankcards as they come up for renewal.

- A specific information campaign for launching the new system (in addition to the campaign that would undoubtedly have to be undertaken before introducing any emissions control measure).

Finally, the trading mechanism raises the question of the degree of openness of the market on which agents are to trade their permits: should it be sector-specific (transportation) or inter-sectoral, should it be domestic or international?

39. A similar proposal was made by a "popular initiative" in Switzerland in March 2000, aiming at cutting road traffic by half. An "ecobonus" was proposed that is to say a levy charged on car utilization and reimbursed to the population at large in equal shares. Those who travel more than the average would have to pay more while those travelling less would be refund.

Table 1. Summary of approaches explored for marketable permits in the transportation sector

	Physical basis	Initial allocation	Trading mechanisms	Advantages	Drawbacks
Upstream permits					
	Carbon introduced into the economy by fossil fuel producers and importers	Free allocation, or Purchase of permits on the international market	Trading with other participants on an open permits market*	Total coverage of carbon introduced into the economy. Least costly permit system to administer.	If allocation is free: - price signal may be less effective (marginal price of the permit is "drowned" in the price to the final consumer); - need for a subsequent tax on operators' profits
Downstream permits applied to goods transportation					
Trucking operators	Carbon in fuel consumed by trucks hauling goods (e.g. permits per litre of diesel)	Free flat annual allocation identical for all vehicles	Trading among participants on an open permits market*	Direct incentive to optimise tons-kilometres / carbon consumption ratio	Implementation and administrative costs

** An "open permits market" is a broad permits market that extends to the entire transportation sector or other economic sectors, and operates at the national, European or even world level.*

Table 2. Summary of approaches explored for marketable permits in the transportation sector

	Physical basis	Initial allocation	Trading mechanisms	Advantages	Drawbacks
Downstream permits applied to passenger transportation					
Road passenger transportation operators	Carbon in fuel consumed by commercial passenger vehicles (e.g. permit per litre of diesel)	Free flat annual allocation identical for all vehicles	Trading among operators or with other participants in an open permits market*	Direct incentive to optimise the passengers-kilometres / fuel consumption ratio	Introduction and administration costs. Need to offer similar incentive for private cars. Need to involve the transportation authorities
Transportation authorities	Urban GHG emissions from routine travel	Free allocation according to technical characteristics of transport supply, demographics and exogenous determinants	Trading on an open permits market*	Direct incentive to the authorities to optimise supply policies and demand management from the GHG emissions viewpoint. Limited introduction and administration costs	Methodological problems in measuring urban "greenhouse" performance.
Automakers	Unit vehicle CO2 emissions (e.g. grams of CO2/km)	Free flat annual allocation identical for all vehicles sold, against a target set by the authorities (averaging principle)	Trading among automakers	Guarantees a unit emissions standard. Introduction and administration costs limited by the small number of participants	Market is limited to automakers. Risk of distorting world trade if market is restricted to the European Union.
Private motorists	Carbon in fuel consumed by private cars (permits per litre of fuel)	Free flat annual allocation identical for all vehicles.	Trading among participants in a domestic or open permits market*	Strong incentive to optimise individual travel in terms of the "greenhouse" objective.	Introduction and administration costs

* An "open permits market" is a broad permits market that extends to the entire transportation sector or other economic sectors, and operates at the national, European or even world level.

What should be the scope of the permits market?

This issue raises two points, one regarding the effectiveness of transfers, the other regarding the control of effort specifically made by the transport sector.

The effectiveness of transfers is the basic condition for minimising total reduction cost and depends on the heterogeneity of reduction costs among participants. This heterogeneity would be particularly effective regarding the different potential adaptations of mobility between private motorists living in dense urbanised areas and those living in rural areas: in that case a specific and closed market restricted to private motorists may be effective from this point of view. Regarding transportation authorities, since they generally have the same profile of density and modal split within each country or even region, the effectiveness of transfer[40] would imply their participation to broader market involving other kind of participants. This is all the more true for road passenger or trucking operators: one obvious idea would be to involve rail operators, which seems however difficult (see below).

In the case of a domestic market restricted to the transport sector, the level of prices on that market would indicate their relative scarcity and make it possible to anticipate future adjustments. For example, the greater the total number of permits initially allocated, the lower their price will be. Conversely, if the number allocated is too low, the price of the permits will be high, but it could be capped at a tax "t", at which price the regulatory authority would redeem the permits.

Through a comparison of the permit price on the domestic transport sector market with that of carbon-referenced permits on a domestic or international inter-sectoral market, the relative effort being made in the transportation sector can be compared to that of other sectors. If its effort is inadequate, this means that other sectors will have to make up the gap, or that the regulatory authority will have to finance it by purchasing permits on international market.

By keeping the market closed, the regulatory authority can thus decide to shift the burden to one sector rather than another, for social or political reasons. Generally speaking, if permit allocation is too restrictive or too lax, or if the tax "t" is not properly calculated, the marginal costs of reducing CO_2 emissions will not be equalised across sectors. Such distortions reduce the effectiveness of the system. They can therefore be regarded only as transitional measures.

The market for tradeable permits should thus ultimately be an open one, *i.e.* all emitters of GHG in a country's different economic sectors should be able to trade their rights, even internationally.

The tax-versus-permits debate

It is clear that the least costly mechanism, in terms of introducing and administering it, is to extend current fossil fuel taxation to take account of the carbon tax concept. The administrative cost criterion, however, needs to be looked at in light of two other criteria: the incentive for behavioural change, and public acceptability. Here, the system of transferable permits with free allocation has some undeniable advantages. First there is the incentive factor: agents selling their unused permits receive an immediate and tangible monetary benefit. And in terms of acceptability, the system avoids placing a further fiscal burden on a product that is already heavily taxed.

40. Apart from methodological measurement problems mentioned in Table 2

One of the major challenges in designing permit systems is how best to reduce implementation and transaction costs. Only by assessing the total cost of reduction (*i.e.* implementation, administration and transaction costs plus reduction costs borne by the participants in the system, less eventual fiscal revenues), and adding in the social and political cost of the measure, is it possible to choose one mechanism over the other, or a combination of both (taxation and permits).

The debate over the effectiveness of price signals and their point of application

In analysing the effectiveness of price signals, we must start from the perspective of changing behaviour by inducing different agents to reduce their consumption through different means and to different levels: individual agents will reduce their consumption to the point where the cost of the last unit of pollution emitted is equal to the advantage derived from the activity producing that unit. Permits and taxes thus have exactly the same marginal effect on the agent to whom the permit or tax is applied.

Depending on where the permit is applied in the fuel cycle (production, distribution and consumption), the system's effectiveness will differ depending on whether the permits are handed out initially free of charge. If permits are auctioned, the price signal will in principle be fully transmitted to the final consumer.

On the other hand, if upstream permits are distributed free there is a risk that as the original price signal from the permits is transmitted through the producers, it will simply be averaged into costs, and will thus be lost sight of ("drowned") in the price of fuel to the final consumer. If the price signal is in effect blurred, the transportation sector may well end up purchasing reduction credits from other sectors, or from abroad, at greater cost for the community than it would have incurred through its own efforts.

This suggests that, if we want the transportation sector to reduce its emissions, and if permits are to be distributed free for reasons of public acceptance, the permits system should be decentralised downstream, rather than confined upstream. This in turn means applying the permit, as far as possible, at the point where it is most likely to effect behavioural change: to motorists, for example, who may decide to modify their travel habits, or to transport operators, who can revise their business plans.

Conclusions and comment

To bring consistency to the various approaches explored, we must focus on the mechanism for co-ordinating the efforts of the participants involved.

Whether we are speaking of the system of upstream permits or a series of downstream permit systems, the concept of carbon content offers a common unit of allocation and exchange. Taking this as the common physical basis makes it feasible both to ensure co-ordination among the participants and to make the permits as universally tradeable as possible with other sectors, which is a condition for minimising the total cost of reduction.

In contrast, the mechanism proposed for the automakers does not allow emissions per kilometre to be converted into CO_2 permits. Yet the variety of vehicle models and the diversity of

automakers, with the differing adaptation costs they imply, should make for a reasonable level of trading within this sector[41].

What the local transport authorities must do, as we have seen, is to develop a mechanism for calculating emissions from routine or commuter travel in their areas of jurisdiction. Under this condition, the results can be translated directly into CO_2 permits, and the universal market for permits could then be accessed.

As for road operators, their market suffers from destructive competition of a multiplicity of small operators and they feel themselves threatened by opening of the EU to lower-wage countries. Thus they are in a weak position when negotiating prices with those who demand transport services. Why not involve the latter in a permit scheme? The extension of the downstream permits previously designed to those who demand transport services is impossible, given the way sendings are managed by hauliers: a same vehicle can be used within a single tour to serve different clients. It would be then impossible to fairly allocate the quotas to the clients. An alternative option would be targeting big shippers (e.g. distributors) and involve them in voluntary agreements. This could be a first step to get more information on the potentials of emission reductions by the shippers before implementing more incentives.

Finally, it has proven difficult, with these downstream permit systems, to incorporate rail carriers, whose energy requirements are mainly in the form electricity. In fact, the only way of bringing them in would be to design a different type of permits system where the initial allocation would be calculated using an efficiency standard prorated for the service provided (either in passenger-km, in the case of passenger travel, or tons-km, for the transport of goods). In such a program, rail operators would be net sellers to road transport operators. This option, however, has not yet shown itself feasible, since service provided is measured only by surveys, and these are unlikely to withstand a legal challenge.

One possible solution to the problem of integrating rail operators into the permits system based on carbon consumption would be for the regulatory authority to set an overall carbon consumption quota for the subsector concerned (passenger or goods carriers, rail and road). A portion of the quota would then be distributed as a basic allocation to rail operators, prorated to the service they provide, and the remainder would go to road carriers of goods, using the mechanism described in Table 1. Road operators needing permits would then buy them from rail operators. Jancovici (2001) has estimated the value of the permits that the SNCF (the French national railway) could sell at 300 millions Euros a year, with a ton of carbon at 76 euros.

Any shift from control based primarily on taxes to one based on rationing by imposing rigorous quotas on final consumers would represent a real social shock, involving a resource that is essential to present-day lifestyles in developed countries, and for which there is no credible substitute over the medium term, at least in some situations. Dealing with this shock remains a major challenge.

As for the consequences of this work, it could be said that it has essentially an educational role to play towards both the transport operators and the administration in charge of transport activity. The main operators concerned, road hauliers and airlines, are currently in difficult economic situations: for road operators this has already been mentioned and for airlines this stems from strong world competition and current slowing down of demand growth. In this context it is difficult for them

41. The literature has examples of proposals for permits to automakers, combining unit emissions and kilometer traveled by vehicles (Albrecht, 2000; Winkelman et al, 2000) or even vehicle ownership (Walton, 1997). See also the section devoted to the California ZEV program.

to consider control of their activity either by reinforcing the current fuel taxes (even implementing them for aviation) or introducing TPs. On the other side the administration faced a "fuel tax revolt" in September 2000 which shows that the conventional instrument reaches its limits. There is recognition of the need to find ways to get out of this environmental deadlock. This work has contributed to open the debate and to give an impulse to research on introduction of TPs within the sector.

4. Conclusions and lessons

Feasible systems: protecting environmentally sensitive areas and reducing unit vehicle emissions

A system of permits applied to traffic of mobile sources in limited geographic areas is technically feasible at an acceptable financial cost, as the Ecopoint program in Austria has shown. In technical terms, this system is similar to ongoing or near future electronic road pricing systems and could be added to at low cost.

Such a system would seem, a priori, to be adequate for protecting sensitive areas of the local environment (pollution, noise, road safety). The demonstrated limitations of the Ecopoint program, inherent in its original design, suggest that if the vehicles involved are to be covered fully the protected region must be one where points of entry and exit are few and readily controllable. A trade-off would have to be found between the number of points to be controlled and the size of the area to be protected. Of course such TPs market focussed at geographic areas (one market for one area) would need the participation of operators of other modes such as rail in order to benefit from heterogeneity of reduction costs and to foster new modal split. Similar potential systems such as "CO_2 credit card" and linked rail-road transit permits (LRRTP) are briefly described in the OECD-EST study on Alpine freight transport (OECD, 1999).

The California ZEV program, although it will become significant only in the near future, shows that a system of credits applied to unit vehicle emissions and tradeable among automakers is quite feasible: its incremental administrative costs are in fact negligible compared to the normal expense of monitoring and enforcing regulations.

Clarity, simplicity and pragmatism can help permit programs succeed

The importance of these criteria for transferable permit programs in general has long been recognised. Their validity first became apparent in an analysis of the conditions for success with the gasoline lead phase-down program in the United States (Hahn and Hester, 1989). This finding is reinforced by an analysis of Austria's Ecopoint program and California's ZEV program. In all three cases, the physical basis is clearly identified (grams of lead per gallon, grams of NO_x per kilowatt-hour, and grams of NMOG per mile, respectively).

The simplicity criterion relates to the rules for managing transfers, which must be facilitated as much as possible if the theoretical advantages of the tradeable nature of the permits are to be fully realised. This is the case with the lead reduction program and the ZEV program, where there is no need for the regulatory authority to become involved in trading among program participants[42].

The pragmatism criterion relates to the design of the program: its various aspects must be able to evolve in light of discrepancies that may emerge between objectives and cumulative

42. The Ecopoint program does not involve transfers, and was not designed for them.

understanding of the adaptation costs borne by program participants. Some good examples of such pragmatism can be found in the lead phase-down program, with its different trading and banking phases, and in the ZEV program, where changes to the credit formulas provide automakers with more opportunities to adapt.

On the contrary, such pragmatism would seem to be lacking in the Ecopoint program which, at least in its current configuration, is in something of a crisis. If a program like this were to be established for the European Union, any amendment would require reaching agreement among the member States, the difficulty of which is readily apparent given their divergent interests.

It is all the more obvious that pragmatism is only possible when the regulatory authority enjoys strong political support, and has sufficiently broad powers.

Political will and possibilities for low-cost adaptation

If a regulatory authority is to negotiate successfully with market participants who are often organised in professional groupings with means for exerting considerable political pressure, that authority will have to have effective powers, backed by a strong political commitment.

When low-cost adaptation is possible there is less need for strong political will. This was the case with the gasoline lead reduction program, since there were affordable technological solutions available for replacing lead, and the consumption of leaded gasoline was already on a downward track because of renewal of the automobile fleet.

On the opposite, the commitment of public power will have to be all the stronger if the range of low-cost adaptation possibilities is limited. An analysis of the ZEV program shows that the dynamic equilibrium between political will and pressures from the automakers has been a constant factor in the program's development. The 1996 reform, which pushed back the ZEV deadline from 1998 to 2003 and established a voluntary agreement, was interpreted by some as a victory for the automobile manufacturers, whose arguments about the number of industry jobs at stake found a certain political resonance. On the other hand, ecological pressure groups, buoyed up by strong public sensitivity to local air pollution, weighed in heavily in public debate over the program, and in keeping the CARB to its ZEV objectives.

It has been argued that there is no proof of such political will when it comes to reducing greenhouse gas emissions, or, at least, that whatever will exists may be insufficient in light of the scope of the changes required. There is no doubt that public opinion is becoming increasingly aware of the gravity of the situation. Yet because the consequences of climate change are seen as far off and to some extent uncertain, and the social costs of imposing abrupt lifestyle changes in developed countries are deemed unacceptable, the required policy decisions have been delayed.

What role can transferable permits play in GHG reduction programs in transportation sector?

As noted in our analysis of the California ZEV program, technical advances with gasoline engines are reaching their limit, and the electric vehicle is not likely to offer a suitable alternative for significantly reducing CO_2 emissions, given the way electricity is currently produced.

Governments cannot, therefore, rely on unit vehicle emissions for reaching CO_2 reduction objectives, but will instead have to focus on vehicle use. The various incentives available for influencing demand are taxes, permits, or a combination of both.

The terms of the debate are now clear. Tax and permit systems have the same allocation efficiency. Since fuel taxes are already in place, extending them to a CO_2 tax would be less costly to administer for multiple mobile sources than a system of permits applied to those sources.

However these duties would have to be much higher than they are at present if they were to meet the objective of reducing emissions. The "tax revolt" experienced in September 2000 in several European countries, as sharply higher oil prices pushed up gasoline prices that were already highly taxed, highlights the limits on public acceptance of significant further increases in these taxes.

On the other side, permits allocated free of charge are seen by agents as a means of avoiding an additional tax. Moreover agents selling their unused permits receive an immediate and tangible monetary benefit. These are arguments in favour of a potentially greater acceptability of permits with free allocation when compared to new taxes.

For this reason, new approaches have been explored for downstream decentralisation of permits in the transportation sector as a possible alternative to CO_2 taxation[43]. This would be a way of seizing the advantages of permits, namely the certainty of achieving the quantitative objective and the further incentive to final consumers to reduce emissions that is inherent in allowing them to sell any unused permits. Moreover, these downstream permits have been designed with a physical basis that is simple and fungible: they can be fully integrated into wider permit systems targeting all sources other than transport, *i.e.* national and even international markets for CO_2 permits.

Nevertheless, this option poses the formidable question of administrative costs, not to mention the social and political challenge of introducing what could well be seen as a form of rationing.

These considerations point to at least four routes for further research:

- Designing downstream permits systems to keep administrative costs to a minimum, as outlined in the preceding sections: technical progress with onboard electronics over the last 20 years has made it possible to introduce systems that would have been previously inconceivable, such as electronic tolls or Ecopoints, and that now hold significant promise for addressing this question.

- Examining the social and political feasibility of such a rationing program.

- Simulating the operation of downstream permits markets (e.g. trading over-the-counter or on a stock exchange), which could also give indications on possible market power and price distortion effects.

- Assessing the total social cost of reductions (which is equal to implementation and transaction costs plus the costs borne by agents, less eventual fiscal revenues), as a basis for choosing case-by-case between taxes or permits.

TPs, taxes and voluntary agreements

The analysis of the previous case studies showed differing ways of coexistence of these instruments.

43. Similar proposals are mentioned for The Netherlands and Sweden in the OECD-EST case studies (1999)

In the US lead phase-down case the lead rights program was the unique instrument used, however with variations in the trading and banking options.

In the Ecopoint case, the quota system coexists with the ongoing fuel tax, an incentive to reduce fuel consumption and thus harmful emissions, and with the European regulations on unit trucks emissions (EURO standards). As already mentioned, given the importance of the Austrian route for European truck traffic, the Ecopoint system has probably had a supplementary effect in accelerating the impact of the overall EURO standard program.

In the European context, with regard to the general issue of fuel consumption, a voluntary agreement (VA) of car makers coexists with the conventional fuel taxes. They are both incentives on the supply side, on one hand, on the demand side, on the other hand, to switch to cleaner cars. However VAs are often seen as a willingness by the industry to escape from tighter regulation or tax, by realising part of the pollution reduction: this is possible when the threat of the public authority is not credible, especially when not backed by a strong public opinion as in the case for the GHG issue (Bureau, 2000). The proposal for TPs on car makers aims at guaranteeing the achievement of a specific target in unit-vehicle emission.

Further, in the design of downstream permits on fuel consumption by the French working group, TPs would coexist with current fuel taxes. In this framework fuel taxes would keep their current status while TPs would specifically address the issue of CO_2 emission.

In the California ZEV case study, within the general credit framework, VA has been established in 1996 with automakers agreeing to produce a certain number of electric vehicles between 1998 and 2000. This VA played a specific role to solve temporarily a difficulty in the ups and downs of the negotiation between CARB and the automakers, pointing to the varying credibility of the public authority as mentioned above.

These remarks show that coexistence is possible when needed and must be designed pragmatically.

In addition the case studies showed short term operation of TPs. Some TP schemes may last only a few years, like the lead phase-down program with a duration of five years from 1982 to 1987, or the Ecopoint system since 1992 until today under its current configuration. This shows that TPs can be implemented and used within a short range to ease and speed up a change in technology. However this does not mean that TPs are confined to short term policy. The ZEV credit program is open since 1994 only but may last several years, even decades, depending on the level of market diffusion of electric vehicles. Moreover, in relation to the very long term horizon of the GHG issue, short term implementation of TPs addressing directly fuel consumption of transport end users would result, as for the tax, in several behavioural changes including reduction in trips or in VKT, modal shifts and land use patterns. These changes, and especially the latter, have long term consequences on mobility patterns. Maintaining these behavioural changes in the long term also implies keeping continuous operation of related TP schemes in the long term.

Other unexplored possibilities

There has been little investigation of possible strategies relating to land use, perhaps because transportation and urban planning sciences have not managed to clarify the relationship between transportation and land use. Yet the literature contains proposals for applying marketable permits to real estate developers in terms of the travel volumes that their projects will generate (see for example Ottensmann, 1998). Such investigation should also take into account the experiments with transferable urban construction rights that have been launched in the United States in the beginning of

the 20th century, as well as the environmentally significant initiatives undertaken during the 1970s and 1980s in France and New Zealand.

Finally, there has been little attention to the supply side of infrastructure, except on the part of local transportation authorities. The scope of the behavioural changes required suggests that governments should not rely solely on the adaptive capacity of final transport users in terms of the current state of supply. Major investments will have to be made, particularly in rail infrastructure, at the same time as permits systems are introduced: it is only from a post-Kyoto perspective, *i.e.* in moving towards the real objective of reducing CO_2 emissions by half, that permits might become a significant source of funding for such infrastructure.

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Chapter 6

A NEW AREA FOR APPLICATION OF TRADEABLE PERMITS: SOLID WASTE MANAGEMENT

by

Roger Salmons
CSERGE, University College London

This report considers the potential for using permit trading as a policy implementation mechanism in the field of solid waste management. The focus is on waste generated as a result of consumption by households, and on waste directly related to this consumption (e.g. waste transit packaging discarded by retailers, etc.). This broadly corresponds to municipal waste.[1] While this is only one part of the overall waste stream it is a significant policy issue, both in its own right and within the broader context of resource productivity. It also offers plenty of scope for the application of permit-trading systems.

1. The waste management system

1.1 System definition

The definition of the waste management system is closely linked to the issue of the appropriate goal for waste management policy. A range of alternative policy objectives have been proposed by regulators, waste managers, environmental groups and other commentators. These include the conservation of resources; offsetting distortionary resource policies; reducing lifecycle externalities; promoting the concept of extended producer responsibility; stimulating demand for secondary materials; and increasing recycling rates. However, Macauley & Walls (2000) argue that, either these objectives should be addressed directly by initiatives in other policy areas, or that they should be considered as contributing to the primary goal of waste policy which is the cost-effective management of the waste stream.

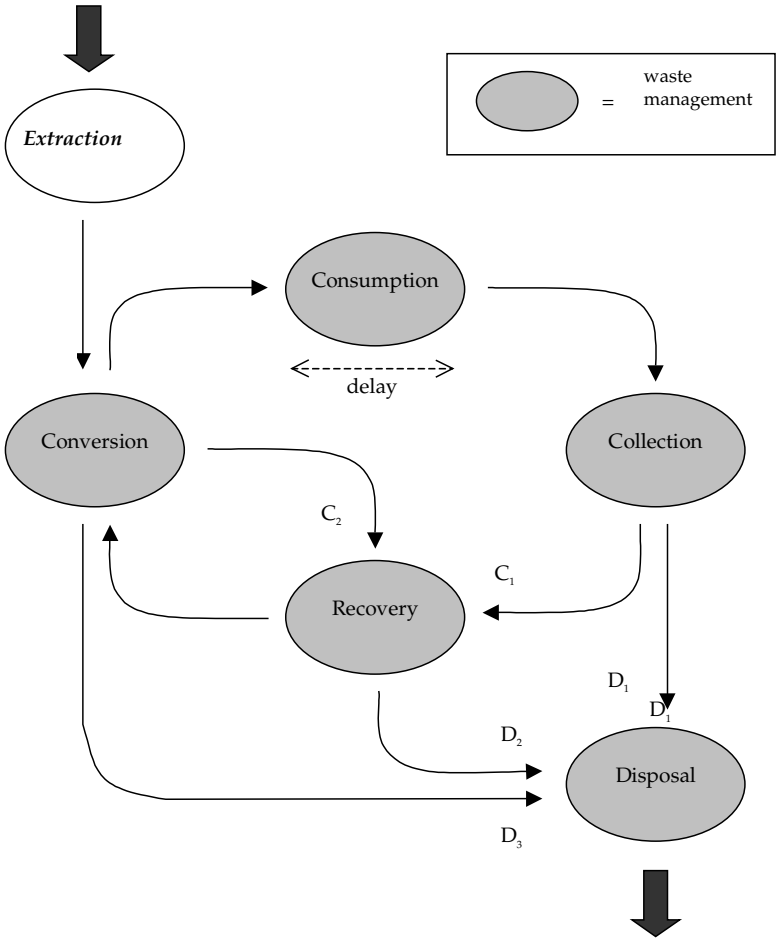
This goal still leaves plenty of scope for the definition of the waste management system. For example, the total social cost will be affected both by the decisions that are made regarding the treatment of waste (i.e. landfill versus incineration versus recycling, etc.), and by the amount of waste that is generated. Pearce & Turner (1993) advocate taking a holistic perspective when making waste management decisions, and in line with this recommendation it is assumed that the waste

1. The EU Landfill Directive defines municipal waste as "waste from households, as well as other waste which, because of its nature or composition, is similar to waste from households".

management system comprises all of the processes of the wider materials flow system, with the exception of the extraction and processing of natural resources.[2]

Figure 1 provides a simple schematic representation of the waste management system. The five system processes are linked by a series of physical flows. For example the flow of virgin materials is denoted by V; the flow of final products by Y; the flow of waste arisings by W; and so on. Between them, the five processes cover all levels of the waste management hierarchy that is set out in the EU Framework Directive on Waste.[3]

Figure 1: Waste management system



2. Under this definition, waste policy would be concerned with the flow of virgin material (i.e. the amount used), but not the process which generates that flow. However, this is not to say that the waste management system is unaffected by policy interventions relating to extraction and processing of resources.

3. Directive 91/156/EEC defines the following priority for managing waste: minimisation; re-use, recycling and reclamation; incineration with energy recovery; disposal to landfill.

Annex 1 provides a breakdown of municipal waste arisings for selected OECD countries. Unfortunately, the figures are rather out-of-date – relating to the early-mid 1990s. However, the general picture that they paint remains valid. While there is some variation in the composition of the municipal waste stream (flow W in Figure 1) between countries reflecting cultural differences, the overall pattern is fairly consistent.[4] In particular – excluding Korea – biodegradable waste accounts for between 54% and 70% of the total. However, a very different picture emerges when one looks at the treatment of municipal waste, with significant differences existing between countries. For example, in the Netherlands only 20% of municipal waste is sent to landfill (D_1), while 40% is either recycled or composted (C_1). In contrast, the corresponding figures for the United Kingdom are 83% and 7%!

1.2 Environmental impacts of waste disposal

A recent report to the European Commission (CEC, 2000) assesses the environmental impacts arising from the disposal of municipal waste to landfill and by incineration. It divides the impacts into three broad types: those arising from emissions to air, water and soil; those arising from the location and operation of the facility; and those arising from the displacement of other sources of energy production. Table 1 provides a summary of the potential *emissions* to air, water and soil arising from incineration and landfill, together with the resultant environmental impacts.

The flue gases resulting from the incineration of municipal waste can contain a variety of pollutants, which have a wide range of environmental impacts. Flue gas emissions can be controlled by the use various treatment processes. However, while these can significantly reduce the emissions of the pollutants to air, they only result in them being transferred to another media. Wet cleaning systems result in discharges of wastewater that contain dioxins and heavy metals. Other cleaning systems generate solid residues that are contaminated with inorganic salts, dioxins and heavy metals. The incineration process itself produces residual solid waste that may contain relatively high concentrations of inorganic salts.

The main impacts arising from the disposal of municipal waste to landfill relate to the emissions of landfill gas and the generation of leachate. The two main components of landfill gas are methane and carbon dioxide, both significant contributors to global warming. The gas may also contain a large number of different volatile organic compounds (VOCs) that have adverse health effects.[5] Emissions may be reduced by the introduction of collection systems, with the gas being flared or used to recover energy. However, the combustion process can lead to the formation of dioxins and other pollutants. Leachate generated in the landfill can be emitted to soil and groundwater, with potential impacts on health and ecosystems. The quantity and composition of leachate will depend on the amount of rainfall, the type of landfill cover that is used, and the composition of the waste. Emissions of leachates can be reduced by the introduction of collection systems and the lining of the site.

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4. Caution should be exercised when making international comparisons of municipal waste data. Definitions may vary between countries (e.g. the figures may relate to waste leaving the household, or waste arriving at landfills), as may data collection methodologies.

5. A recent study of all landfill sites in the United Kingdom (Elliot *et al*, 2001) found small excess risks of congenital abnormalities and low birth weights in populations living within 2 km of sites. However, there are currently no causal mechanisms to explain the findings.

Table 1. Environmental effects of emissions from landfill and incineration

		Impacts					
	Medium	Health effects	Reduced agric. yield	Forest die-back	Damage to buildings	Climate effects	Eco-system
Incineration							
CO2	Air					*	(())
CO		(*)				*	
NOX		*		*	*		(())
SO2		(*)	*	*	*		((*))
HCl, HF		((?))	(())	(())	(())		((?))
Dioxins		(*)					((*))
Heavy metals		(*)					((*))
VOCs		(*)	(())				
Particulates (PM10)		*			*		
Dioxins	Water	((?))					((?))
Heavy metals		((?))					(())
Salts							((?))
Landfill							
CH4	Air					*	(())
CO2						*	(())
Dioxins		(*)					((*))
VOCs		(*)	(())				
Dust		((?))			((?))		
Leachate	Soil & water	((?))					((?))

* Measurable impact
(*) Partly measurable impact
((*)) Non-measurable impact
(()) Non-measurable minor impact
((?)) Non-measurable uncertain impact
Source: CEC (2000), Tables 5.1 and 6.1

The impacts arising from emissions depend on the level of activity at the facility (i.e. the amount of waste that is processed). Other impacts arise from its existence, and are independent of it level of operation. Many of these disamenity impacts are common to both incinerator plants and landfill sites – i.e. noise, dust, odours and visual pollution. Additional impacts arising from landfill relate to litter and the presence of vermin.

Energy recovered during the incineration process, or from the methane collected by landfill gas collection systems will result in a reduction in the production of energy from conventional sources, and a hence a reduction in the environmental impacts associated with these sources. Displaced impacts include global warming, depletion of fossil energy resources, and wastes arising during the fuel generation process. The scale of these displacement impacts will depend on the energy source that is displaced.

The magnitude of the environmental impacts will vary between individual facilities depending on the composition of the waste that it receives; the technological characteristics of the facility; and the efficiency of any energy recovery system. The technology will reflect the environmental standards that were in force at the time of its construction, while the recovery efficiency will depend on the use to which the energy is put. For example, if the energy recovered is used to generate electricity, the recovery rate is only around 25%, but if it is used to generate both electricity and heat (i.e. in CHP plants) the recovery rate rises to around 83%. These differences are graphically illustrated in Table 2, which shows the monetised values of the external costs for landfills and incinerators under a range of different assumptions about the abatement technologies and the energy recover systems that are used.

Table 2. External costs of landfill disposal and incineration (EURO per tonne of waste)

Impact arising from	Landfill scenarios		Incinerator scenarios		
	L1	L2	I1	I2	I3
Greenhouse gas emissions	5.0	8.0	0.8	0.8	0.8
Air pollutant emissions	0.1	0.0	20.0	50.0	69.0
Leachate emissions	0.0	1.5	0.0	0.0	0.0
Sub-total	**5.1**	**9.5**	**20.8**	**50.8**	**69.8**
Disamenity	10.0	10.0	8.0	8.0	8.0
Pollution displacement	-4.0	0.0	-71.0	-21.0	0.0
Total	**11.1**	**19.5**	**-42.2**	**37.8**	**77.8**

L1: Modern containment landfill satisfying latest EU Directive (EC/31/1999), with leachate collection system, and landfill gas collected to generate CHP.
L2: Old landfill site without leachate or landfill gas collection systems
I1: Incineration plant meets the requirements of the proposed EU directive on incineration of waste (Common Position (2000/C 25/02), with recovered energy used for CHP
I2: Incineration plant meets the requirements of the existing EU directive on incineration of waste (89/369/EEC), with recovered energy used for electricity generation only
I3: Incineration plant does not meet the requirements of the existing EU directive on incineration of waste. An electrostatic precipitator is used to clean flue gases, and no energy is recovered.
Source: CEC (2000), Table 9.1 and 9.3

1.3 Waste management failures and policy responses

The waste management system is prone to a number of so-called "failures" (Pearce & Turner, 1993; Goddard, 1995; Macauley & Walls, 2000). The failures can be grouped under four broad headings.

- Information failures: There has been a lack of comprehensive, consistent and current information about the scale and composition of the waste flows in the system, on which to base waste management decisions.

- Analytical failures: Waste management decisions have been taken piecemeal rather than within a holistic (or systems) framework, and there has been a failure to apply proper economic cost-benefit analysis. This has resulted in the adoption of inappropriate policies and targets.

- Government failures: The practice of funding domestic waste collections out of general taxation, rather than charging directly for collection and disposal services, means that there is no incentive for households to control the amount of waste that they produce. This has resulted in an excessive amount of waste being generated

- Market failures: Where market prices do apply, these reflect only the internal costs incurred by the agents, and fail to properly reflect the environmental damage costs that arise – i.e. there is a divergence between private and social costs. This has resulted in too low a level of recovery and source reduction.

Over the last decade, steps have been taken to address these failures in most countries. Information on the scale, composition, and treatment of waste arisings has improved, and a number of cost-benefit studies have been undertaken.[6] However, these studies are often based on incomplete

6. For example, the *ex ante* study undertaken for the European Commission on the costs and benefits of achieving reuse and recycling targets for different packaging materials (RDC-PIRA, 2001), and the *ex post* evaluation of the Swedish producer responsibility legislation by Radetski (2000).

data, and some suffer from methodological shortcomings. Consequently, there is still a long way to go before the information and analytical failures are eliminated.

In theory, the "first best" approach to dealing with the government and market failures would combine measures to internalise the external costs of waste disposal, with measures to make households bear the full disposal costs of the different types of waste that they generate. Unfortunately, this ideal is not attainable – at least with current technologies. However, it can be approximated by a combination of higher technical standards for the design and operation of landfill sites and incinerators, and the introduction of volume charges for households (Macauley & Walls, 2000; Godard 1995). The replacement of flat taxes with charges based on the volume of waste collected is consistent with the polluter pays principle,[7] and the available evidence suggests that it can have a significant impact on household behaviour. For example in nine studies across four European countries the introduction of volume charging resulted in a reduction in household waste that ranged from 15% to 50% (ACR, 1999).

The use of volume charging is growing. It is in widespread use in seven European countries – Austria, Belgium, Finland, Germany, Luxembourg, Sweden and Switzerland. It has also been adopted by over 4000 communities across the USA, accounting for around 12% of the population (ACR, 1999; Miranda *et al*, 1998). However, the adoption of volume charging is far from universal. For example, waste collection is still funded out of taxation in France, Greece, Portugal, Spain and the United Kingdom; as it is for 88% of the population in the USA. Furthermore, even where it has been implemented it is not practicable to differentiate the charge between different types of waste with different environmental impacts (e.g. between biodegradable waste, glass and plastics). Consequently, there will continue to be a need for other policy responses – either as a substitute for volume charging, or as a complement to it.

A wide range of alternative policy responses have been adopted, or proposed, to change the various flows within the waste management system. Essentially, these approaches fall into two broad categories – price-based responses and quantity based responses. Price-based responses include disposal charges; product charges; virgin material taxes; recycling credits and subsidies; deposit refund systems; differentiated packaging charges; tax credits for recycling equipment.[8] The more common approach however has been to adopt one (or more) of the following quantity-based responses.

- Recycled content standards (or procurement standards) set minimum limits for the amount of recovered material that a product must contain (or that total purchases must contain). They can be expressed relative to the amount of virgin material used in production (i.e. R/V in Figure 1), or relative to total output (i.e. R/Y). Assuming that the resultant rise in the cost of production is passed on to consumers then this should reduce the level of consumption (Y), and hence waste arisings (W). While the amount of virgin material used in production is necessarily reduced, the impact on the amount of recovered material used is ambiguous, depending on the relative magnitude of the substitution and output effects.

- Recycling targets set minimum limits for the proportion of household waste that is diverted for recovery (i.e. C_1/W). They may relate to total household waste or to specific components of the waste stream. If the targets are applied to local authorities then, in the absence of any mechanism

7. In accordance with the polluter pays principle, the EU Directive on Waste (91/156/EEC) requires that the cost of waste disposal be borne by the final holder (i.e. by the householder), or the producer of the product from which the waste arises.

8. Details of the various responses can be found in Turner et al (1994); Pearce & Turner (1993); Tietenberg (2000); Macauley & Walls (2000).

to pass the increased costs back to households, there will be no reduction in waste arisings (W). Furthermore, without corresponding measures to shift the demand curve for collected materials (C_1), it is possible that there will be problems of excess supply. Alternatively, responsibility may be imposed on producers under extended producer responsibility (EPR), with the target expressed relative to production (i.e. C_1/Y).[9] In this case, the fact that producers bear the net costs of collection and recovery means that there should be a reduction in production (Y), and hence waste arisings (W).

- Landfill restrictions set limits for the absolute amount of waste going to landfill (i.e. D_1). This approach is more usually applied to specific components of the waste stream, e.g. biodegradable waste. In the extreme, the disposal of the particular waste component to landfill may be banned. The restriction will necessarily lead to an increase in the amount of waste diverted for recovery (C_1). However, unless there is a mechanism for passing the resultant increase in cost on to the consumer there will be no impact on the level of waste arisings (W).

Despite the popularity of quantity-based responses with regulators, they have been subject to a number of criticisms. In particular, it has been claimed that the targets have been set arbitrarily without any analysis of the costs and benefits (i.e. there has been an analytical failure) (Goddard, 1995; Pearce & Turner, 1993), and that the costs have been excessive. While it is true that failure to undertake proper cost benefit analyses may lead to the adoption of inappropriate targets, it is also important to recognise that the cost of meeting a given target will depend crucially on the implementation mechanism that is used. For example, the cost of achieving a national recycling target may be lower if local targets are differentiated to take account of differences between rural and urban communities. Permit trading provides an ideal way of providing this flexibility for all three types of quantity-based response, and thus reducing the cost of achieving the respective objectives.

2. Designing tradeable permit schemes

Tradeable permit schemes can take many different forms, with detailed design specifications varying considerably between different policy applications. In particular, there are likely to be significant differences between the designs of trading schemes for waste management applications and those for traditional permit trading applications such as air pollution.

Essentially, tradeable permit schemes can be classified into one of two broad approaches (Godard, 2001).[10] Under a "cap and trade" approach, a fixed number of permits are created, and these are allocated or auctioned to companies, which are then free to trade them on the open market. Under a "baseline and credit" approach, companies are given performance targets (baselines), and they can generate credits to the extent that they beat their targets, which can then be traded on the open market. In the first case there is a fixed, vertical *ex ante* supply curve for permits, while in the second case, the supply curve of credits is only determined *ex post*, and may be upward sloping rather than vertical.

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9 . Extended producer responsibility requires producers to assume responsibility for the life-cycle environmental impacts of their products (see footnote 7). The most common manifestation of EPR is a requirement to arrange for the collection and recycling of a given proportion of "end-of life" products, or component materials.

10. Godard (2001) also discusses two other families of tradeable permit systems – transferable usage rights, and averaging. However, essentially these are just particular applications of permits (to natural resources), and credits (to product specification and performance).

In terms of their implementation, there can be significant differences between the two approaches. For example, greater use may be made of financial derivatives (such as forward contracts) under a "baseline and credit" approach, to compensate for the fact that credits can only be generated at the end of the compliance period when a firm's performance is known. However, from a theoretical perspective all types of trading system can be analysed within a single conceptual framework, using a number of generic building-blocks.

In any permit-trading system it is important to make a distinction between the market for permits (i.e. the new market that is being created) and the existing market in which the regulator wishes to intervene – the target market.[11] In traditional permit trading applications such as air pollution, the supply side of the target market is usually missing (i.e. it is an "incomplete" market). However, this need not necessarily be the case. For example, in all of the waste management applications identified at the end of the previous section, the object of the intervention is a "complete" market, where both the demand and supply sides exist.

A permit scheme works by influencing behaviour in the target market. This is achieved via two rules that link the markets together. The generation rule determines how many permits are created (i.e. the supply), while the performance rule determines how many must be redeemed to the authorities (i.e. the demand). In combination, these two rules cause either the demand curve, or the supply curve, in the target market to shift; with the direction of the shift depending on definition of the rules. Hence is possible to use a permit trading scheme either to reduce the level of activity in the target market – as is the case with emissions limits, or to increase it – as would be the case with recycling targets.

When designing a permit trading scheme, there are many aspects that need to be considered (Godard, 2001). Decisions must be taken regarding (*inter alia*) the scope and scale of the scheme; the legal status of permits; the degree of temporal and spatial flexibility that will be allowed; the monitoring systems that will be used; and the sanctions that will be applied in the case of non-compliance. However, it is the choices relating to the definition of the generation and performance rules, and their assignment (i.e. to whom each rule is applied), that are of fundamental importance. Together, these decisions determine which type of scheme is used (i.e. "cap and trade" or "baseline and credit"); which parties have an obligation to acquire permits; and which parties have the initial property rights over permits.

The choices open to regulator for these decisions (i.e. the choice-set) will depend on a variety of factors that are likely to vary between different policy applications. In particular, the number of alternatives will be greater:

- if the policy objective is expressed in absolute terms, rather than relative terms;
- if the objective is to reduce the level of activity in the target market, rather than to increase it;
- if the target market is "complete", rather than "incomplete"

Thus, for example, a regulator will have greater flexibility in designing a trading scheme to implement an absolute limit for the amount of waste that is sent to landfill, than to implement a target recycling rate.

When designing or analysing a permit-trading system, it is important to identify all of the markets and sectors that are affected by the introduction of the scheme (i.e. to define the appropriate

11. For simplicity, the term 'permit trading system', 'permit market, and 'tradeable permits' are used in relation to both approaches.

boundary for the trading system). As a minimum, the system will include the target market and the permit market, and the sectors involved in these two markets. However, a downstream product market, or an upstream factor market, could also be included, together with the participating sectors. In some applications, it may also be appropriate to include a related market if it is a close substitute or complement to one of the other markets in the system. Inevitably, the decision over which markets and sectors should be included will be a matter of judgement, and the boundary will vary from application to application. Together, the assignment of the performance and generation rules, and the definition of the system boundary determine the trading system structure.

If all of the markets within the trading system are perfectly competitive, then the choices regarding the assignment of the generation and performance rules (i.e. obligations and initial property rights) will affect only the distribution of costs between the sectors. However, when firms can take advantage of their market power to influence prices, these choices can affect the total cost of a permit scheme (i.e. its economic efficiency). Furthermore, if the policy objective is expressed in relative terms, they can also affect the environmental outcome. The implications of market power for the design of permit trading schemes is a complex issue, and is beyond the scope of this paper. However, one study that has considered the issue in relation to a particular waste management application – extended producer responsibility – is discussed in Annex 2.

The various design concepts identified above are illustrated in Box 1 for a particular (hypothetical) policy application – an absolute limit for the aggregate emissions of CO_2 by electricity generators. This traditional application of permit trading provides a useful comparison for the waste management applications that are considered in the next section (see Box 2 and Box 3).

3. Waste management applications of permit trading

The potential for using permit-trading schemes in the field of waste management has been recognised for some time. For example, a report for the UK Department of Trade and Industry in 1992 on the potential for using economic instruments to recover resources from waste (ERL, 1992) included tradeable permits among the thirteen different instruments that were considered. It concluded that transferable recycling targets for municipal waste, and tradeable targets for recycled content of products, both had the potential to encourage recycling and improve cost efficiency. This belief is shared by Turner *et al* (1994), although they caution that the realisation of the potential efficiency gains offered by permit trading schemes will depend on the size and competitive structure of the markets in which they are used.

A small number of potential waste management applications have been discussed in the literature. Dinan (1992) analyses a "baseline and credit" scheme that was being considered by the US Congress at the beginning of the 1990s to implement an aggregate performance standard for the recycled content of newsprint.[12] Under the proposal – which was never implemented – all newsprint suppliers would have been subjected to a common recycled content standard (i.e. recycled newsprint / total newsprint $\geq \rho$). They would be allowed to generate $(1-\rho)$ credits for each ton of recycled newsprint that they produced, and would be required to redeem ρ credits for each tone of virgin newsprint that they produced. Hence, if a particular supplier just satisfied the performance standard, it would be able to "purchase" sufficient credits from itself to satisfy the performance rule (i.e. $N \rho (1-\rho) = N (1-\rho) \rho$, where N is the total quantity of newsprint produced). If however its average rate fell short of the target, it would have to purchase additional credits from other firms. Similarly, if a firm beat the performance target, it could sell the excess credits that it generated.

12. H.R. 873 and S. 397.

Box 1. Conceptual framework for a CO2 emissions trading system

Rules

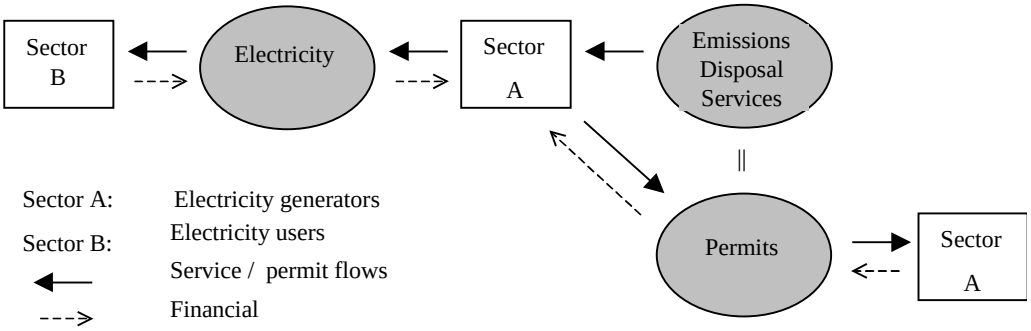
The following generation rule and the performance rule define a "cap and trade" system; where the total number of the permits is set equal to the aggregate emissions limit. The assignment of the generation rule to the electricity generators implies that they are granted the initial property rights for the permits (e.g. under grandfathering).

Rule	Definition			Assignment
Generation	Permits sold	≤	Initial allocation	All generators
Performance	Permits purchased	≥	Emissions	All generators

Under this definition for the generation rule, a generator is prevented from selling more permits than its initial allocation (i.e. engaging in speculative buying and selling). The rule also implicitly prevents a third party from selling permits – although not from purchasing permits and retiring them. The performance rule implies that each generator will sell all of its initial allocation. However, some or all of the sales will be to itself (i.e. "internal sales") at zero net cost.

Trading system structure

For the purposes of this example, it is assumed that there are no close substitutes for electricity, and that there is a wide variety of abatement options available to the generators. Therefore there are no related markets, and the trading system structure is relatively simple; comprising just three markets and two sectors. The target market is the "incomplete" market for emissions disposal services, in which the supply side is missing.[13] The downstream product market is the market for electricity.



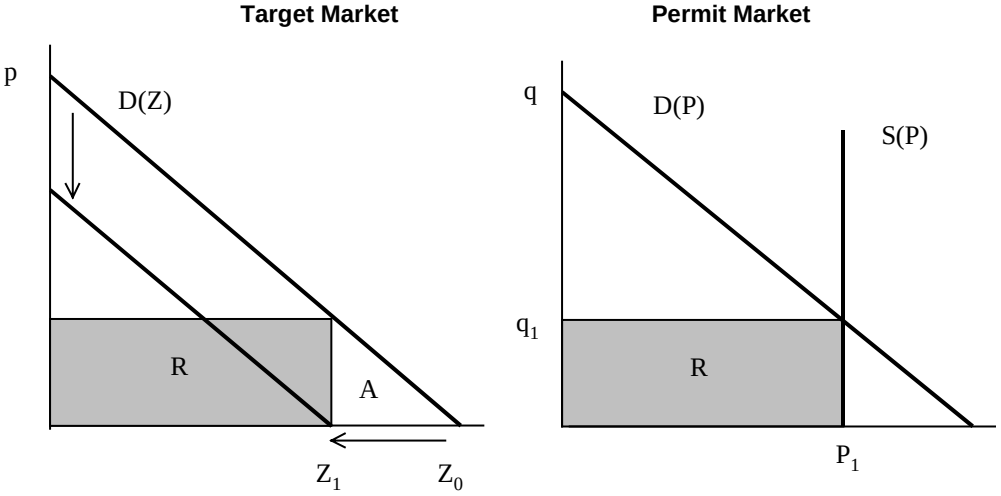
13. It is instructive to interpret the output of emissions to the atmosphere as an input of emissions disposal services provided by the atmosphere.

Box 1 (continued)

Short run impacts

For the purposes of this example, it is assumed that the demand curve for electricity is flat (i.e. that the price is constant). This implies that the introduction of the trading scheme will impose no cost on electricity users, and that all of the impacts can be measured in the target and permit markets.[14]

The supply curve in the permit market is vertical, reflecting the fact that there is an absolute limit for CO_2 emissions. The demand curve is identical to the demand curve in the market for emissions disposal services, which in turn is equal to the marginal cost of emissions abatement (i.e. the supply curve for abatement).



The introduction of the trading scheme causes the demand curve for emissions disposal services to shift downwards, by an amount equal to the permit price (q_1).[15] Consequently, the amount of emissions disposal services used by the electricity generators decreases from Z_0 to Z_1 (= P_1). The resultant reduction in the operating profits of the generators is given by the area of the triangle A.

If the initial property rights had been assigned to the government (and then auctioned), then the reduction in generators' operating profits would be equal to R+A, with the permit revenue (R) accruing to the government.

14. This unrealistic assumption has been made in order to simplify the example. In reality, the scheme will impose costs on electricity users in addition to those borne by the generators.

15. It should be noted that the introduction of the permit-trading scheme does not create a supply curve for emissions disposal services. This is in contrast to the introduction of an emissions tax which does have the effect of creating a supply curve (i.e. the government becomes the supplier of the service).

Dinan shows that, provided that the credit and newsprint markets are both competitive, then the outcome will be cost efficient; with the scheme being equivalent to a combined tax / subsidy.[16] He goes on to consider whether the pre-condition of efficient markets was likely to be satisfied in the case of the US newsprint market, focusing on four issues: compliance; transaction costs; market structure; uncertainty over property rights. While there were some issues related to compliance and liability rules, the main concern arose from the likelihood of market power problems on the supply side of the permit market – forcing up permit prices; and the potential for firms to user their dominance in the permit market to gain market share in the newsprint market.

An example of an application to recycling is provided by Sprenger (1999), who evaluates the potential for using a system of tradeable permits to implement the target refilling rates for beverage containers that are included in the German Packaging Ordinance.[17] The system that he considers is based on a series of absolute targets for the amount of beer that could be sold in disposable packaging on the German market in a particular year. Permits, which would be denominated in 1,000 litres per year, would give the holder the right to sell that volume of beer in disposable packaging, and they would be freely tradeable and completely transferable. A number of design issues are discussed, including the point in the distribution chain at which the system should be imposed; the alternative allocation mechanisms that could be employed; the options for market organisation and control; and the need for effective monitoring and control.

The benefits of using a permit trading system are assessed in terms of environmental effectiveness; economic efficiency; distributive impacts; administrative feasibility; and political acceptance. Sprenger concludes that it offers considerable advantages over alternative policy instruments – ensuring that the environmental goal is met, and promoting static and dynamic efficiency. By combining free allocation (for existing companies) with a permit reserve (for potential entrants), it should be possible to overcome any political resistance to the introduction of the scheme, while avoiding the creation of barriers to new market entrants.

More recently, a report for the European Commission (ERM, 1999) has considered the possibility of using an EU wide permit trading system to finance the recovery of waste electrical and electronic equipment (WEEE). The proposed scheme is similar to that operating in the United Kingdom for packaging waste (see below), although there are a number of important differences. In particular, the recovery target would be defined in absolute terms (i.e. tonnes of equipment recovered), with individual targets being calculated for obligated producers on the basis of their respective market shares. Like the UK scheme however, obligated producers would not be required to meet their recovery targets themselves (as they would in a take back scheme). Rather they would be required to purchase certificates from reprocessors as proof that the necessary tonnage of products had been recovered.

An interesting aspect of the proposed scheme is a provision to "weight" the individual obligations to favour producers of products with lower environmental impacts, or that are easier to dismantle and recover. It is hoped that this will create incentives for improved eco-design, although the report recognises that it may be difficult to establish objective criteria for determining the weighting factors.

The relative scarcity of potential applications in the literature, is mirrored by the small number of actual examples of permit trading applications in the area of solid waste management.

16. Production of recycled newsprint is "subsidised" by the sale of credits, while the production of virgin material is "taxed" by the purchase of credits.
17. Ordinance on the Avoidance of Packaging Waste (Verpackungsverordnung), 1991

One might expect to find some waste management applications in the USA – which has led the world in developing permit trading as a policy tool. However, in a recent authoritative review of permit trading experience in that country (USEPA, 2001), no mention is made of any applications in this policy area.

One country where permit trading is being applied to waste management issues is the United Kingdom. Although not conceived as such when they were introduced in 1998, packaging waste recovery notes have rapidly evolved into a functioning tradeable compliance credit system. In a separate initiative, a "cap and trade" scheme for the disposal of biodegradable municipal waste to landfill is currently in its final design phase. These two schemes are now considered in detail.

3.1 Packaging Waste Recovery Notes

a) Policy background and status

The 1994 EU Directive on Packaging and Packaging Waste[18] set targets for recovery and recycling of packaging waste. Member states are required to recycle between 25% and 45% of total packaging by 2001, with total recovery rates of between 50% and 65%. There is also a minimum recycling rate for each material of 15%. The Directive was transposed into UK legislation in March 1997 by statutory Packaging Regulations[19], which set a series annual targets for the amount of packaging that was to be recovered and recycled from 1998 onwards. The targets for the later years have subsequently been revised on separate two occasions – most recently in November 2000 (see Table 3).

Table 3. Packaging recycling and recovery targets

		UK producer responsibility targets				EU Dir. targets
		1998	**1999**	**2000**	**2001**	**2001**
Total recovery	original	38%	38%	43%	52%	50 – 65%
	revised		43%	45%	52%	
	revised				56%	
Material recycling	original	7%	7%	11%	16%	15%
	revised		10%	13%	16%	
	revised				18%	
Total recycling		n/a	n/a	n/a	n/a	25 – 45%

The implementation of the Packaging Regulations represented the culmination of a four-year process that had started with the UK Government's challenge to the packaging chain to set up a voluntary scheme to recover packaging waste. Following the sector's rejection of the voluntary approach in 1994, the Government embarked on a lengthy consultation process with representatives of the packaging chain (and other interested parties), before publishing draft Regulations in July 1996 (DOE, 1996).[20] During this process the discussions focussed on two main issues – the level of the recovery and recycling targets, and who would be responsible for meeting the targets (i.e. whether the obligation would be imposed on a particular sector within the chain, or shared among all sectors). Much less attention was paid to *how* the targets would be achieved. In particular, no consideration

18. Directive 94/62/EC

19. Producer Responsibility Obligations (Packaging Waste) Regulations 1997

20. For a discussion of the voluntary initiative, and the reasons for its failure, see Eden (1996).

was given to the potential for using a system of tradeable credits to implement the targets, and the Packaging Regulations made no provision for the introduction of such a scheme. While there was a general acceptance of the advantages of a market-led approach, it was envisaged that this would be achieved largely through a system of collective compliance schemes, which would use the fees that they received from their members to fund the necessary expansion of collection and reprocessing activities.[21]

The emergence of a tradeable credit system can be traced back to two separate initiatives. In July 1996 (i.e. at the same time as the draft Regulations were being published), the largest compliance scheme (Valpak) announced that – rather than arranging directly for the collection of waste packaging – it would contract out this responsibility to the material reprocessors. Contracts would be signed with reprocessors to recycle a specified amount of material. Once this had been done, each reprocessor would send Valpak a "certificate of confirmation". Up until that point, it had been implicitly assumed that compliance schemes would be responsible both for the financing of waste collection activities, and for arranging for those activities to be undertaken. This decision had the affect of de-coupling these two responsibilities.

The Packaging Regulations specified that proper evidence would be required to prove that waste packaging had actually been recovered, but they left the precise definition of what would be considered acceptable to subsequent statutory Guidance from the Environment Agency. When this was published in July 1997, it introduced a standardised form of evidence – called Packaging Waste Recovery Notes (PRNs) – that would be issued by accredited reprocessors and accepted as proof that material had been recycled. The main motivations for the introduction of the scheme were the deterrence of fraud by reprocessors, and the reduction of the administrative burden on the obligated businesses. However, by creating a transparent "common currency", the introduction of PRNs facilitated the evolution of the reprocessor contracts introduced by Valpak into a tradeable compliance credit system.

b) Description of scheme

In addition to setting the aggregate targets shown in Table 3, the Packaging Regulations impose a number of obligations on individual producers and users of packaging materials that satisfy certain threshold conditions.[22] These "obligated producers" are (*inter alia*) required to:

- register with the relevant Environment Agency[23] and provide data on the amount of packaging materials produced or used;
- take reasonable steps to meet their individual recycling and recovery obligations;

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21. The Regulations also made provision for obligated companies to meet their own recovery and recycling targets; either through their own activities, or by contracting with third parties to collect waste packaging on their behalf and arrange delivery to reprocessors.

22. When the Packaging Regulations were introduced they applied only to businesses that handled more than 50 tonnes of packaging, and that had a turnover of more than £5 million. The second threshold was reduced to £2 million from the start of 2000. It is estimated that companies below these threshold levels account for around 9% of total packaging. Thus a 56% recovery rate by obligated companies should ensure that the overall recovery rate exceeds 50%, as required by the Directive.

23. Companies in Scotland must register with the Scottish Environment Protection Agency (SEPA). Any references in the text to the Environment Agency implicitly include SEPA.

- provide the Environment Agency with annual of certificates of compliance, supported by acceptable evidence that recycling and recovery has actually taken place;

The individual recycling and recovery targets are determined according to the principle of shared responsibility, under which the total obligation is broken down between the different stages of the packaging chain (see Box 2).

By the beginning of 2001 over 5800 obligated producers had registered with the Environment Agency, and a total of 229 reprocessing sites had been accredited; these being owned by 186 independent companies. As an alternative to registering directly with the Environment Agency, an obligated producer can choose to register with a collective compliance scheme, which – in return for the payment of an annual fee – will assume responsibility for discharging its obligations. The recycling and recovery obligations for each compliance scheme are then calculated as the sum of its members' individual obligations. As can be seen in Table 4, the large majority of producers have chosen to follow this route, with compliance schemes accounting for around 80% of total registrations.

Table 4. Breakdown of registered producers and accredited reprocessors for 2001

Registered producers	No. of reg.s	Est. Obl 2001 (kT)	% of total	Accredited reprocessors	No. of sites	No. of co.s
Valpak	2984	2,500	52.6%	Aluminium	10	10
Wastepack	466	300	6.3%	Glass	25	19
Biffpack	351	440	9.3%	Paper	43	34
Paperpak	223	110	2.3%	Plastics	80	68
Onyxpak	220	250	5.3%	Steel	13	7
Difpak	153	100	2.1%	Wood	22	21
Cleanpack	102	120	2.5%	Material recycling	193	159
10 other schemes	219	130	2.7%	Other recovery	36	27
Compliance schemes	4718	3950	83.1%	**Total recovery**	**229**	**186**
England & Wales	727					
Scotland & NI	365					
Direct registration	1092	800	16.9%			
Total	**5810**	**4750**	**100.0%**			

Source: Environment Agency; Materials Recycling Handbook (2001).

While PRNs are not the only form of evidence that will be accepted by the Environment Agency, the administrative burden associated with the use of other forms has resulted in the almost universal use of them as evidence of compliance. The accreditation scheme is voluntary, with reprocessors having to provide information to the Environment Agency about their activities, and to abide by certain rules regarding the administration of PRNs. Accreditation is on a site basis and must be renewed annually, with the Environment Agency having the right to refuse renewal. Reprocessors are required to submit quarterly reports on amount of waste packaging reprocessed, and are allowed to issue PRNs up to this amount.

The Environment Agency Guidance also made provision for PRNs to be issued by overseas reprocessors that were covered by recognised accreditation schemes operated by local or national governments. However, this was replaced in 1999 by a separate accreditation scheme for exporters of waste packaging materials for reprocessing. Companies satisfying certain requirements are given Agency Accredited Exporter Status (AAES) and are allowed to issue Packaging Waste Export

Recovery Notes (PERNs).[24] In first nine months of 2000, exports accounted for almost 10% of total recycling; being particularly significant for steel (46%) and plastics (19%).

The large majority of PRN sales are covered by contracts between the reprocessors and compliance schemes. However, the number of spot trades, and secondary transactions between obligated parties is increasing. This has been encouraged by the introduction of two electronic services. Evidex provides an electronic registry service, facilitating the transfer of title between the trading parties. The web-based Environment Exchange goes further; anonymously matching potential buyers and sellers, and executing transactions.[25] At the start of 2001, forward contracts for PRNs were also introduced. There are currently around 300 members of the Exchange (including 85 accredited reprocessors) and the volume of trades has doubled every year, reaching 100,000 tonnes in 2000. As one would expect, the Exchange is much more relevant to individually registered producers than compliance schemes. Almost 80% of total PRN purchases were by individual producers, accounting for 11% of their total obligation. In contrast, purchases by compliance schemes represented only around 2% of their total obligation.

Failure to comply with the obligations is a criminal offence, with a maximum penalty of £20,000 in a magistrates court (or potentially an unlimited fine in a higher court). However, because obligated producers are only required to "take reasonable steps to recover and recycle packaging waste", in effect companies can only be prosecuted for failing to register or submit a certificate of compliance, or for providing false information. Since the beginning of 2000, there have been 44 prosecutions, with penalties (including legal costs) averaging around only £3250. This was only slightly higher than the average cost of compliance which was around £3000 in 2000 (see below), although this takes no account of other non-financial costs such as the management time involved, and the damage to reputation.

Monitoring is based on a system of self-reporting, backed up by periodic audits undertaken by the Environment Agency. Accredited reprocessors are required to submit quarterly returns showing details of the amount of waste processed and the PRNs that they have issued. Obligated producers (and compliance schemes) must submit an annual certificate of compliance, signed by an approved director, and accompanied by summary details of the PRNs that it has acquired to meet its recycling and recovery obligations. The Environment Agency audited 367 directly registered companies and 805 members of compliance schemes during 2000. It also audited a further 867 companies that were not registered, uncovering around 200 free-riders (i.e. firms that should have registered, but had not done so).

24. For simplicity the acronym PRN is used for both packaging waste recovery notes and packaging waste export recovery notes.

25. Details can be found at the Environment Exchange website http://www.t2e.co.uk

Box 2. Conceptual framework for the PRN system

Rules

The PRN system is a "baseline and credit" system, with a relative target. The generation rule is applied to material reprocessors, while the performance rule is applied to obligated producers.

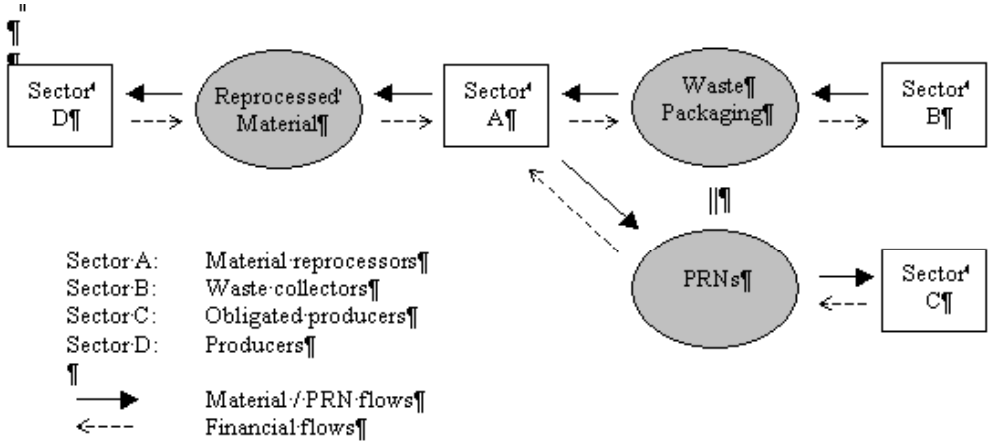
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Rule	Definition	Assignment
Generation	PRNs sold ≤ waste packaging accepted	Material reprocessors
Performance	PRNs purchased ≥ ρ_i × packaging material used	Obligated producers in sector *i*

The recovery target (ρ_i) for a company in sector i is equal to the total target recovery rate multiplied by a predefined sector share (i.e. $\rho_i = \rho \times \sigma_i$). The shares are currently 6% for raw materials manufacturers; 9% for converters; 37% for packer / fillers; and 48% for sellers. So for example, a supermarket that handles 100,000 tonnes of packaging would have a total recovery obligation in the year 2000 of 21,600 tonnes (i.e. 100,000 × 45% × 48%).

Trading system structure

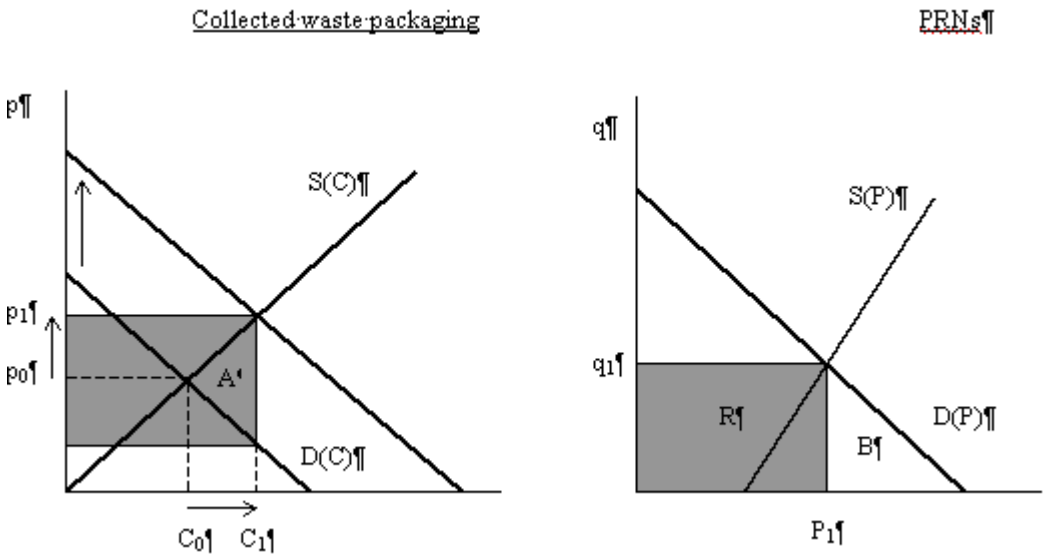
The trading system comprises four sectors and three markets – all of which are "complete". The target market is the market for collected waste packaging material, which comprises material reprocessors (on the demand side) and waste collectors (on the supply side).



Short run impacts

The demand curve in the PRN market is downward sloping, reflecting the fact that obligated producers can reduce the number of PRNs that they have to purchase by reducing the amount of packaging that they use. The supply curve for PRNs reflects the difference between the supply and demand curves for collected waste packaging (i.e. S(C) - D(C)). The total cost to the obligated producers is equal to the cost of acquiring the PRNs from the reprocessors (the shaded rectangle R), plus the decline in operating profits resulting from the reduction in the amount of packaging used – given by the triangle B. Thus, the value of PRN revenues understates the cost of the scheme to the obligated producers.

Box 2 (continued)



The introduction of the trading scheme causes the demand curve for waste packaging to shift upwards, by an amount equal to the PRN price (q_1). The market price of waste packaging rises – but by less than the PRN price, and the amount purchased increases from C_0 to C_1. Operating profits for the waste collectors increase while those of the reprocessors decline. Overall, the market suffers a net operating loss – given by the area of triangle A. However, the revenue that the reprocessors receive from the sale of PRNs to the obligated producers (the shaded rectangle) more than offsets the reduction in their operating profits.

If the generation rule had been applied to waste collectors then the scheme would have caused the supply curve to shift downwards, causing the market price for waste packaging to fall. However, the size of the total benefit (R – A), and its distribution between reprocessors and waste collectors would have been exactly the same.

The aggregate cost of the PRN scheme across all three sectors (i.e. reprocessors, waste collectors and obligated producers) is equal to cost in the PRN market less the benefit in the market for waste packaging: i.e. (R + B) – (R – A) = A + B. This may be greater or less than the total value of PRN revenues (R), depending on the relative slopes of the supply and demand curves in the two markets.

PRNs are not a statutory instrument; being defined in Guidance from the Environment Agency rather than in the Packaging Regulations. This has resulted in a degree of uncertainty and confusion over property rights and trading rules. The introduction to the Guidance states that *"[it] will be updated from time to time …"*, and that PRNs *"… will normally be considered …"* as providing evidence of compliance (EA, 1997). This ambiguity regarding the continued acceptance of PRNs as evidence of compliance, might be expected to undermine their future value. However, because they only have a twelve month life (banking is not allowed), this is unlikely to have been a significant problem in practice. A bigger problem was caused by the failure to assign initial property rights.[26] In competitive markets, the outcome should be the same whichever side of the collected waste market is given the initial rights. Prices of collected waste will be higher if the rights are given to the

26. It was clear that reprocessors had the administrative responsibility for issuing PRNs. The confusion arose over who they should issue them to – the waste collectors or themselves.

reprocessors, and lower if they are given to the waste collectors – the difference being exactly equal to the PRN price (see Box 2). However, in the absence of any clear rules reprocessors adopted different policies; causing confusion in the markets for both PRNs and collected waste packaging. Concerns have also been expressed that the lack of any clearly defined trading rules could result in speculation by non-obligated companies (or individuals) driving up PRN prices, although there is no evidence that this has occurred.

In response to these and other concerns (see below), a number of amendments were made to the PRN system in 1999. Accredited reprocessors are now only allowed to issue PRNs to obligated producers and compliance schemes (or their agents), and they must give obligated producers that deliver waste packaging first refusal on the resultant PRN. This was intended to prevent speculation by non-obligated parties, and to clarify the initial property rights. Reprocessors must also provide an annual return to the Environment Agency showing the total revenue generated from sales of PRNs in the previous calendar year, and the proportions of this revenue used to fund the expansion of the collection infrastructure; additional investment in reprocessing capacity; and the development of end-use markets.

c) Assessment of the scheme

Box 2 provides a simplified conceptual framework for the PRN system. In theory the system has a number of attractions: it should reduce total compliance costs; it is consistent with the polluter pays principle; and it integrates environmental costs into business decision making (Bailey, 2000). However, a number of concerns have been expressed about the operation of the scheme in practice (Bailey, 1999; 2000; O'Doherty & Bailey, 2001; DETR, 1998; ENDS, 2001). In particular, it has been claimed that:

- the system will not result in sufficient waste packaging being collected to meet the targets set out in the Directive;
- the price signals provided by the PRNs have been too small to change the behaviour of obligated producers, in terms of reducing the amount of packaging that they use;
- the revenues from PRN sales are not being used to develop the reprocessing and collection infrastructures as was intended;
- the small number of firms in some reprocessing sectors, together with barriers to new entrants has allowed reprocessors to exercise market power – raising PRN prices and earning excess "windfall" profits;
- the dominant position of Valpak on the demand side of the market has driven up the prices of PRNs for smaller obligated producers and compliance schemes.

Essentially the first three concerns relate to the environmental effectiveness of the scheme, while the last two relate to its economic efficiency. Is there any evidence to suggest that any of these concerns are justified?

Environmental effectiveness

It is difficult to assess whether the concerns over the achievement of the final target figure are justified. So far, the recovery targets have been achieved in every year – both in aggregate and for individual materials (see Table 5). Between 1998 and 2000 total recovery increased by around 500,000 tonnes (i.e. around 15%). However, the rate of improvement will have to accelerate substantially in

2001 if the final target is to be met; with total recovery needing to increase by a further 900,000 tonnes. In particular, significant increases will be required for aluminium and plastics if they are to meet their respective minimum recycling requirements.

Table 5. Packaging waste reprocessed (000's tonnes)

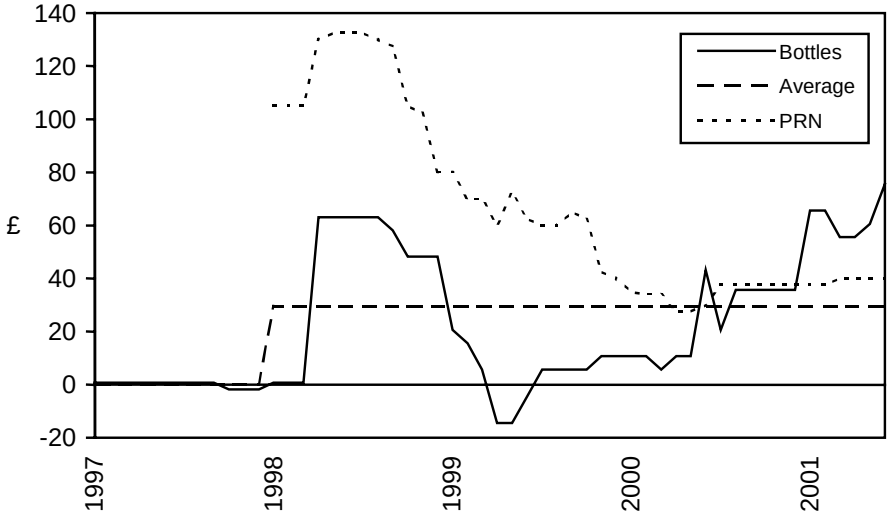
	Actual				Minimum recycling requirement	
	1998*	1999*	2000	Rate	2001	Rate
Aluminium	14.5	15.4	16.3	14.3%	20.6	18.0%
Steel	182.4	225.2	239.0	35.6%	120.8	18.0%
Plastic	125.5	198.5	204.4	12.9%	285.1	18.0%
Glass	503.8	582.6	715.0	34.3%	375.3	18.0%
Paper	1894.1	1820.7	1879.7	56.4%	612.3	18.0%
Wood	170.0	94.0	296.6	99.1%	107.0	18.0%
Total recycling	2890.4	3010.3	3351.1	41.4%	n/a	
EfW	448.4	496.3	500.8			
Total recovery	**3338.7**	**3506.6**	**3851.8**	**47.6%**	**4752.0**	**56.0%**
(increase)		**167.9**	**345.2**			

Source: DEFRA (2001).

* No data for Northern Ireland in 1998. Estimated data included in total recycling and EfW for 1999, but not in individual material figures.

Including energy recovery and composting, the total reprocessing capacity for packaging materials in the UK is currently around 4.1 million tonnes per annum (DETR, 1999a). Together with the capacity of overseas reprocessors, this should be sufficient (just) to handle the necessary increase in recovery. Thus, the achievement of the target will depend crucially on the ability of the system to increase the collection of post-consumer packaging – particularly for aluminium and steel. Hence the concerns about the lack of investment in the collection infrastructure. These have been fuelled by the relatively small number of reprocessors that have entered into direct partnerships with local authorities to expand household collection schemes (Bailey, 2000). However, this ignores the indirect support that may have been provided through increased prices for collected waste packaging.

Figure 2: Increase in the price paid by reprocessors for waste plastic bottles



Note: Price of bottles is the average of the maximum quoted prices for natural HDPE and colourless PET bottles. PRN price is the mid-point between the high and low quoted prices from reprocessors.

Source: Materials Recycling Week.

If the system is working properly then the introduction of the PRN scheme should have caused the price of collected waste to rise, but by less than the full PRN price (see Box 2). While it is always difficult to establish an appropriate counterfactual against which to make a comparison, there is some evidence to suggest that this has indeed occurred – at least for some materials. Immediately following the introduction of the scheme in 1998 there was a step change of around £50-60 in the price paid by reprocessors for waste plastic bottles, which represented 50% of the average PRN price (see Figure 2). The price fell back sharply in 1999, reflecting the decline in the price of plastics PRNs, before settling down approximately £10 higher than the pre-scheme level; stepping up again in the middle of 2000. Over the three-year period as a whole, the average price paid was £30 higher than in 1997, equivalent to 65% of the average PRN price. Following a further steep price rise at the beginning of 2001, the implied price impact (at around £60+) is now greater than the PRN price, which suggests that it may no longer be valid to use the 1997 price as the baseline for the comparison. However, it could also be explained by the expectation of rising PRN prices in the second half of the year. This rudimentary analysis is given some credence by the comments of one of the leading recyclers of waste plastic packaging, who claimed in 1998 that sales of PRNs for waste plastics had resulted in increased prices being paid for post-consumer plastic containers.[27]

A similar impact can be detected in the case of waste cardboard, where the average price differential over mixed paper waste has increased by around £5 following the introduction of the PRN scheme, although the change is not statistically significant. This represents 37% of the average paper PRN price. In contrast, it has not been possible to find any evidence of a rise in the price paid for used aluminium beverage containers. Thus, while these results are far from conclusive, they do suggest that a proportion of the PRN revenue is filtering back to the waste collectors in the form of higher material prices.

Unfortunately, the data on the amount of packaging material used is not sufficiently robust to assess whether there has been any impact on packaging design and usage. However, it is true to say that – to date – the costs of compliance have been relatively low. Based on average PRN prices in the final quarter of 2000 (see Figure 3), the average variable cost of compliance in that year was around £700 for smaller firms (i.e. turnover of £2-5 million) and £4000 for larger firms (i.e. turnover > £5 million). These amounts represent only around 0.02% - 0.04% of turnover. At this level of cost, it seems unlikely that many packaging reduction initiatives will have been undertaken. Of course, this is a result of the level at which the targets were set rather than the use of a trading system, although to the extent that the scheme has promoted efficiency it will have reduced the incentive for action by producers.

While the scale of the compliance costs may not have provided an adequate incentive for packaging reductions, it may explain the apparent high levels of compliance by obligated companies, despite the low level of the penalties that have been imposed. Prior to undertaking the audits in 2000, the Environment Agency estimated that there were around 250-500 free-riders. If this was correct, then it represented only 2-3% of obligated businesses. However, with compliance costs due to double in 2001, the incentive for businesses to free-ride will increase unless there is a corresponding increase in the level of penalties.

27. Article by Andrew Simmons, director of Recoup Ltd in *Wastes Management*, May 1998 pp 24-25.

Economic efficiency

At the time that the Regulations were introduced the Government expected costs to be around £200 million per annum, while Valpak believed that they could be as high as £600 million.[28] The actual costs have proved to be much lower. In 2000, PRN revenues were around £45 million (excluding administrative costs), and while they will increase in 2001 they are unlikely to exceed £100 million. Of course, this does not necessarily imply that the outcome is cost efficient, although it does suggest that the scale of any efficiency losses is likely to be low. Unfortunately, without detailed information about the costs and production technologies of all reprocessors it is impossible to determine directly whether the outcome is cost efficient.[29] However, an analysis of the competitive structure of the market and the trends in PRN prices can provide a good indication of whether this is likely to be the case.

With the exception of plastics, all materials exceeded the minimum recycling rate in 2000 (see Table 5). Consequently, PRNs for these materials are interchangeable, both with each other and with those issued by energy recovery companies. Thus the supply side of the market can be divided into two segments – PRNs for plastic packaging, and all other types of PRN. On the demand side, the fact that the Environment Exchange is the only significant arena in which secondary transactions can take place means that the purchasers of PRNs can also be segmented into two groups – the 20% that belong to the Environment Exchange and the 80% that do not. Thus in effect there are four separate PRN markets, reflecting the segmentation on each side of the market. The demand side segmentation is particularly useful for analysing the competitiveness of the PRN markets, as any market power problems are likely to be reflected in price discrimination between the two segments.

At the end of 2000 there were 68 independent companies reprocessing plastic packaging, and 118 reprocessors of other materials and energy recovery companies. All of these companies sell PRNs directly, while slightly under half also sell through the Environment Exchange. The four largest plastics reprocessors account for approximately 33% of plastics PRN sales, with the largest company having a 15% market share. For all other materials and energy recovery the figures are lower – with a 4-firm concentration ratio of 25% and the largest company accounting for 10% of the market. These figures suggest that market power is likely to be more of a problem in the market for plastics PRNs. However, in neither case is market concentration particularly high.

On the demand side there around 1100 potential purchasers of PRNs, of which over 200 are members of the Environment Exchange. As can be seen in Table 4, Valpak is by far the largest player, accounting for over 50% of total demand. In theory this should allow it to act as a monopsonist (with a competitive fringe) – forcing down the price of PRNs. However, in order to exercise this potential market power, Valpak must be able to manipulate its demand for PRNs. Since this is determined by the sum of its members individual obligations, it would require a high degree of control over their actions, which does not seem very plausible. Thus in practice, its seems likely that the purchasers of PRNs will all act as price takers.

Figure 3 shows the trends in the PRN prices quoted by reprocessors for direct sales since the start of the scheme, and the annual average prices of transactions through the Environment Exchange from 1999 onwards. Prices during the first year were high – particularly in the case of plastics where the PRN price peaked at around £130 in the middle of the year. However, this appears to have been due to a lack of understanding and trust of the system, and to "panic buying" by compliance schemes

28. ENDS Report No. 264, January 1997

29. In the cost efficient outcome, the PRN price should be equal to the difference between the marginal cost to the reprocessor of collected waste and the value of the marginal product of reprocessing.

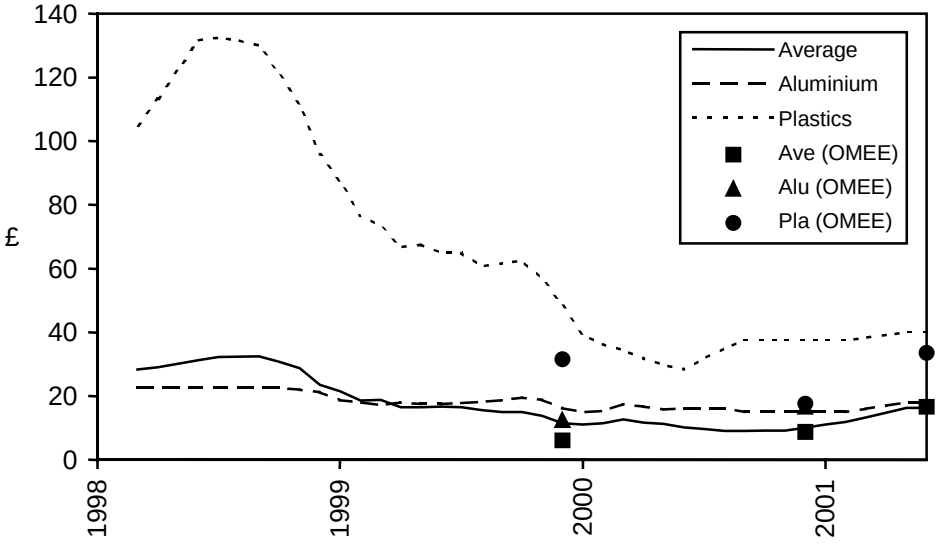
that were erroneously forecasting a shortfall in the supply of PRNs by the year-end, rather than any strategic behaviour on the part of the reprocessors. By the end of the year the market had started to settle down, with PRN prices falling sharply for all materials. In recognition of these initial teething problems, the analysis of the price trends starts at the beginning of 1999.

There are a number of things that one would expect to see if the market is working properly. Firstly, PRN prices should fall as the number of accredited reprocessors increases, and rise as the recycling and recovery targets become more stringent. This has indeed been the case. Between the beginning of 1999 and the end of 2000, PRN prices declined as the increase in the number of accredited reprocessors outweighed the relatively modest increase in the target rates. This downward trend is particularly noticeable in the case of plastics, where the expansion of accreditation to overseas reprocessors (following the introduction of PERNs) led to significant reduction in the price in 2000. As expected, prices have turned up in the first half of 2001 reflecting the step increase in the target recovery rate.

In the absence of any market power problems, the prices of direct sales and Exchange sales should be the same for all types of PRN. Furthermore, one would expect the price of plastics PRNs to be higher than those of other materials, which in turn should all be equal to the price of energy recovery PRNs. As can be seen from Figure 3, there does indeed appear to be a segmentation of the market between plastics and all other types of PRN. Prices for the second group have moved together since the start of 1999 within a narrow band around an average of £10-£15. In contrast, the price of plastics PRNs has been much higher – with quoted prices ranging from £40-£80. While the prices quoted by reprocessors in 1999 for non-plastics PRNs were higher than the actual prices of transactions on the Environment Exchange, the price differential had been eliminated by the end of the following year. In contrast over the same time period, the difference between prices quoted by reprocessors for plastics PRNs and the prices of exchange transactions remained at £17-£19; narrowing to around £7 in the first half of 2001. Of course, the actual prices paid for direct sales may have been lower than the quoted prices. However, the persistence of the price differential together with the higher concentration on the supply side of the market for the plastics PRNs raises the suspicion that reprocessors of plastic packaging may have been engaging in price discrimination between the two market segments.

Figure 3:PRN prices





NOTE: The lines show the moving quarterly averages of the mid-point between high and low quoted prices from reprocessors. The OMEE prices are the annual average price of transactions, except for 2001 which are the averages for the first six months.

Source: Materials Recycling Week; OM Environment Exchange

In summary, the concerns that have been expressed about the scheme do not seem to be borne out, and it appears to be functioning reasonably well. The recovery and recycling targets have been met in every year to date. However, it is true to say that these targets have been relatively easy to achieve, and that the real test of the scheme's environmental integrity will be in 2001, with the step change in the recovery target. There is some evidence to suggest that PRN revenues are filtering through to the waste collectors in the form of higher prices paid by reprocessors for waste packaging material, although this is not statistically significant. However, the relatively low cost of compliance means that there is unlikely to have been much reduction in the quantity of packaging materials used.

Compliance costs have proved to be much lower than was expected when the Packaging Regulations were introduced. The fact that all materials apart form plastics have exceeded the minimum recycling target means that the PRNs for these materials are fungible. This has reduced the potential for market power problems, and there is no evidence of any strategic behaviour in the PRN markets for these materials. There is some evidence to suggest that reprocessors of plastic packaging materials have been acting strategically – indulging in price discrimination between different market segments. However, their ability to do this will diminish as more producers join the Environment Exchange. It also needs to be viewed in perspective. If it led to the price of plastics PRNs being £10-£15 higher than the competitive level in 2000, then this added £2-3 million to total compliance costs (i.e. around 5-7%).

If there is a criticism of the scheme, then it is that the recovery targets were set too low – particularly for the interim years. Not only would higher targets have reduced the concerns over the United Kingdom's ability to meet its obligations under the Directive by the 2001 deadline, the higher PRN prices would have provided a stronger incentive for packaging reduction by producers. In addition, the resultant higher prices for collected waste packaging would have stimulated the development of the waste collection infrastructure.

3.2 Landfill permits for biodegradable municipal waste

a) Policy background and status

The *EU Landfill Directive*[30] came into force in July 1999, and is required to be transposed into national law within two years. The Directive has two broad objectives: the harmonisation of controls on design, operation and aftercare of landfill sites throughout the European Union; and the reduction of greenhouse gases emissions from landfill. In relation to the second objective it sets successive targets for the reduction of biodegradable municipal waste (BMW) that is sent to landfill. For the United Kingdom, these targets imply a substantial reduction for the amount of BMW that can be sent to landfill. For example, if waste arisings continue to grow at 1.5% pa, then by the final target year the implied reduction against the business-as-usual projection is more than 70%.

In the United Kingdom, responsibility for the management of municipal waste is split between waste collection authorities (WCAs) and waste disposal authorities (WDAs). The former are responsible for arranging the collection of waste from households and other premises (usually by a private contractor) and for local recycling activities, while the latter are responsible for arranging its final disposal to landfill or incineration. On average each WDA is responsible for the disposal of

30. Council Directive 99/31/EC

waste from seven WCAs.[31] There is no financial transfer between the two – each is funded by local tax revenues or central government support grants. However, there is a mechanism – recycling credits – by which WCAs receive payments from their WDA for any waste that they divert.

Table 6. Targets in the EU Landfill Directive

Target years[32]	Reduction versus amount of BMW produced in 1995	Implied UK target for BMW sent to landfill (million tonnes)	Reduction versus business-as-usual projections of BMW sent to landfill	
			0% growth	1.5% growth
2006 (2010)	25%	13.1	19.4%	30.6%
2009 (2013)	50%	8.7	46.3%	55.7%
2016 (2020)	65%	6.1	62.4%	72.1%

In response to the Directive, the UK Government issued a consultation document in October 1999 that set out four possible options for implementing the targets (DETR, 1999b). These included two alternative permit-trading schemes; one based on the amount of BMW accepted by landfill operators; the other on the amount of BMW sent to landfill by local authorities. The responses to the consultation document were overwhelmingly in favour of a system of tradeable permits for local authorities (over 70% of responses), and in May 2000 the Government announced that this it would introduce such as scheme in England (DETR, 2000).[33] No firm date has been set for the implementation of the scheme – the Government has indicated that it will bring forward the necessary primary legislation "as soon as the legislative programme allows." However, the Directive requires Member States to have put in place their strategies by July 2003, and it seems likely that the scheme will be up and running by that date.

Work is well underway on the detailed design of the scheme, and in March 2001 a second consultation document (DETR, 2001) was published, which provided details of the Government's initial thinking on a number of issues, and invited comments from interested parties. The following description of the trading scheme is based on this document.

b) Description of the scheme

The permit scheme will be applied to the amount of BMW that is sent to landfill by waste disposal authorities (WDAs). As such, it will seek to influence the demand side of the market for landfill services (see Box 3). Permits will be allocated free-of-charge to WDAs, with each permit providing the holder with the right to send a set amount of BMW to landfill in a specified time period.

A number of alternative algorithms are being considered for allocating the aggregate targets in Table 6 to individual WDAs, and hence determining the number of permits that they will receive. The three main options are based on either the current amounts of BMW sent to landfill, or the total

31. In some areas of the country the responsibilities are combined in a single body – called Unitary Authorities. For simplicity these are treated as WDAs that receive waste from a single WCA. However, they have been excluded from the calculation of the average number of WCAs for each WDA.

32. The UK qualifies for a derogation included in the Directive which allows member states that landfilled more than 80% of their municipal waste in 1995 to extend the target dates by up to four years.

33. The announcement related to England only – the devolved administrations are responsible for implementing the Directive in Scotland, Wales and Northern Ireland. However, they have subsequently followed England's lead in adopting a trading scheme.

level of BMW waste arisings in 1995. The Government's initial preference is to base the allocations for both the initial year of the scheme (e.g. 2003), and the three target years (i.e. 2010, 2013, and 2020), on the current landfill rates. Thus – before trading – all WDAs will be required to make the same proportional reduction in the amount of BMW that they send to landfill.

Interim annual targets will be calculated for the intermediate years – based on linear interpolation between the target figures and an initial quantity that will be set equal to latest year for which data is available. Permits will then be allocated to WDAs for the entire horizon of the scheme (i.e. up to 2020). These permits will be freely tradeable between WDAs. However, there will be no compulsion for WDAs to engage in trading, and authorities may choose to join together to pool their permits (i.e. create a "bubble") as an alternative to trading.[34]

Permits will relate to a particular target year (i.e. vintage). However, to provide greater flexibility, the proposals allow for unlimited banking of permits; except into or across any of the three target years, when no banking will be allowed. This will have the affect splitting the trading scheme into three separate time periods: 2003-2009; 2010-2012; and 2013-2020; with unrestricted banking within each period. However, any permit that is not used by the end of its period will be lost. No borrowing of permits will be allowed, either within or between the three periods. These restrictions on temporal transfers will reduce the cost efficiency of the scheme to a certain degree. However, they are necessary in order to ensure that the United Kingdom complies with its obligations in each of the three target years.[35]

The initial ownership of permits, and all subsequent transfers, will be recorded electronically on a central registry that will be maintained by the UK Environment Agency. Only WDAs will be allowed to hold permits, and a transfer must be recorded on the registry before the buying authority can use the acquired permits for compliance purposes. The proposals make a distinction between the transfer of permit ownership and trading. Transfers may occur without a trade (i.e. a financial transaction), and there may be trades that do not result in a transfer. For example, call options may not be exercised at the end of their term. A wide range of trading arrangements are envisaged – covering direct bilateral trades between individual WDAs, and indirect trades via brokers and electronic exchanges, and there is no intention to impose any restrictions on the resultant market price of permits. Information on the average prices of indirect trades should be widely available in the market. In addition, it is proposed that where trades result in a transfer of ownership, the price will be recorded on the registry, and aggregate price information will be made available periodically by the Environment Agency.

WDAs will be required to submit annual landfill returns to the Environment Agency, providing details of the quantity and composition of the municipal waste that they have sent to landfill in that year. These returns will be cross-checked against returns submitted by the landfill operators, detailing the total amounts of municipal waste that they have received from each WDA. Due to the problems with measuring the biodegradable content of waste, it is proposed that it will be calculated using a standard percentage (currently 60%). However, this value will be kept under review, and work will be undertaken in to develop a protocol that could be used to calculate the biodegradable content for individual authorities.

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34. WDAs are public bodies that may have a range of motivations in addition to cost minimisation. As such, they may prefer to meet their own diversion targets even if this is more costly than purchasing additional permits.

35. The Directive requires that member states achieve the stipulated reductions in each target year, not an equivalent cumulative reduction by the final year.

At the end of each year, the landfill returns will be reconciled against the permit holdings on the registry, and any WDA that holds insufficient permits will be deemed to be out of compliance and will be subject to the imposition of sanctions. The current proposal is for a system of civil financial penalties to be included in the legislation. However, provision would be made for these penalties to be waived or suspended by the Secretary of State conditional on the authority taking corrective action. In addition, the offending authority may be subject to intervention under the Best Value regime.[36]

Box 3. Conceptual framework for the Landfill Permit System

Rules

The Landfill Permit (LFP) system is a "cap and trade" system, with an absolute target. Both the generation rule and the performance rule are applied to Waste Disposal Authorities (WDAs).

Rule	Definition			Assignment
Generation	LFPs sold	≤	Initial allocation	WDAs
Performance	LFPs purchased	≥	BMW sent to landfill	WDAs

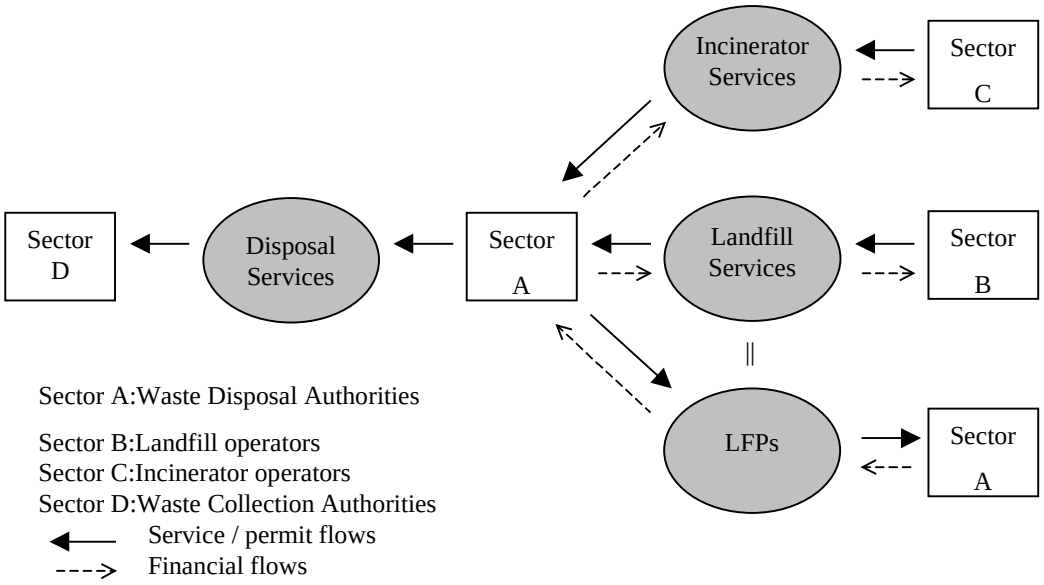
Trading system structure

The trading system comprises four markets and four sectors. The target market is the "complete" market for landfill services, which comprises WDAs (on the demand side) and landfill operators (on the supply side). The market for disposal services is an "incomplete" market, in that the WDAs receive no payment from the WCAs for the disposal service that they provide. The market for incinerator services is included in the trading system because any reduction in the amount of waste sent to landfill results in a corresponding increase in the amount sent for incineration.[37]

36. The Best Value regime was established by the Local Government Act 1999, and requires that local authorities "make arrangements to secure continuous improvement in the way in which its functions are exercised, having regard to the combination of economy, efficiency and effectiveness.". If an authority does not meet its performance targets, then the Secretary of State may direct the authority to prepare a performance plan; instigate a public inquiry; direct the authority to take specific actions; take control over the management of certain local authority functions.

37. For simplicity, it is assumed that the demand for disposal services is fixed, and that incineration is the only viable alternative disposal option for the WDAs.

Box 3 (continued)



Short run impacts

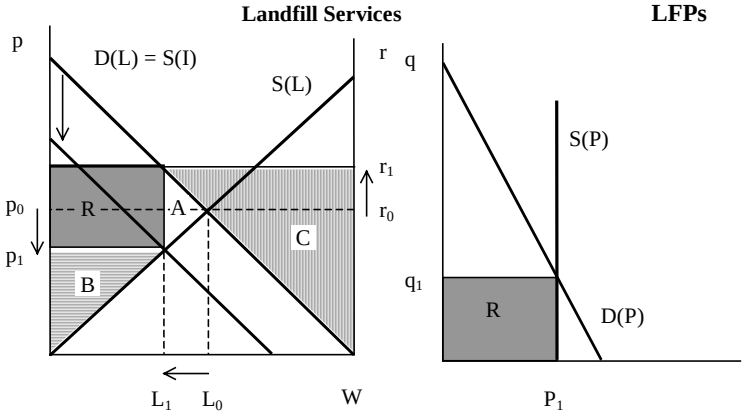
For a fixed amount of total biodegradable waste (W), the demand curve for landfill services D(L) is given by the supply curve for incineration services S(I). The price of landfill services (p) is shown on the left-hand vertical axis, and the price of incineration services is shown on the right-hand axis. Prior to the introduction of the scheme, the two prices are equal (i.e. $p_0 = r_0$), and the total disposal cost of the WDAs is equal to $p_0 \times L_0 + r_0 \times (W-L_0) = p_0 \times W$.

The supply curve in the LFP market is vertical, reflecting the fact that there is an absolute limit for the amount of waste that can be sent to landfill, while the demand curve is given by the difference between the demand and supply curves for landfill services (i.e. D(P) = D(L) - S(L)).

The introduction of the trading scheme causes the demand curve for landfill services to shift downwards, by an amount equal to the LFP price (q1). This causes the price of landfill to fall, and the price of incineration to rise, so that in the new equilibrium the difference between the two prices is equal to the LFP price. The amount of waste sent to landfill decreases from L0 to L1 (= P1), with a corresponding increase in the amount going to incineration. The operating profits of the landfill operators (B) decline, while those of the incinerator operators (C) increase. Excluding the (notional) revenues from the permit sales (R), the WDAs suffer an increase in total disposal costs. However, when these revenues are taken into account, the impact is ambiguous, depending on the relative slopes of the supply curves for landfill and incineration. In aggregate, there is a net loss – given by the triangle A.

If the two rules had been applied to the other side of the target market, then the resultant distribution of the (unchanged) net loss would have been different – with the WDAs suffering an increase in total disposal costs, and the impact on landfill operators being ambiguous. However, if the initial property rights had been assigned to the government (or another third party), then the distribution of costs and benefits between the three sectors is unaffected by the assignment of the performance rule.

Box 3 (continued)



c) Assessment of the scheme

Box 3 provides a simplified *ex ante* analysis of expected impacts of the Landfill Permit scheme. At this stage it is of course not possible to provide an *ex post* evaluation of the scheme. However, there are two important questions that can be addressed.

- Is the scheme capable of delivering the target reductions?

Achievement of the targets will require a fundamental change in the way that municipal waste is managed. There are two potential concerns over the capacity of the proposed scheme to deliver these changes – its ability to induce behavioural changes at all stages of the waste pipeline, and the credibility of the sanctions for non-compliance.

The permit scheme applies to WDAs. In practice their only viable diversion option is incineration with energy recovery. While there will probably be sufficient capacity to meet the diversion targets for the first target year, it is highly unlikely that it will be possible to meet the final year targets by incineration alone. Even if it was possible to do so, it is unlikely that it would be cost efficient. Therefore, both on the grounds of environmental effectiveness and economic efficiency, it will be necessary to develop a mix of diversion activities which includes those (such as composting) that can only viably take place at the point of waste collection.

Developing the appropriate mix should not be too problematic for the eighty or so WDAs that are also responsible for waste collection.[38] For the remaining forty WDAs the Government hopes that the requirement to develop Municipal Waste Management Strategies in conjunction with their

38. These are known as Unitary Authorities.

respective WCAs will foster closer working relationships and encourage an integrated approach to waste management. However, with each WDA having an average of seven upstream WCAs, this administrative process will inevitably be bureaucratic and inflexible, and is unlikely to yield the optimal levels of diversion at the collection stage. An alternative approach – which has not yet been considered – would be to extend the current system of recycling credits, and link it explicitly to the landfill permit system. Under the current system, the value of recycling credits is set equal to the (avoided) cost of landfill. However, if instead it were to be set equal to the cost of landfill plus the market price of landfill permits, then this would provide the correct incentive for WCAs to undertake diversion activities. Under this approach, it would be worthwhile for WCAs to undertake diversion activities up to the point at which the marginal cost is equal to the value of the credits, which is itself equal to the marginal cost of incineration to the WDAs.

Of course, this relies on the assumption that WDAs will comply with the performance rule. This in turn relies on the existence of credible sanctions in the case of non-compliance – i.e. penalties that are both meaningful and certain. The proposal to allow the Secretary of State to waive or suspend penalties undermines this credibility, particularly given the fact that the WDAs are a local government body. There may be political pressures to avoid imposing additional costs on local authorities, and hence increase local taxes or reduce services. As such, there may be a tendency to give offending authorities a second chance. While persistent offenders will be subject to intervention under the Best Value regime, it is unlikely that this will be as effective a deterrent as a certain financial penalty.

- Is it sensible to apply the scheme to the WDAs?

The Government's original consultation document put forward two options for a permit-trading scheme – the WDA scheme that has been pursued, and an alternative scheme under which permits providing the right to receive biodegradable waste would have been applied to landfill operators. As was discussed in section 2, the two approaches represent alternative choices for the assignment of obligations and initial property rights. The WDA scheme received overwhelming support in the consultation exercise. To that extent one can assume that it was the better option in terms of political acceptability. However, is it necessarily the best choice in terms of the distributional impacts and economic efficiency?

As the analysis in Box 3 demonstrates, the increase in the total disposal costs of the WDAs would have been greater if the scheme had been applied to the landfill operators. Consequently, if the Government is concerned primarily with the impact on public expenditure, then it would appear to be correct in its decision to apply the permit system to the WDAs.

This view is reinforced when one considers the potential for market power problems. There are around 120 WDAs, and although some doubts have been expressed about their commercial ethos (and hence likelihood to trade), there is no reason to suspect they will act strategically. However, the same may not be the case for the landfill operators. A report on the competitiveness of the UK waste management industry (DTI, 1997) shows that while there are around 1,200 licensed landfill operators in the United Kingdom, the combined market share of the largest four companies is 25%, and the largest ten is 40%. The continuing acquisition of local authority sites by private sector companies, and the barriers to new entrants are expected to increase the level of concentration in the sector. Furthermore, the relatively high cost of transporting waste around the country, together with the application of the "proximity principle" to WDAs, means that the landfill market is effectively a series of local markets. This gives the individual landfill operators a degree of local monopoly power. The implications of this relatively complex market structure for behaviour in the permit market, and for the interaction between the permit and landfill markets, are not totally clear. However, it seems likely that the cost efficiency of a permit-trading system applied to the landfill operators would have been undermined by strategic behaviour.

4. Conclusions

The waste management system is prone to a number of failures. In particular, the practice of funding domestic waste collections out of general taxation means that households have no incentive to control the amount of waste that they produce (a government failure). This has been exacerbated by a failure to incorporate fully the environmental costs associated with waste disposal and other waste management activities in market prices (a market failure). Together these failures have resulted in excessive amounts of waste being generated, and insufficient levels of recycling and recovery.

In response to these failures, increasing use is being made of volume charging for households. However, its adoption is far from universal, and even where it has been implemented it is not practicable to differentiate the charge between different types of waste with different environmental impacts. Consequently, there will continue to be a need for other policy responses – either as a substitute for volume charging, or as a complement to it. A wide range of alternative policy responses have been adopted, or proposed. Some of these are price-based (e.g. the UK landfill tax). The more common approach however has been to adopt one (or more) of three quantity-based responses: recycled content standards (or procurement standards); recycling targets, particularly under extended producer responsibility; and landfill restrictions. Ultimately the costs imposed by these responses will depend on the levels chosen for the targets (which should be based on proper cost benefit analysis). However, they will also be effected by the flexibility of the mechanism that is used to implement them. Permit trading provides an ideal way of providing this flexibility, and it can be used for all four types of quantity-based response.

Permit trading schemes (of various types) have been proposed to implement recycled content standards (e.g. newsprint in the USA), and reuse targets / extended producer responsibility (e.g. beverage containers in Germany, and waste electrical and electronic equipment in Europe). In the United Kingdom a tradeable compliance credit scheme has been in operation since 1998 in relation to the recovery of packaging materials, and a permit trading scheme to implement reductions in the amount of biodegradable waste that is sent to landfill is in the final design phase. The former appears to be functioning reasonably well, and it demonstrates that permit trading can be used in waste management applications with relative targets, such as recycled content goals and extended producer responsibility initiatives.

While tradeable permit systems can take many different forms, they can all be classified into one of two broad approaches – "cap and trade" systems and "baseline and credit" systems. Under either approach, the system is defined by two rules: a generation rule which determines the supply of permits / credits, and a performance rule which determines the demand.

The choices open to regulator regarding the definition of these two rules, and their assignment to particular sectors will depend on a variety of factors that are likely to vary between policy applications. In particular, there are important differences between traditional policy applications such as emissions of air pollutants, and waste management applications. In traditional applications, the supply side of the target market is missing; the objective is to reduce the level of activity; and in most cases, the objective is expressed in absolute terms. In contrast, in all of the waste management applications identified above, the object of the intervention is a complete market (i.e. both the demand and supply curves exist). Furthermore, with the exception of landfill restrictions, the aim is to increase activity, and the objective is usually expressed in relative terms. These differences result in the choice sets for waste management applications being very different to those of traditional applications.

It is possible to use either trading approach to implement landfill restrictions. However, when applying permit trading to recycled content standards, recycling targets and extended producer responsibility – where the aim is to increase activity and the objective is expressed in relative terms – it is only possible to use the "baseline and credit" approach. On the other hand, the fact the target

market is complete, means that the regulator can choose whether to intervene on the demand side or the supply side of the market. When making this choice, careful consideration should be given to the implications for the distribution of costs, and to the need to minimise market power problems.

In traditional applications of permit trading, the choice regarding the assignment of initial property rights has implications for the distribution of costs within the target sector, and between the government and the target sector.[39] When the target market is complete however, any intervention will impose costs on both sides of the market. Consequently, in the case of waste management applications, the assignment of initial property rights also can have implications for distribution of costs between sectors. If both sides are in the private sector, the regulator may be indifferent between the two options. However, if one side is in the public sector (as is the case with the waste disposal authorities in the UK market for landfill services), it may be desirable to choose the option that minimises the cost to this sector.

While the distribution of costs is clearly a significant issue, it is also important to compare the potential for market power problems under the different options. Strategic behaviour in the permit market, the target market, or the output / input markets of the participating sectors, can affect the cost efficiency of the outcome. If the target is expressed in relative terms, then it can also affect the environmental outcome. In traditional applications, market power can arise in the permit market, or the product market, or both. In practice, there is probably little that the regulator can do about the problem, other than choose an alternative implementation mechanism. Of course, in these applications the issue of market power in the target market does not arise. However, in many waste management applications this market may be subject to significant imperfections – on one side or the other. The market analyses conducted for the potential application of trading to the recycled content of newsprint in the USA, and for the actual applications to EPR for packaging waste and landfill targets for biodegradable waste in the UK, suggest that strategic behaviour can be a significant issue. Fortunately, the flexibility to apply the trading scheme to either side of target market provides regulator with an opportunity to address the problem without having to resort to an alternative mechanism.

The potential for market power problems may depend on the definition of policy objective. For example, the experience of the UK PRN system suggests that it is less likely that material reprocessors will be able to exercise market power in a permit trading system when there is an aggregate recycling target, than when there are a series of material-specific recycling targets. Thus, while the regulator may be indifferent between assigning the property rights to the material reprocessors or the waste collectors under an aggregate objective, it would be better if they were assigned to the waste collectors under material-specific targets.

Finally, while it may not be possible to achieve the optimal solution with a single policy response, it is desirable that the impact of the intervention should spread through the waste management system (i.e. to cause both an increase in diversion and a reduction in waste generation). In some cases – such as recycled content standards and extended producer responsibility – it may be sufficient to rely on the resultant changes in market prices to provide the necessary incentives. However, where markets are missing it may be necessary to develop additional mechanisms to complement the permit system. For example, the impact of the UK landfill permit system could be extended upstream to the waste collection authorities by explicitly linking the value of recycling credits to the price of the permits.

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39. When the initial property rights are assigned to the target sector, the foregone revenue should be treated as a cost to the government.

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ANNEX 1: BREAKDOWN OF MUNICIPAL WASTE FOR SELECTED OECD COUNTRIES

a) Composition of waste arisings

Country	Year	Paper and board	Food and garden waste	Plastics	Glass	Metals	Textiles and other
USA	1997	38%	24%	9%	6%	8%	15%
Japan	1990	38%	32%	11%	7%	6%	7%
Korea	1996	13%	33%	11%	3%	3%	37%
France	1997	25%	29%	11%	13%	4%	18%
Germany	1997	41%	23%	3%	22%	8%	3%
Netherlands	1996	27%	39%	5%	6%	2%	20%
Spain	1997	21%	44%	11%	7%	4%	13%
UK	1990	37%	19%	10%	9%	7%	18%

Source: OECD (1999), table 7.2B

b) Disposal and recovery routes (*)

Country	Year	Landfill	Incineration without energy recovery	Incineration with energy recovery	Recycling	Composting
USA	1996	55.4%	0.7%	16.5%	21.9%	5.4%
Japan	1993	29.7%	75.6%		4.2%	0.1%
Korea	1996	68.3%	3.3%	2.2%	26.2%	-
France	1995	46.1%	13.0%	36.8%	1.9%	8.3%
Germany	1993	51.3%	17.4%		23.3%	5.4%
Netherlands	1996	20.3%	30.9%		16.7%	24.7%
Spain	1996	76.8%	0.5%	4.1%	0.6%	15.6%
UK	1996	83.4%	2.6%	5.9%	5.8%	1.2%

Source: OECD (1999), table 7.2C

(*) Figures may not sum to 100% due to double counting of residual waste from incineration that is landfilled, and / or omission of other diversion routes.

ANNEX 2: MARKET POWER AND DESIGN OF TRADEABLE PERMIT SYSTEMS

The economic efficiency of a permit trading scheme is likely to be undermined if firms can take advantage of their market power to influence prices in any of the constituent markets (i.e. act strategically). In the context of traditional permit trading applications, Hahn (1984) has considered the implications of market power in the permit market, while Mauleg (1990) and Sartzetakis (1997) have investigated the interactions between permit and product markets. Of course, the issue of market power in the target market does not arise in these applications. However, in many waste management applications this market may be subject to significant imperfections – on one side or the other. In these situations the choices that are made regarding obligations and initial property rights may have a significant bearing on the outcome.

The issue is investigated by Salmons (2001) in the context of a credit-trading scheme that is used to implement a recycling target for a particular material under extended producer responsibility. A simple analytical model is used to investigate how choice of sector to receive the initial property rights can affect the outcome when firms are able to act strategically in one of the sectors. The trading system that is assumed for the analysis is essentially the same as that outlined in Box 2, except that there is only one obligated sector, and the assignment of the generation rule is left open.

As one would expect, when there is perfect competition in both the target market and the credit market (i.e. all firms are price takers in both markets), the assignment of the generation rule has no impact on the outcome. The physical flows (i.e. the amount of material reprocessed, etc.) and the total costs are identical – although the distribution of costs differs between the two cases.

However, when there is only one reprocessor – which can act as a monopsonist in the target market and a monopolist in the credit market – the outcomes differ depending on which sector receives the initial property rights. Unfortunately, it is not possible to draw any definitive conclusions from the generic analysis regarding the sign and magnitude of these differences. Consequently, Salmons uses a simple simulation to explore the potential implications of the different property rights regimes.

Under the assumptions used for the simulation, the "natural" recovery rate for the material is 11%.[40] Figure A1 shows selected results of the market simulation for different values of the target

40. The following assumptions are used in the simulation. The production function for reprocessed material is given by $y = k^{1/2} w^{1/2}$ where k is capital and w is collected waste material; the aggregate production function for waste material is given by $w = l^{1/2}$ where l is labour; the producers' aggregate demand for the material is given by $D(z) = 10 - z$ where z is the amount of material used; and $p_z = p_k = p_l = 1$, $p_y = 4$ are the exogenous prices of z, k, l and y respectively.

recovery rate above this level. It is clear form these results there are significant differences in the outcomes, depending on whether the property rights are given to the reprocessor (RM) or the collectors (WC). Interestingly, for most variables the differences diminish as the target recovery rate increases

The first two panels show the impacts on the equilibrium prices of credits and collected waste material. There are two obvious points to note. Firstly, both prices are higher if the initial property rights are given to the reprocessor. Secondly, the impact of a rise in the target recovery rate is very different under the two regimes. When the reprocessor has the initial property rights, the price of credits falls continuously as the target recovery rate increases, while the price of collected waste rises. In contrast, when the rights are granted to the waste collectors the situation is reversed, with the price of credits rising and the price of collected material falling – at least until the target recovery rate climbs above 65%.

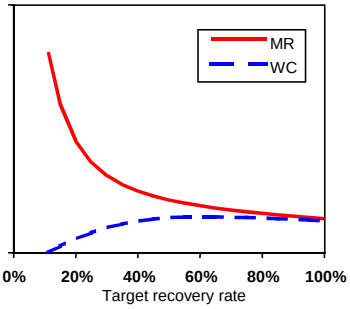
The next three panels show the impacts on the flows of the material within the waste management system. The imposition of the performance rule on the producers has the effect of increasing the price of the material – by an amount equal to the credit price multiplied by the target recovery rate. Consequently, the higher credit price when the reprocessor has the initial property rights results in less material being used in production (i.e. greater source reduction). Thus, for any given recovery rate, the amount of material reprocessed and the amount going to landfill are both lower. Under both property rights regimes, the amount of material going to landfill decreases continuously as the target recovery rate increases. However, when the initial property rights are granted to the waste collectors, the amount of waste material reprocessed peaks at a recovery rate of around 60%, declining thereafter.

The better environmental outcome when the reprocessor has the initial property rights is only achieved at the expense of a higher economic cost. As can be seen in the final panel, the aggregate cost of achieving any given recovery rate is higher – with higher profits for the reprocessor being outweighed by lower profits for the producers and for the waste collectors. When the rights are assigned to the waste collectors, aggregate profits actually increase for relatively low recovery rates (i.e. up to 30%). This is because at these levels, the scheme is also effectively acting as an instrument of market regulation – reducing the deadweight loss arising from the monsopony power of the reprocessor (with the producers rather than the government providing the subsidy). Provided that the waste collectors have the property rights, this gain is sufficiently large to outweigh the cost imposed on the producers (i.e. the reduction in their profits). However, when the reprocessor has the property rights, the scale of the efficiency gain in the target market is reduced, reflecting the smaller increase in the amount of waste material reprocessed. The gain is no longer sufficient to offset the cost imposed on the producers, which is itself higher as a result of the reprocessor's monopoly power in the permit market.

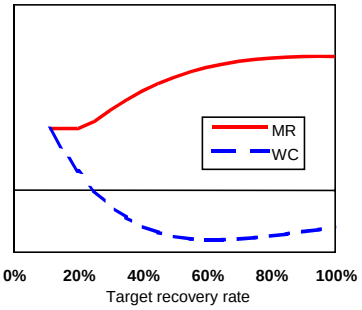
Of course, these results reflect the particular assumptions that were made for the simulation. However, they illustrate clearly the potential sensitivity of the economic and environmental outcomes to the assignment of initial property rights when there is asymmetric market power in the target market.

Figure A1: Selected results of market simulation

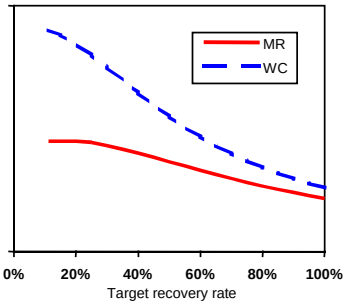
A) Price of credits (£)



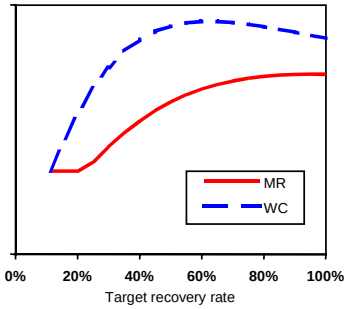
B) Price of collected waste material (£)



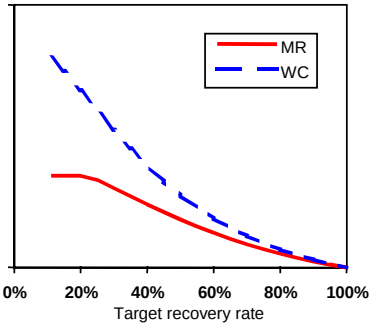
C) Material used by producers (tonnes)



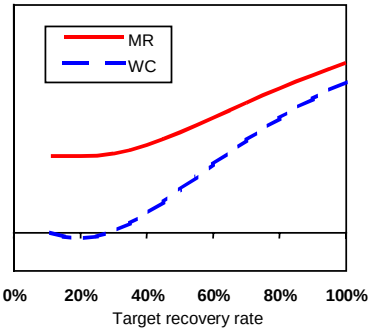
D) Waste material reprocessed



E) Waste material going to landfill (tonnes) Cost (£ million)



F) Aggregate cost (£ million)



Chapter 7

TRADEABLE PERMITS IN WATER RESOURCE PROTECTION AND MANAGEMENT

by

Andreas Kraemer, Eduard Interwies and Eleftheria Kampa
Ecologic, Institute for International and European Environmental Policy, Berlin

1. Introduction

The present review aims to analyse recent developments on the wider application of tradeable permits in water resource management in OECD Member countries. It builds on the findings of Kraemer and Banholzer (1999) on the use of tradeable permits in water resource management and water pollution control providing updates and recent developments of the trading programmes previously reviewed in that paper. Additionally, it discusses new schemes of tradeable permits on water consumption and pollution control which are already under implementation or under consideration and provide significant lessons learned. An effort has been made to present the schemes of tradeable water rights in relation to the existing (or potential) water allocation mechanisms of the respective study areas.

The description and discussion of each programme of tradeable permits attempts to cover as much information as possible on the institutional set up of the programme, the preparation process that led to its establishment, as well as on the nature of permits, programme participants, allocation method and monitoring of the trading rules. Comments on the advantages and potential drawbacks of each scheme are also included where appropriate.

Nevertheless, it should be pointed out that due to the scarcity of information sources on some of the schemes discussed, not all the above points could be covered. After the analysis of published and unpublished material, updates and new information on schemes implemented in the US and Australia have been included [see Annex for key information on case studies reviewed by Kraemer and Banholzer (1999)]. Significant recent developments on the schemes of Chile, as included by Kraemer and Banholzer (1999), have not been found. However, information on two more OECD Member countries, Mexico and Spain, has been incorporated, as well as a summary of recent considerations on the trade of water abstraction licences in the UK.

Regarding the structuring of the case studies in this report, the same typology system has been followed as by Kraemer and Banholzer (1999). Therefore, the case studies have been discussed as in two separate fields of application: tradeable water abstraction rights for quantitative water resource management and tradeable water pollution rights for the protection of water quality.

The first section of the paper discusses the nature of market structures in water resource management systems, with specific reference to the institutional settings and experiences in the US, Australia, Mexico and Spain. The following section introduces case studies of tradeable permit schemes (updates, new, considered and hypothetical) in the water resource sector. A brief section with suggestions on potential further applications of water-based tradeable permits has also been added,

focusing mainly on trading at an international level. The paper finally dwells on the reasons for resistance to water trading so far, closing up with a section of conclusions based on this review.

2. Policy backgrounds

There are diverse experiences with water rights trading as well as different regulatory regimes in place regarding the use of water in different countries. It is, therefore, difficult to speak of a general policy background to water-based tradeable permits.

Water trading has developed in different parts of the world mainly under the pressure of water scarcity and focuses on different sectors, as far as the institutional context allows to do so. In Australia, for instance, trading of water rights has focused on agricultural use, whereas in the US, water rights have also been traded between different types of water users.

For the purposes of this review, therefore, the policy background to water trade shall be briefly discussed on a country by country basis. The federations of Australia and the US are the two OECD Member countries which so far have accumulated considerable experience with water-based tradeable permits. Although several markets have been established on rights to use (and pollute) water in some of the individual States of the two federations, there have recently been efforts to provide guidelines and a policy framework concerning water trade at the federation level.

The experience of two other OECD Member countries, Spain and Mexico, on tradeable water abstraction rights is also discussed on the national policy level and within individual study areas. Mexico provides an interesting case of new experiences on water markets after the 1992 National Water Law was initiated and provided for the decentralisation of water resources, the creation of Water User Associations, and the formalised transfer of water use concessions between individual irrigators. In Spain, on the other hand, local, historically-grown water markets have existed for a long time. However, a new water law which aims to incorporate market systems in water management at a national level only came into force in 1999.

Informal, small-scale water trading among farmers has also been common throughout the developing world (Barlow, 1999). The informal markets for groundwater in the irrigated areas of southern Asia are especially well known and function apparently rather effectively in India (Saleth, 1998) and Pakistan (Meinzen-Dick, 1998). However, since the focus of this review is on OECD Member countries, the institutional development and experiences gained from these systems will not be further discussed here.

2.1 United States

In the US, water rights and water allocation are the domain of the States, and trading schemes vary considerably between States. Some of the world's most developed and progressive water markets are located in the western United States (Landry, 2000), mainly because the growing population, water scarcity, and a mix of water allocation systems based on prior appropriation allow for well regulated market transfers (Hearne and Trava, 1997).

In the western and south-western States, the first to use a water resource acquires a *senior right* to use a certain quantity of water, according to the "*prior appropriation" doctrine* and the principle "first-in-time, first-in-right". Further interested parties are later allocated *junior rights* which provide them, however, with lower water right security (Kahlenborn and Klaphake, 2001). Water rights are

also allocated on condition that the water is put to "*beneficial use*",[1] in connection with the rule "use-it-or-lose-it". Thereby, rights that are not used for a certain period of time are withdrawn. Further restrictions on water rights are placed by the doctrine of *preferential use*, according to which the States list the water uses that gain priority at times of water scarcity. Nevertheless, the strict requirement for beneficial use has been recently loosened in some western States such as California, Idaho, Montana, Washington and Oregon, to allow for the allocation and exchange of water rights for environmental protection (instream flows) (Landry, 1998).

The southern State of Texas has gained some experience with water markets with the most important one having emerged in the Rio Grande Valley. Other programmes discussed here are the California Drought Water Bank and the market of the Northern Colorado Water Conservancy. The latter has been one of the most interesting and active water markets in the State of Colorado.

Other States such as Utah, Nevada and Oregon have also created water markets to bring demand and supply of water into balance. An interesting recent development has been the interstate trade which has begun to form on the Colorado river. California, Nevada and Arizona are entitled to shares of the river water, according to the 1922 Colorado River Compact. California and Nevada face water shortages and high costs for alternative supplies, while Arizona has cheap surplus water. Recently, an agreement between States allows Arizona to market a portion of its Colorado river allotment to California and Nevada (Landry, 2000). Under the agreement, California and Nevada may store river water in Arizona's underground aquifers, banking it for the future (Landry & Anderson, 1999). Until now, the State of California has benefited from unused allocations by upstream States, which run downstream to California. With the bank in place, Arizona and Nevada can start charging California for water or start storing it (Larry & Anderson, 1999). However, in case of such large-scale transfers of water to users of different geographic regions, third-party impacts must be taken into consideration by restrictions placed by the regulator (Schiller & Fowler, 1999).

Regarding the trading of emissions to water, despite the considerable effort by the US EPA and individual states to implement the concept, it has yet to live up to its full promise (NCEE, 2001). EPA, in particular, has been on the forefront of the effluent trading concept and it composed a set of guidelines for developing trading programmes in 1996 (EPA, 1996a). New efforts by the EPA to implement its so far little-known provision for Total Maximum Daily Loads (TMDLs)[2] in areas with impaired water quality are expected to vastly increase the use of effluent trading (NCEE, 2001), so as to lower compliance costs of affected sources. Parties to the water trading negotiate within the overall loading capacity determined under the TMDL. Trades can occur within TMDLs through development of final allocations among participating sources or, if a TMDL is already in place, by revisions of allocations to reflect proposed changes in individual load reduction responsibilities by trading (EPA, 1996).

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1. During the 1800s, beneficial uses included mining, ranching, farming and manufacturing (Schiller & Fowler, 1999).

2. A Total Maximum Daily Load (TDML) should be developed by States and is the process under the Clean Water Act that establishes the maximum pollutant load a water body can receive without violating water quality standards. A TMDL describes how much pollution can be discharged into a water body and who is allowed to discharge it. It includes allocations of pollutant loads among sources: wasteload allocations for point sources, load allocations for non-point sources, background loadings from natural sources, and margins of safety to ensure achievement of water quality goals (EPA, 1996). States establish TMDLs for every location that will not meet water quality standards given the current regulatory framework.

In brief, under the 1972 Water Act, each State must identify polluted waters and establish TMDLs, which determine the amount by which sources of pollution would need to be reduced to meet the State's ambient water quality standards. In 2000, the EPA published final new rules concerning the TMDL Programme, which have not yet taken effect. The new rules provide for a more comprehensive list of impaired waters which should be submitted every 4 years. It also strengthens efforts to put in place clean-up actions and sets goals of attaining water quality standards within 10 years, if possible. Across the US, approximately 40,000 new TMDLs are now required of which about 45% include non-point sources. This brings new emphasis on non-point sources, which have not yet been forced to reduce their pollution loads (Young & Karkoski, 2000).

A non-point source is essentially any source of pollutants that is not a point source, for instance sheet flow from pastures and runoff from paved areas. A point source is a discrete discharge of pollutants through a pipe or similar conveyance, usually associated with industrial and sewage treatment plants. Recently, the impact of non-point sources of pollution on water quality has been increasing in relation to the impact of point ones. As an indication, 43% of the US waters included in the 1998 Impaired Waters List were impaired because of non-point sources, only 10% because of point sources, and 47% due to a combination of point and non-point ones (EPA, 2001). There has so far been an imbalance between non-point sources, such as agriculture, that have not yet been forced to reduce pollution loads, and point sources that have made significant investments in pollution reduction (Young & Karkoski, 2000). To date, most programmes aiming at reducing non-point agricultural pollution have relied on voluntary adoption of 'Best Management Practices' (BMPs), which are technologies or management measures that have been identified as effective in preventing or reducing pollution and usually consist in pesticide and fertiliser storage and handling, nutrient management, drainage management, and even wetland restoration. However, a major disadvantage has been that voluntary BMPs are unlikely to yield adequate results unless the required pollution controls are inexpensive (Young & Karkoski, 2000).

In the following sections, recent developments on the effluent trading programmes of the Dillon Reservoir, Cherry Creek Reservoir and the Tar Pamlico Basin, as well as the regional scheme of selenium trading among farmers in the California San Joaquin Valley are discussed. The US EPA and several States have recently been actively involved in a number of other on-going and under development effluent trading projects. The Chesapeake Bay Nutrient Trading programme will be described here in more detail, discussing the recently developed fundamental principles for trading within this scheme of multi-state character.

2.2 *Australia*

In the context of the water reform in the Commonwealth of Australia, water authorities seem increasingly prepared to work within a market-oriented resource management framework (Pigram, 1999). In Australia, water trading schemes operate in Queensland, South Australia, Victoria and New South Wales (NSW). Transactions are almost totally confined within the irrigation sector, while intersectoral transfers, e.g. rural to urban water use, are the exception. Certain constraints on trade such as spatial and volumetric restrictions, are prompted by a fear of growth in corporate ownership of water rights and possible adverse economic and social effects for regional communities (Pigram, 1999).

An issue recently receiving attention in Australia is provision of water for environmental needs, so as to fulfil flow requirements for aquatic ecosystems. More specifically, there has been a debate concerning the interface between consumptive water entitlements, the environment and non-consumptive users. Both environmental users, such as organisations aiming to conserve aquatic ecosystems, and non-consumptive users, such as the tourism industry and commercial and recreational fishing in rivers and estuaries, seek to secure stronger definitions of their rights in law. It

is suggested that more work needs to be done on mechanisms which meet both the needs of the environment and the concerns of water users (Productivity Commission, 2000). Rights to environmental allocations would need to be assigned in the form of a licence to a Government or independent body responsible for managing those rights. Any water entitlement assigned to the environment actually needs to be assigned to a body able to represent and make decisions on behalf of environmental interests (Sieber et al, 2000). In this context, there is also a pressing need for rules governing water trade that will recognise environmental constraints to trade.

In the three States of south-eastern Australia (NSW, Victoria and South Australia), water trade started to develop almost in the same time in the 1980s. In NSW and Victoria, temporary trade was introduced first, while permanent trade was introduced later in 1989. In South Australia, permanent trade was introduced from the outset, since it was considered necessary for long term investment in the horticulture and viniculture of the State (Bjornlund & McKay, 2000).

In this paper, a short description of the water trading framework in the south-east of the State of South Australia is provided as well as a discussion of the current status of the irrigation trading scheme in NSW. In NSW, trading is now the dominant channel for re-allocation of water, mainly for irrigation. As an indication, in 1997/98, 10% of the total entitlement for consumptive water use was traded (NSW DLWC, 1999). In Australia, in general, temporary trade has been more common than permanent one. Permanent trade accounts for only 10% of temporary trade by volume, in both the Murray Region of NSW and within the Gouldburn Murray Irrigation District in Victoria (Young *et al*, 2000). Since there is on-going institutional development and therefore insecurity regarding water market and water entitlements, market participants are discouraged from trading water permanently.

The Murray-Darling Basin Commission, which co-ordinates the interstate co-operation over the river Murray, initiated a Pilot Interstate Water Trading Project in 1997, in order to assist the progressive introduction of permanent interstate water trade (Cummins & Associates, 2000). The experience gained in the first two years of this pilot project has been reviewed (Young et al., 2000) and is discussed here briefly. Concerning trading with emissions to combat water pollution, the NSW EPA is progressing with the Hunter Salinity Trading Scheme. It has now established an internet site to facilitate trades. The system of salinity credits and debits for interstate salinity trading in the Murray Darling Basin is also continued.

Finally, concerning the South Creek Bubble scheme in the Hawkesbury-Nepean river, the results for the first three years of its operation have been rather positive. The activities carried out in NSW within this scheme are being conducted within a state-wide reform programme of the NSW Government aimed at achieving clean, healthy and productive use of water (NSW EPA, 2001c).

2.3 Mexico

In 1992, Mexico initiated a new national water law which decentralised water resources management and allowed the transfer of water use concessions between individual irrigators. The new water law coincided with a series of policy reforms including the privatisation of communal land holdings (*ejidos*) in the agricultural sector, the transfer of canal systems operation to water user associations (WUAs), the revision of the role of the National Water Commission (CNA), and in general more liberalised trade policies (Hearne & Trava, 1997). The Constitution was modified and a new agrarian law was passed that redefined land property rights and stimulated commercialisation of (ejido) land and national waters (Kloezen, 1998). The new water law also facilitated the creation of WUAs which are granted titles of water concessions and permits to use irrigation infrastructure by the CNA (Hearne & Trava, 1997).

In Mexico, there is a clear difference between water property rights and concessions. Property rights of water are exclusively held by CNA, but concessions are granted to individual users and WUAs. The concessions have a maturity of 30 years, and can be bought or sold freely as long as transactions do not negatively impact the water rights of others (Schiller & Fowler, 1999). The water law also defines that the tradeable part is not limited to consumptive use only but is the full share including return flows.[3] The law does not provide for third-party rights to return flows, which means that return flows are made available to downstream users at no charge (Kloezen, 1998).

The 1992 National Water Law maintains a central role for the CNA in the management of water and the regulation of transfer of water-use concessions. The federal government therefore continues to play a dominant role in the regulation of water-use within an administered system. The CNA still has considerable power to restrict water use, especially in times of emergency water shortages. However, it is argued that the discretionary powers of the CNA should be used with moderation, so as not to undermine the benefits of secure water-use concessions (Hearne & Trava, 1997). Private transferable water-use concessions are granted to individuals, WUAs and firms for a period ranging from 5 to 50 years (renewable) (Hearne & Trava, 1997). By gaining concessions, WUAs legally assume operation and maintenance responsibilities (Kloezen, 1998). Within the Irrigation Districts, the CNA initially allocates water to WUAs, which are organised in irrigation units (modules). They in turn allocate concessions among their users (Hearne & Trava, 1997).

An important requirement of the 1992 National Water Law is that the CNA creates a Public Registry of Water Rights (REPDA) listing all concession holders, so as to control the assigned volumes of water (Hearne & Trava, 1997). Registration also guarantees concession holders legal certainty over their right to use the water (Kloezen, 1998).

Although trades of both groundwater and canal water between individual farmers have occurred before the 1992 Mexican National Water Law, bulk trades of water between WUAs started only after the law came into practice and still occur in a very limited number of districts (Kloezen, 1998). The new water law basically facilitated trade processes by devolving more authority to the users, by allowing trade and by dramatically reducing the size of the irrigation bureaucracy. In general, the water markets in Mexico involve trading of water for irrigation purposes, especially through the WUAs. However, transfers of irrigation water to municipal and/or industrial uses with compensation in the form of improved irrigation technology, are increasingly considered as an option since they can prove beneficial for the participants.

In this review, water trading which developed in the Mexically Valley and the Alto Rio Lerma Irrigation District is discussed.

2.4 Spain

Until now, water markets in Spain have evolved historically and have been existent only in a few places. For instance, local markets involving fully privatised water rights have been supported by favourable institutional arrangements and have played an important role for centuries in Valencia, mostly for groundwater. These local markets still exist due to the 75-year moratorium established in the 1985 Water Law for all wells registered before that date, so that the water pumped from them remains tradeable private property (Garrido, 1998*a*). Regarding the 1985 Water Law (29/1985), it imposed legal constraints on water trading, and aimed therein to place all water resources under control of the public domain. Thus, the only responsible for the management of water resources has been the public administration. Agents were converted into temporary rights-holders, and were impeded to change their water rights characteristics (Garrido, 1998*b*).

3. Return flow is the amount of water returned to streams and rivers.

Apart from Valencia, a water market has evolved in the region of Alicante and in Andalucia. The latter concerns a local arrangement for water trading between the city of Seville and one irrigation district established in 1994 with specific legal approval. The reason was that Seville's water storage proved insufficient to cover residential demand (Garrido, 1998a). Water markets have also functioned in the Canary Islands and the area of Tarragona in eastern Spain as discussed below in more detail.

As of 1999, a new water law (46/1999) has come into force which modifies the 1985 Water Law and aims to introduce more efficient use of water, incorporate market systems in water management and thus enable the establishment of formal water markets. The modified water law foresees, among others, that rights-holders can lease their water ("contracts of transfer of concessions"), subject to basin agency approval, to users of equal or higher seniority. The price for the transfer of water rights shall be fixed through mutual agreement between the two parties. Nevertheless, the basin authority can still intervene and set maximum prices in case of monopoly. Water rights may also be transferred through a water bank, "centros de intercambio", established and run by the basin authorities, which shall hold public offerings and set the price of the offered concessions (Garrido, 2000). The "contracts of transfer of concessions" and the water banks are expected to place professional challenges unknown so far in the water management sector in Spain. It has been actually debatable whether the contracts of transfer of water rights in Spain should be characterised as a water market, although they do include three features that approximate a market: voluntary participation of the parties, variable duration of the transfers, and freely negotiated prices (Garrido, 2000). Experience from the new water law on trading schemes is still to be obtained and it will depend largely on the use made of the new scheme of water allocation.

3. Case Studies

3.1 Tradeable water abstraction rights

Trading in water abstraction rights is still mainly related to agricultural use, although it appears in some cases that water is shifting from agricultural to higher valued uses, such as industrial and municipal. The latter are considered higher valued uses in terms of benefits gained in service or value of output product for the same amount of used water.

This section describes some recent developments in three trading schemes in the US, as well as an insight to the recent establishment of instream flow marketing for environmental protection in the western US. In Australia, intrastate and interstate water trade schemes in the territory of the Murray-Darling Basin Commission (MDBC) continue. In fact, the MDBC is planning to move to a permanent interstate trade scheme by progressively removing institutional and administrative barriers in the individual States.

Mexico has also gained more experience on water markets after the 1992 National Water Law. As far as Spain is concerned, two historically evolved water markets are described. Finally, a summary is provided of the plans to modify the water abstraction licensing systems in England and Wales, incorporating water trading. This scheme is still under consideration.

3.1.1 Recent developments in the United States

a) Northern Colorado Water Conservancy District, US

The Northern Colorado Water Conservancy District (NCWCD) is one of the most active water markets in Colorado and distributes the water from the Colorado Big Thompson Project (C-BT). It distributes water imported from the abundant water supplies of Colorado's west slope to the north-eastern Colorado. C-BT water shares (units) can be bought, sold or leased. Cities in particular have been major actors in the water market of this region (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex).

Initially, very few shares were traded within the scheme but as urban areas and industry in the region expanded, there was increased trading activity. Trading is done by posting offers on the NCWCD's bulletin board and through brokers. Two characteristics of the system greatly facilitate trading: 1) the extensive distribution system and 2) the special status of return flows from users. In Colorado, when transfers of imported water occur, there is no legal responsibility for return flows and the owner can use or exchange the return flows. This fact makes trading easier in comparison to other States where other water users establish rights on return flows (Howe, 1998). Another main reason for active trading is that water may be exchanged with a minimum of red-tape, and through relatively quick transaction process (Landry, 1998*b*).

Although the project was initially designed for agricultural use, the ownership of NCWCD shares has shifted, through the years, from agricultural to municipal and industrial uses. More specifically, while 82% of the CBT shares in 1962 were owned by agriculture, this percentage was lowered to 55% in 1992 with municipal and industrial uses gaining ground (Howe, 1998). The high quality and reliability of C-BT water makes it highly desirable for these purposes. Water has moved slowly from lower valued (primarily agricultural) to higher valued uses (municipal and industrial) (Howe, 1998). In general, Colorado has one of the most active and successful water markets with tens of thousands of acre feet[4] of water traded each year through private, voluntary transactions (Landry, 2000).

b) California Drought Water Bank, US

The California Water Bank has been one of the most well known drought response activities that the California Department of Water Resources (DWR) has undertaken. It was established for the first time in 1991 and worked quite efficiently for public and private users. The DWR is basically the only buyer, who buys amounts of water (m^3) from irrigation districts with a surplus and sells it to those with a deficit. In 1992, the US Congress passed the Central Valley Project Improvement Act, which allowed water contractors to sell water outside their districts (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex).

The Bank operated again in 1992 and 1994 (DWR, 2000). The Department began organising a 1995 bank anticipating another dry year, but this option was finally not exercised due to unexpected wet conditions. Selling price in 1991 was US\$0.14/m^3 and purchase price US\$0.10/m^3. In 1992 and 1994, prices were lower: selling price fell to US\$0.05/m^3, while purchase price was US\$0.04/m^3 (Archibald and Renwick, 1998). In 1991, 50% of the water purchased by the Department came from land fallowing, 30% from groundwater substitution, and the remainder from reservoir storage. In

4. An acre-foot is the volume of water necessary to cover one acre to a depth of one foot (approximately 1,233 m^3).

1992, however, almost 80% of bank purchases came from groundwater substitution, 20% from reservoir storage and no water from land fallowing, which (combined with improved water volumes) reduced the water purchase and sale price. Land fallowing contracts were considered more expensive than other sources, due to increased third-party impacts (Archibald and Renwick, 1998).

In general, the 1991 and 1992 drought water banks were able to meet the critical needs of all participants (Archibald and Renwick, 1998). Its success was partly based on the complete control of the whole process by an efficient and respected State Agency, and partly on the fact that sellers were supported by the enacted legislation (Garrido, 1998b). Water users in regions of bank sales, however, expressed concerns about third-party impacts of the land fallowing and groundwater substitution associated with the water bank of 1991 and 1992. This prompted the DWR to carry out groundwater monitoring programmes, and to fund external reviews of the 1991 and 1992 Banks. The reviews included economic evaluation of third-party impacts, as well as a programmatic environmental impact report in 1993 which covered operation of potential drought banks over the next 3 to 5 years (DWR, 2000). It is interesting to note that future banks are likely to differ from the previous banks, since almost 30% of California's counties have enacted local management ordinances for groundwater, which restrict or control groundwater export from a county. This shall affect future drought water banks by preventing them from implementing transfers involving groundwater (DWR, 2000).

In 1994, the California DWR also introduced an option market to add some security and flexibility to a potential 1995 water market. In short, the mechanism consisted in offering the irrigators the possibility to purchase options to be exercised in the event of water scarcity. Irrigators who offered their water supplies to the option market received financial compensation in the form of a deposit. If the water was finally needed, the sellers would receive the negotiated price for the exercise of the option. The improved hydrologic conditions, however, resulted in the option markets not being used and the irrigators made use of their water like in a normal year. The advantage of the option market is, nevertheless, that it allows adequate time for more elastic supply of water to the bank and a price structure that varies as the state of water supplies becomes better known by April, before the dry season begins (Howitt, 1998).

c) Water trading of surface water and groundwater in Texas, US

Concerning the market of surface water in the State of Texas, this has mainly been active for agricultural use, followed by mining and municipal use, along the Rio Grande (reviewed by Kraemer & Banholzer, 1999. For key information of this previous review, see Annex). The water market in the Lower Rio Grande Valley has emerged over the last three decades and is the State's most active water market. In the period 1996-2000, the volume of water traded ranged from 2,000 to 25,000 acre feet (Landry, 2000). Reasons for its success have been the well-defined water rights and the fact that water can be traded without holding public hearings as in other parts of Texas.

Regarding groundwater in Texas, much of the legal evolution concerning transferability has not been completed yet (Griffin, 1998). In Texas, groundwater follows the principle of absolute ownership, meaning that landowners may extract groundwater from beneath their land as long as their use is not wasteful. Therefore, the landowner does not hold title to specific units of groundwater beneath his land, i.e. a fixed groundwater quantity. Thus, the rough method to transfer groundwater has been to transfer the land (Griffin, 1998), or to pump the water and then sell it, in case the site of withdrawal is close to the buyer. In connection to this, there have been recent discussions regarding (ground)water property rights in Texas. Apparently, the purchase of land in western Texas in order to obtain water rights is the goal of companies who aim at gaining the property rights to the groundwater aquifers and then sell the water for urban consumption (The Economist, 2001, 26.05). This plan, however, meets with protests from environmentalists who fear the over-pumping of the aquifers. Texas' strong tradition of property rights does not enable the Groundwater Conservation

Districts to control such actions. Lately, it is considered to pass a bill that would establish more Groundwater Conservation Districts and enable them to charge a fee of at least 2,5 cents for every 1,000 gallons water being pumped in order to study the effects of pumping and to set up replenishment projects (The Economist, 2001, 26.05).

In addition to the surface water and groundwater markets discussed above, 1993 Texas legislation established a complementary marketing institution known as the Texas Water Bank (Griffin, 1998), which functions as a broker (Laundry, 2000). However, a range of regulations and bureaucratic uncertainty have contributed to a reduced interest in the water bank (Landry, 2000). In fact, only few deposits have been made and so far no transactions have occurred (Landry, 2000). Even though the Bank has been idle, however, this institution may gain importance in future years. It also serves as evidence of political commitment to achieve economic efficiency in water allocation (Griffin, 1998). The need for better institutions has been exposed, as elsewhere, under the pressure of growing water scarcity. After a drought, the 1997 Texas Legislature passed legislation destined to provide new opportunities for water marketing and regional planning. Unfortunately, as the legislation is implemented, the focus still appears to be on centralised planning and little attention is paid to water markets (Landry, 2000).

d) Instream flow marketing for environmental protection, western United States

Inadequate flow is one of the main reasons for not complying with water quality standards according to the Clean Water Act. Low flows are also challenging in the context of recovery efforts for endangered fish species. Therefore, the region of western US has recently placed more attention on increasing stream flows to fulfil flow management requirements of aquatic ecosystems and protect wildlife habitats (Landry, 1998). Actually, pressure to consider the value of free-flowing water and establish instream flow rights increased dramatically since the 1960s and 70s. In response to the increasing pressure in the western US, several legislative and policy changes have occurred to ensure market transfers as an option for protecting instream flows. The States of the Pacific Northwest region, including California, Idaho, Montana, Washington and Oregon, have all allowed public agencies and private individuals to acquire water rights for instream flows following slightly different approaches. In the Rocky Mountain region, it is required that instream flow rights be held by the State, while in the south-west region, States have continually rejected legislation on water markets for environmental protection. However, such markets have actually evolved in Arizona, New Mexico, and Nevada through court decisions (Landry, 1998).

Water sales for instream flows were reported in nine of eleven western States between 1990 and 1997 with estimated spending of US$ 61 million on leases and purchases of water (Landry, 1998). Since 1990, more than 2 million acre-feet of water have been leased, 130,000 acre-feet purchased and 245,000 acre-feet donated. Markets for instream flows have been using existing structures for operation and negotiations such as the Idaho water bank, but mostly federal agencies have been responsible for managing the acquisition schemes, such as the Nature Conservancy which served as a broker by negotiating certain purchase agreements. Most of the traded water was used to augment flows on major rivers in the western US. However, the results and cost-effectiveness of this approach are not certain yet in the context of actual ecological benefit. The federal government, mainly through the Bureau of Reclamation, has been responsible for 70% of the water quantity acquired in this period. For instance, the Bureau of Reclamation initiated in 1991 an acquisition programme in Idaho in the context of the salmon recovery efforts in the Snake River (Landry, 2000). Other actors in the water market have been State agency acquisitions as well as private environmental organisations, such as the Oregon Water Trust, the Environmental Defence Fund and the Nature Conservancy. Annual leases have been the most common way to obtain instream flows, while donations of small water amounts have recently become an important method of acquisition by private organisations. The average purchase price for instream water rights was approximately US$400/acre-foot. The efforts of

private organisations have actually proven more successful than of State and federal governments. Private organisations focus their restoration efforts on tributary streams, where small amounts of water can have significant positive effects to the aquatic ecosystem, rather than on large rivers (Landry, 2000).

3.1.2 Recent developments in Australia

a) Interstate water trading: Murray Darling Basin Commission, Australia

The Murray Darling Basin Commission (MDBC) was established in 1988, to operate and distribute the water assets of the States of New South Wales (NSW), South Australia, Queensland and Victoria. Although opportunities for interstate water trade existed, by 1997 this option was not available to irrigators (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex). Actually, water trading between States was a rare event and had only taken place on a temporary short-term basis (Pigram, 1999).

However, after the introduction of a cap on water diversions from the rivers for consumptive uses within the basin, water trading is increasingly seen as one of the main mechanisms available to achieve the desired improvements in efficiency. To facilitate water trade, the direct link between water and land is in the process of being broken in each State and increasingly water entitlements can be traded independently of land (Murray Darling Basin Commission, 2001).

Since 1997, the MDBC is conducting a Pilot Interstate Trading Project, which mainly aims to assist the progressive introduction of permanent interstate water trade (Cummins & Associates, 2000) and expand the number of traders able to participate in the water trading market (Young et al., 2000). The Pilot Interstate Water Trading project is limited to the permanent transfer across State borders of high security water entitlements held by private diverters in the Mallee region of the Murray Valley. High security water entitlements, which are the object of trade in this case, are fixed yearly volumes which do not vary from year to year and are available in all but the worst drought years (Young et al., 2000). Such entitlements are typical for town water supply and irrigation of permanently established crops, such as orchards and vines.

States in the basin lack so far a simple, transparent and quick water registration system (Cummins & Associates, 2000). Therefore, improving the administrative system is considered a number one priority. It is also recognised that intrastate trading problems first needed to be sorted out before a solid interstate trade comes in place (Cummins & Associates, 2000). For example, inter-valley trading is not yet possible between Murrumbidgee and Murray systems in New South Wales. In South Australia, the penalties for exceeding the licensed water allocation are often less than the price of temporary water transfers, and thus the incentive is to exceed allocation rather than trade water (Cummins & Associates, 2000).

Within the pilot trading, an interstate water transfer takes place between the seller in one state and the purchaser in another state, while it requires the approval of the water licensing authorities in both states. With the transfer of water from one state to another, there will be adjustments made to the state allocations under the Murray Darling Basin Agreement by the MDBC at the end of each financial year. The Commission will adjust state water caps in accordance with the trade, to ensure there is no overall increase in the diversion of water within the Basin. States are required to annually report on the transfers, including volume of water and the impacts (social, economic and environmental) of trade.

The pilot interstate project had allowed for 51 trades (a total of 9.8 gigalitres) by 2000 across the borders of South Australia, Victoria and NSW. Most trades involved a broker acting for the seller and a separate broker acting for the buyer. Collectively these trades are worth over A$9.9 million, while more than 90% of the water has moved to South Australia (Young *et al*, 2000). According to the project results, interstate trading is increasing the value and economic efficiency of water use in the basin. During the first two years, virtually all water has gone to higher value uses. The current system, nevertheless, has been characterised rather complex from the administrative point of view, since trade documents may take long to process. Interstate trading can also be expected to have a negative impact on river salinity in South Australia (Young *et al*, 2000).

Major issues to be dealt with in the context of the pilot project are: the specification of transparent property rights with State jurisdictions; the establishment of policy solutions to river salinity impacts from water trading; the consideration of administrative barriers; and rigour establishment in States' water licensing systems (Young *et al*, 2000).

b) Water trading in the south-east of the State of South Australia

Formal water markets in the State of South Australia were first implemented in 1983. The water markets followed an embargo on new water allocations in 1969 (Bjornlund & McKay, 2001). Proponents of efficient value irrigation projects needed water, which they could not have due to the embargo. Permanent and temporary trade only for private diverters was therefore introduced to accommodate these demands. In 1995, trade between private diverters and government irrigation areas was made possible by the introduction of the 1994 Irrigation Act, and the issuing of licences to irrigation areas under the 1990 Water Resources Act.

In the south-east of South Australia, number of trades increased significantly in 1998 (57 transfers in total) compared to 3 transfers that occurred in 1993 (Young & MacDonald, 2000). Under current rules, any trade requires a hydrogeological assessment held within the Department for Water Resources, in order to give permission for the proposed trade. Apparently, one of the main issues restraining water markets in the region is the way that water allocations are defined, allocated and registered (Young & MacDonald, 2000). Until now, a small number of volumetric water rights existed. In the near future, water rights will also be defined by water licences with a volumetric taking allocation. Permanent and temporary trading has the potential to deliver economic, environmental and social improvement in this area where water resources are not yet over-allocated and interest in investing in water- and land- related activities exists. It has been proposed that water allocations should be separated from access rights. A simple accounting system to track the status of each licensee's water should also be set up, and dealing and settlement procedures should be further facilitated (Young & MacDonald, 2000).

c) Irrigation trading: New South Wales, Australia

A system of mainly temporary as well as permanent transfers of water entitlements exists in New South Wales (NSW). In the Murray Region, water exchange was initially restricted to members of Murray Irrigation Limited which manages a large part of water resources in the area. The first experiences with trading in 1997 resulted only in a small amount of water being traded and showed the volatile nature of temporary water markets as well as the inexperience of irrigators with water trading (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex).

However, in the wider area of New South Wales, over 10% of the total entitlement to consumptive users was traded in 1997-98, mainly through temporary transfers (NSW DLWC, 1999).

The price recorded in these trades is highly sensitive to both fluctuations in the weather and the level of announced allocations. Trade within bulk licensees, such as Murrumbidgee Irrigation, accounts for almost one-third. Water trade within and across regions concentrates mainly in the southern region of NSW (Murray, Murrumbidgee and Lower Darling), where irrigation activity is higher. Trading occurs through formal exchanges, for example by the Southern Riverina Irrigators District Council which is the representative body of Murray Irrigation Limited, and informal agreements. The majority of trade takes place directly between water users, often based on landholder-to-landholder handshakes. Temporary and permanent trades in NSW, in the 1997/87 season, are estimated to have increased the value of irrigated agriculture by around A$30 and A$35 million respectively, compared with the situation of no trade. However, the magnitude of the financial benefits gained from trading is still not widely appreciated (NSW DLWC, 1999).

Water trading is still heavily impeded in unregulated streams and groundwater systems. Just in mid 1998 trading on unregulated rivers was announced (NSW DLWC, 1999). Permanent trade in all valleys is substantially impeded, mainly due to uncertainty about the future of the entitlement, thus causing major opportunities for investment and economic benefits to be lost or delayed. An important impediment to the current trading system is the significant delay in getting the trades approved by the Department of Land and Water Conservation (DLWC).

In the context of the NSW water reform, a new trading framework has been proposed as of June 1999 (NSW DLWC, 1999). The new framework proposes to integrate better defined volumetric water entitlements and site-use rights, promote better third-party protection and reduction of impediments to trade, as well as more effective administrative arrangements, information disclosure, trading rules, and policy monitoring. To begin with, it is suggested that the Water Act and regulations need to be amended to separate land and water rights (NSW DLWC, 1999). At the moment, there are multiple forms of water rights in NSW, many of which can be converted to a standard volumetric entitlement.

3.1.3 Experiences from other countries

a) Mexically Valley, Mexico

The lower Colorado River irrigates the Rio Colorado Irrigation District in the Mexically Valley. This is an arid area, and it borders with the city of Mexicali (Hearne & Trava, 1997). The district is irrigated by a mix of surface and groundwater. The irrigators include 7,067 ejiditarios (holders of communal land) and 7,627 private owners of land. The Irrigation District was managed by 22 modules (irrigation units managed by WUAs) from 1993 to 1995. As of 1996, however, water fees (water services fees, annual rehabilitation fee, soil use fee for private wells) were not completely 'negotiated' with the irrigators, according to the latest data collection (Hearne & Trava, 1997).

There is a relatively well developed market for water-use concessions in the Mexically Valley, with water-use rights being sold and leased without the accompanying land. In 1994-95, annual leases amounted to an estimated 857.5 million m^3 of water transferred. In addition to irrigation water, the district also delivers 100 million m^3 of water annually to the bordering cities in the States of Sonora and Baja California. This water, however, is not part of the irrigation district's allocation from the river, but comes from deep wells. The future growth of the city of Tijuana and other cities provides Mexico with a strong motivation to initiate water transfers from agricultural to urban uses. Transfers of irrigation water to municipal and/or industrial uses with compensation in the form of improved irrigation technology, for instance, can prove beneficial (Hearne & Trava, 1997).

b) WUAs in the Alto Rio Lerma Irrigation District (ARLID), Mexico

In the Alto Rio Lerma irrigation district (ARLID) in central Mexico, water trading has taken place between WUAs (Kloezen, 1998), which have been created after the provisions of the 1992 National Water Law. ARLID is situated in the State of Guanajato and has about 24,000 water users (55% ejiditarios and 45% small private owners). The district is subdivided in 11 modules, each one managed by an individual WUA. They all have concessions to water and infrastructure for a period of 20 years. WUAs, however, are in general poorly informed and often confused about their rights and concessions. Until 1997, although concessions had been allocated to the WUAs, none of them had officially been recorded in the REPDA register (Kloezen, 1998). Each WUA makes its own annual irrigation plan to distribute the water among its users. In 1992, a hydraulic committee was founded to determine the operational rules and targets for the district and monitor actual water distribution. The committee, where WUAs meet with CNA representatives and local State officials, additionally plays an important role in setting the quantity and price of the traded water. In ARLID, no trading of whole concessions has occurred but only of concession fractions and for a period shorter than the duration of the concession. The first trade between WUAs occurred in the 1995 summer season, and prices were entirely negotiated between the WUAs involved in the transaction. In 1996, because of rain, only one WUA bought water for US$ 2.00/1000 m^3, which was twice as much what the same WUA had paid in 1995. This price reflected better the operation and maintenance costs of the water transport to the users (Kloezen, 1998).

In 1997, the Limited Responsibility Society (LRS) was founded by the 11 WUAs. It took over the responsibilities for main system management from CNA. In that year, unexpected heavy rains had given additional amounts of water to the district, and there were intense discussions on how to allocate it. It was decided that WUAs who had saved water and had already met the requirements of the irrigation plan could sell some of the extra water to WUAs who had failed to follow the irrigation restrictions and needed more water (Kloezen, 1998). Three trades took place. In a meeting of the hydraulic committee, it was decided that WUAs that sell water would receive the price that farmers of the buyer-WUA normally pay for water allocation.

In all, the magnitude of water trading between the WUAs has been relatively small, although in the summer, traded water is approximately 30% of total water use. In some cases, WUAs use their entire yearly concession by the end of the winter irrigation season and have no water left. In this case, WUAs mainly buy additional water so that they do not jeopardise their credibility to the farmers, who have already paid for water. The price paid for traded water is considered less important than the social and political arguments that WUAs should maintain solidarity in order to effectively negotiate with CNA and public and private agencies. For WUAs, it is politically important to occasionally help out other WUAs, since they all form part of the Limited Responsibility Society and the hydraulic committee, which negotiates the yearly concessions with CNA (Kloezen, 1998).

c) Canary Islands, Spain

In the autonomous community of the dry Canary Islands, a historically grown, not very transparent water market has evolved. On the Islands, where the national 1985 Water Law did not apply, water resources have been exclusively in the hands of the private sector and completely detached from land property (Garrido, 1998b). Water actually belongs to whoever extracts it (Tremolet, 2001). Market transactions are common, but they are often not reported or registered by any public agency (Garrido, 1998a). Meanwhile, water on the islands is very scarce, which is aggravated by intensive agriculture and the increasing demands of the tourism industry. The above pressures have encouraged efforts to invest in infrastructure in order to capture deep groundwater (Tremolet, 2001), which has resulted in complete exhaustion of natural springs. The industry of water trade is attracting many investors, and water fetch prices are often over US$ 0.50-0.60 per m^3 (Garrido, 1998b). The private investors created numerous water communities, a specific form of limited company with shares, whose ownership gives rights to a percentage of water flow and responsibility to finance part of the infrastructure (Tremolet, 2001). Water can be sold on an annual or temporary basis. Buyers are farmers, hotel owners or municipal water companies (Tremolet, 2001). Water and shares are sold by specialised intermediaries.

Water mining on the Canary islands has been subject to a laissez-faire policy, leading to unsustainable exploitation of aquifers, monopolistic behaviour and a lack of transparency and information regarding the transaction of water rights (Garrido, 1998b). A law was passed in 1990 to impose more rigid conditions on hydrological constrains and water rights and also give greater role to water public institutions. This aimed to return water resources and bodies to the public trust following the Spanish constitutional mandate of the 1985 Water Law, and convert mining firms to holders of rights conceded by the Islands' water authorities (Garrido, 1998b). The 1990 Water Law maintained however a private status for all existing infrastructure for another 75 years, but all waters extracted through new infrastructure are regulated via a concession system and through tighter governmental control (Tremolet, 2001). According to the 1990 Water Law, supervising agencies were created at the island level, which are responsible for completing water management plans. The private sector, nevertheless, is now in need of modernising its activities. Characteristically, it has gradually lost confidence in the water market since the mid-1980s, since it views publicly run infrastructure and transportation networks as unfair competition (Tremolet, 2001).

d) Siurana-Riudecanyes irrigation district, Spain

The water management scheme of the Siurana-Riudecanyes irrigation area in the Camp de Tarragona has been characterised by the establishment of a historically grown water market. The irrigation area was granted its initial water concession in 1904 under the condition that two-thirds of the water would be used for irrigation and one-third for urban water supply, especially for the city of Reus. Meanwhile, the Irrigation Subscribers Association of the Riudecanyes Dam was created as a public corporation of the community of irrigators and local corporations, responsible for the management and the costs of construction and maintenance of the dam and the system infrastructure (Tarrech *et al*, 1999).

Water has been allocated through titles and public offering. Water titles in the area have always been tradeable among the members of the Association, which include both farmers and municipalities. The market is limited to municipalities and landholders of the region because the ownership of land within the irrigation area is a requirement for the ownership of a water title. Water rights include irrigation water rights, and urban supply water rights, which the city of Reus concentrates into a large block. The Association has often transferred "extra" water to urban uses for a negotiated price when scarcity made it necessary.

The maximum prices paid were US$ 0.52 per m^3 for yearly leases and US$ 9.5 per m^3 for permanent transfers, and occurred in the mid 1980s mainly due to the scarcity of water in the province and high water demand from agricultural products (Tarrech et al, 1999).

Concerning the administration of the water market, an official exchange administered by the Association was formed in 1982. The intermediary role of the Association is not based on any legal regulation, but rather on the confidence of the system participants. The Association has also been responsible for the direct management of all water resources in the area.

The water market has contributed to effective water allocation, since it has provided mobility of scarce water in an area characterised by small landholders, high crop diversity and changing urban supply needs. The success of the Association in operating the system lies in the small size of the area, the active role of its members, and its ability to adapt to the specific needs of the region. This management system is apparently quite efficient since the maintenance costs of the Riudecanyes dam, managed by the Association, are one-tenth of the costs for the Siurana dam, which lies in the same irrigation district but is managed by the regional water authority (Tarrech *et al*, 1999). In this scheme, the users actually participate in the administration of water, and their expertise in local issues effectively influences the development of the water market. The city of Reus has also been involved in funding and managing the system, and most importantly, it has fulfilled its water needs by participating in the water market (Tarrech et al, 1999).

3.1.4 New scheme in the United Kingdom

The current legislation in England and Wales already allows abstractors to transfer licences and to benefit financially from the transfer in a number of ways, such as through sale, rent or lease of the land on which abstraction takes place (DEFRA, 2000*a*). Nevertheless, to date little use has been made of these mechanisms across England and Wales.

According to a research report on economic instruments for water abstraction prepared for DEFRA (DEFRA, 2000*a*), a scheme has been established in one river catchment of the Anglian Region of the Environment Agency to allow the seasonal transfer of water between different areas of land for agricultural purposes. Under the scheme, water transfers are allowed through the drain and river system within various Internal Drainage Boards, enabling large growers to take their licence or a third party's licence and apply the water to another piece of land. It also allows small landowners to rent small land parcels without an abstraction licence and then move water from their own reservoir to these land parcels. During the three years of running as a pilot scheme, licence holders had to notify the Environment Agency of the water volume transferred and the fields on which the water was to be used. After a 3-year pilot phase, the scheme ran on a self-enforcement basis, with self-metering and metering by the Environment Agency. Only a few transfers have taken place, but it has provided spray irrigators with assurance that they can meet their contractual requirements.

However, the same research report (DEFRA, 2000*a*) concluded that in other cases of water transfer the parties involved were frustrated because of the long time it takes for the Environment Agency to agree and complete the process. Other reasons identified as restricting transfers of water are site–specific restrictions and conditions of use, regarding for instance the land on which the water is to be used, prior agreement of the value of the authorisation and the details of the transfer contract**.**

In 1998, a consultation process was initiated by the British Government, on whether to facilitate and promote more widely abstraction licence trading in England and Wales, while ensuring full protection of the water environment. The consultation paper 'The Review of the Water Abstraction Licensing System in England and Wales (DEFRA, 1998) raised the possibility of a trading system for abstraction licences which would operate together with both regulatory and voluntary

measures. The Government response to this consultation, 'Taking Water Responsibly' (DEFRA, 1999) proposed changes to the water abstraction licensing system that would simplify arrangements for license transfers. For example, applicants for abstraction licences would need only to demonstrate that they have rights of access to the intended abstraction point, rather than ownership or occupation. The Government also proposed giving the Environment Agency (EA) powers to require abstractors to enter into water resources management arrangements and to provide information helpful to the EA to secure the proper use of water. It proposed that the EA could compel one water company to seek a bulk supply from another, and require one water company to surrender abstraction licences in favour of another company. Financial and contractual arrangements would be determined by the Director General of Water Services (OFWAT), the economic regulator for the water supply and sewerage service industry. The changes proposed in 'Taking Water Responsibly' that required primary legislation were included in the 'Water Bill – Consultation on Draft Legislation' (DEFRA, 2000*c*).

A further consultation paper 'Economic Instruments in relation to Water Abstraction' was published on the likely scope for and effectiveness of water abstraction license trading (DEFRA, 2000*b*). In principle, water trading should promote optimal water distribution between and within different sectors of use. On the issue of market creation, the research concluded that there is potential for trade in a number of catchments where there is unmet demand, with interest expressed for both long-term and short-term trades. It was concluded, however, that scope for short-term trade may be greater. Regarding allocation of rights, it was assumed it would continue to be based mainly on the current 'first past the post' system with the possibility of auctioning licences in some tightly defined circumstances. The research also indicated the importance of rapid approval times by the EA and of the existence of brokers. Transaction costs were considered unlikely to be so high as to prevent trading from taking place. However, where information on environmental impacts would be necessary, costs could be significantly high.

Following this consultation, the Government decided ['Tuning Water Taking' (DEFRA 2001)] that voluntary licence trading from any catchment will be further facilitated probably from 2002, limited only by the need to protect the environment by considering abstractions and the return of water to the environment. The EA will be the principal facilitator of abstraction licence trading continuing to exert regulatory control and will be responsible for limiting environmental impacts should companies reactivate currently unused licenses ("sleeper" licences) for trading. The EA will also establish a precursor licence trading web site on which details of all completed licence trades can be posted.

3.2 Lessons learned on water trading (abstraction)

Reviewing recent developments in existing and new water trading schemes reveals a growing concern about the environmental consequences of water trading, primarily in Australia and the United States. Concerns mainly relate to inadequate instream flows (leading to endangerment of wildlife habitats, certain fish species etc.). The acknowledgement of increasing instream flows as a "water use" into trading schemes can offer an interesting solution, as can be seen in the western United States and is reflected in the proposed improvements in Australia.

Due to the limited experience with groundwater trading, the relevant potential consequences for the environment have not yet been revealed. Although considerable differences in the design of water trading schemes exist, the danger of over-exploitation of groundwater seems a widespread difficulty. This especially holds for cases in which groundwater rights are linked to land ownership. Setting local groundwater abstraction limits is an option that needs to be assessed on a case-to-case basis in order to protect regional aquifers.

In relation to this, it is of importance to point out the danger of large corporations entering the market, buying block water rights and then depleting the water resources in an area. Barlow (1999) discusses the role of superior purchasing power of corporations (see Texas case) as well as large cities which can force up the cost of water enough to drive farmers, small towns and indigenous people out of the market.

A main problem in implementing water trading schemes is thus the linkage between the ownership of land and related rights to abstract (or impound) water. There appear to be two fundamentally different approaches to overcoming this obstacle:

- One is to separate land ownership and water rights, whilst maintaining private property rights to water. Subsequently, water management can be determined by private trading. Environmental regulation and meeting other (social) water management functions can be hampered where government agencies are weak.

- The other is to separate land and water rights, and to transfer water rights into public trust or even State ownership. Subsequently, water management is usually determined by regulation and (public) planning policies. Environmental and other functions not directly related to economic activities can thus be ensured more easily, but the efficiency of water allocation to economic activities can be lower.

In between these approaches, practice often leads to intermediary solutions, based on collective institutions such as associations, boards or committees. Water being a natural resource requiring management at the regional level – and neither at the global or the strictly local level – institutions for facilitating water trading often resemble and build upon traditional institutions for water management, such as water users associations, irrigation co-operatives and such like.

The new water law in Mexico points in this direction by supporting the creation of water users associations. A first assessment of this recent development would be positive, but it would be premature to make a sound judgement as empirical evidence of the new law's effects is still limited. Another example is the way in which Spain integrated water trading systems that had grown historically into the new policy framework under the water law of 1999. This example, and notably the case of the Canary Islands, also highlights the difficulties of transferring water entitlements to public control.

Evidently, there is growing interest in interstate trading in order to address transboundary water management issues, notably in Australia and the United States. The relevant developments of water management institutions and regimes need further investigation. Although an increased consideration of the river basin management concept in trading schemes can be considered more efficient than managing trade according to historically developed administrative boundaries, the increase in size and complexity of the markets poses the danger of increasing transaction costs and processing time for trade approvals.

The possibility of introducing options markets for water trading in certain institutional settings seems an interesting possibility for adding security and flexibility to a water market. The establishment of the option bank in California can be seen as an important step forward for reducing the impact of droughts at the price of only low transaction costs while third-party effects need to be further investigated (Howitt, 1998).

3.3 Tradeable water pollution rights

Experience to date with permits to pollute water resources is limited, but it appears that trading can be part of the answer to achieve better water quality (Faerth, 2000). Examples of trading systems in water pollution exist in relation to individual substances and parameters, since a much higher degree of complexity is introduced when attempting to consider the effects of a number of substances within a trading system.

In this section, some recent developments on three point-non-point nutrient trading programmes currently in place in the US are provided, as well as information on the Chesapeake Bay scheme and its fundamental principles on nutrient trading, which have been recently developed. Nutrient trading is increasingly seen as an option to improve water quality in the US, and in this context, the results of a theoretical analysis on the issue in three study areas are discussed. The case of selenium trade among irrigation districts in the US San Joaquin Valley is also briefly presented.

As far as the Commonwealth of Australia is concerned, this section provides some recent development on the salinity trading cases in the Murray-Darling Basin and the Hunter river, as well as on the nutrient trading within the South Creek Bubble Licence.

3.3.1 Recent developments in the United States

a) Nutrient Trading: Tar-Pamlico River (North Carolina)

The Tar Pamlico Basin was designated "nutrient sensitive water" and was given a basin-wide "bubble" (annual, collective nutrient loading cap for 14 point source dischargers of the Tar-Pamlico Basin Association) for nutrient pollution in 1989. The Association facilitates trade among member pollution dischargers, and non-member farmers. The Association was also required to make a monetary contribution into a non-point source (NPS) fund that is used to implement agricultural best management practice (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex).

Under Phase I (1991-1994), municipal sources were allowed to offset excess discharges with nutrient reduction credits obtained through contributions to the NPS fund. Phase I nutrient reductions were greater than the desired goal, due to the low-cost improvements at the municipal wastewater treatment facilities. It is worth noting that the estimated cost would be US$7 million to achieve the same level of wastewater treatment plant nutrient reductions that can be achieved with US$1 million by investing in non-point source pollution control (Great Lakes Trading Network, 2001).

Phase II of the programme will run until December 2004, and foresees a 30% reduction for nutrients. A main component of Phase II is wetland restoration and identification of areas of major non-point pollution in order to set action priorities. Two signatories to Phase I of the programme, the Environmental Defence Fund and the Pamlico-Tar River Foundation, did not endorse Phase II, because they were concerned about the programme's ability to address non-point pollution sources and the nutrient cap for point sources (EPA, 1996b). Under the Tar Pamlico Basin Nutrient Trading Programme, point source/point source trading has occurred under Phase I and continues under Phase II, allowing point sources to optimise the cost of achieving the nutrient cap established for the Association. To date, point/non-point source trading has also occurred in excess of US$750,000 (Great Lakes Trading Network, 2001).

Although an in-depth evaluation of the Tar-Pamlico trading scheme is so far missing in the literature, it is considered in overall a quite successful programme. Nevertheless, discussions on Phase

II have indicated potential problems of trading to deal with non-point pollution sources. It may be worth evaluating in more depth the success of the specific instrument of tradeable permits, by comparing the results of trading with the potential results (and costs) of alternative pollution reduction instruments in the Tar Pamlico Basin.

b) Nutrient Trading: Lake Dillon (Colorado)

Lake Dillon, a tourist attraction and a significant source of water supply for Denver, has been under significant pressure from phosphorus discharges. Four municipal treatment plants, sixteen small treatment plants, one industrial plant and numerous non-point sources discharge waste into the reservoir. Runoff from towns and ski areas is the main non-point source of phosphorus, along with selected inadequately managed septic systems (EPA, 1996*b*). After annual discharge rights for every point discharger were allocated, trade between point and non-point sources of phosphorus around Lake Dillon has been allowed since 1984 at a 2:1 ratio. Until 1988, no trading was approved by the basin management authority, since critical loads were not exceeded (reviewed by Kraemer and Banholzer, 1999, For key information on this previous review, see Annex).

Until the end of 1996, only a few trades had taken place between point and non-point sources. The Lake Dillon phosphorus trading programme has refocused at maintaining equitable non-point-non-point source trading and enforcement. New non-point sources must offset all of their discharges by using a trading ratio of 1:1 with existing non-point sources. The co-operative management approach that grew out of developing the option of the trading programme is considered by a number of stakeholders as the reason why Lake Dillon has succeeded in maintaining high water quality. When point-non-point source trading occurs, point source discharge permits include information on the record of the credit amount, specified construction requirements for non-point source control as well as monitoring, reporting requirements, and operation of non-point source best management practices (BMP) (EPA, 1996*b*). The lake Dillon trading programme is co-ordinated by the Summit County Water Quality Committee, which distributes phosphorus credits, identifies potential BMP projects, ensures monitoring is performed and non-point source pollution reduction programmes, such as covering of septic tanks, are implemented.

Trading has still been very slow due to limited population growth and a recession in the region. Moreover, the wastewater treatment plants have found cheaper means of controlling phosphorus than were previously envisioned. In the future, however, opportunities for further control at the treatment plants are thought to be limited, and population growth seems to be evident, leading to the conclusion that more trading activity is likely (NCEE, 2001).

c) Nutrient trading: Cherry Creek (Colorado)

The Cherry Creek reservoir is an important recreation area and water supply source. A total phosphorus standard was developed in 1984 for the reservoir, as well as a Total Maximum Daily Load (TMDL) (EPA, 1996*b*) to prevent eutrophication and maintain water quality standards established by the Colorado Water Quality Commission. The Cherry Creek Trading programme allows certain point source polluters to earn phosphorus reduction credits through the control of non-point source phosphorus discharges (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex). The TMDL requires urban non-point sources to reduce phosphorus loads by implementing best management practices. However, non-point sources, which account for approximately 80% of the basin's phosphorus load, have to reduce their loading by 50% on their own, and only reductions beyond these required non-point reductions can qualify for trading (Great Lakes Trading Network, 2001).

Initially, the Authority has the possibility to engage in two types of trade: trades of phosphorus reduction credits generated through authority water quality improvement projects and trades of credits generated through private projects. More specifically, the Authority has four completed non-point source water quality improvement projects that generate phosphorus reduction credits under the trading programme. Credits from Authority projects are placed in a Trade Pool for transfer to individual dischargers. The Authority also reviews similar privately constructed projects and assigns credits to the private party accordingly. All credits are quantified through direct water quality monitoring. Dischargers may purchase credits from the Trade Pool, if they fulfil certain requirements. They should namely demonstrate the requisite need for the increased phosphorus allocation, their wastewater treatment facility should operate and continue to operate so as to achieve expected phosphorus levels, and they should comply with the existent effluent limits. The Authority itself transfers credits to dischargers from the trade pool on a long term basis, but does not convey ownership of credits in such transfers (EPA, 1996b).

Development and credit use are required to be consistent with a basin plan established by the Cherry Creek Basin Water Quality Authority. The Cherry Creek trading programme is currently being revised to reflect baseline allocations under an updated TMDL (Great Lakes Trading Network, 2001). To date, there has been no need to trade at Cherry Creek since phosphorus effluent still remains below the allowed limits. When regional economic growth compels wastewater treatment facilities to achieve greater phosphorus reductions, the credits will be available (NCEE, 2001).

d) Selenium trade in San Joaquin Valley in California, US

In San Joaquin Valley, California, as in other areas of the US, the voluntary adoption of Best Management Practices (BMPs) to counteract water quality degradation resulting from non-point sources of pollution failed to achieve the desired pollution decrease (Young & Karkoski, 2000). Although the technology to control pollution was available, farmers were concerned about costs and the administrative difficulty of starting a compliance programme. The conclusion of a study by the Environmental Defence Fund in 1994 was that a combination of input fees for farmers and tradeable discharge permits for non-point sources of pollution among irrigation districts offered potential advantages over BMPs to control agricultural pollution in the valley. Since 1999, the proposed system has been in place and has reduced pollution discharges substantially (Young & Karkoski, 2000). A permit was issued by the local pollution control agency to a consortium of irrigation districts to enforce a regional cap on farm-level pollution discharges, consistent with a TMDL.

Initially, farmers requested permission to use an existing federal canal, the San Luis Drain, to convey their drainage directly downstream to a San Joaquin River tributary. In exchange, the US Bureau of Reclamation required that the discharge would meet federal law and policy. The above resulted in the Agreement for Use of the San Luis Drain in 1995, valid for 5 years, between the US Bureau of Reclamation and the consortium of irrigation districts, represented by a joint powers authority, the Grassland Area Farmers. The Use Agreement is *like* a point source discharge permit, specifying the amount of selenium[5] load that can be discharged by month and by year. Within this framework, irrigation districts and farmers begun to develop their own mechanisms to control drainage. The allowable selenium discharge was divided among the districts according to a formula, and monitoring is carried out by a Regional Drainage Co-ordinator hired by the consortium (Young & Karkoski, 2000).

The Grassland Area Farmers have developed a trading system that provides flexibility and further incentives to reduce discharges. In 1998, they adopted the "Rule Establishing a Tradeable

5. Selenium is a toxic pollutant, which accumulates where there is heavy industry or agricultural drainage containing pesticides.

Loads Programme for Water Year 1998". Any or all of an allocation could be traded within the group for money, services or other legal consideration as long as the trade was consistent with the regional allocation for each month and each year. Other parties are not allowed to enter the market, since only member of the Grassland Area Farmers can use the San Luis Drain. The 'Rule establishing the Tradeable Loads Programme for Water Year 1999' authorised the Grassland Area Farmers to impose discharge fines to a district that exceeded its allocation. The primary economic incentive that has been adopted within individual districts to comply with the allocated discharge is tiered water pricing (Young & Karkoski, 2000).

As a result of the programme, water quality has improved, and selenium discharges have been reduced by a third during the first two years. Regarding the trading programme itself, during its first year, one trade was executed. In the second year, eight trades were executed involving six of the eight programme participants. In addition, the tradeable allocation programme has functioned as a cost-sharing mechanism and provided flexibility to the districts to adjust their initial allocations. Transaction costs for the executed trades have been minimal since the monthly meetings among the districts to exchange monitoring information make it easy to contact potential buyers and sellers (Young & Karkoski, 2000).

3.3.2 Recent developments in Australia

a) Interstate salinity trading case: Murray-Darling Basin, Australia

Interstate salinity trading came into force in 1992 as part of the Murray-Darling Basin Salinity and Drainage Strategy, administered by the Murray-Darling Basin Commission, on behalf of the States of New South Wales, Victoria and South Australia. The interstate salinity trading is based on a system of salt credits and debits. The salt pollution rights are not freely traded by industries or individuals, but are exchanged between the governments of the participating states. Although credits are tradeable between the States, they are generally applied within each State to offset debits from drainage entering the river system (reviewed by Kraemer and Banholzer, 1999. For key information on this previous review, see Annex).

The Salinity and Drainage Strategy has been successful in achieving a net reduction of 57 EC (Electrical Conductivity) units in the lower river Murray. However, investigations throughout the 1990s showed that increasing salinity in the Basin is threatening the further success of the Strategy. Therefore, a new Basin Salinity Management Strategy 2001-2015 has been developed to ensure that further activities in the Murray-Darling Basin against salinity are successful. The new Strategy establishes a Basin-wide target, with Queensland also participating, for river salinity at a level of less than 800 EC units for 95% of the time over 15 years at Morgan, South Australia (downstream State). The end-of-valley target is in effect a 'cap' on salinity pollution. The effective date for the new arrangements was 1 January 2000 (Murray Darling Basin Ministerial Council, 2000).

The system of salinity credits continues, but now operates basin-wide. Each government will contribute to joint or individual works that will reduce the salinity of the shared rivers, thus earning salinity credits. Any work within a State that further reduces salinity in the shared rivers will attract additional credits for that State. All States will incur debits based on the basis of the estimated shortfall in protecting shared rivers and for specific actions such as drainage that increase salinity in the shared rivers. A review of the salinity debit and credit accounting system will be undertaken after 2015 (Murray Darling Basin Ministerial Council, 2000).

b) Salt Pollution trading case: Hunter River, NSW, Australia

The Hunter River Salinity Trading Scheme is Australia's first active emissions trading scheme, put in operation in 1995 by the Environmental Protection Agency of New South Wales (NSW EPA), and has proved very successful (NSW EPA, 2001*a*). Each discharger is allowed a specified percentage of the total allowable salt load, which is calculated in relation to conductivity levels. The scheme was initially limited to coal mines and the power generation industry of Pacific Power. Initial experience showed that conductivity levels remained within the target limits, with only a few trades occurring. A major advantage of the scheme is an extensive monitoring network which monitors each authorised point of discharge. Low trade was due to uncertainty about long-term needs, arrangements for longer term allocations (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex) and inexperience with the scheme (NSW EPA, 2001*b*). It is possible that the purely paper-based trading mechanism has inhibited the potential volume of trades. The NSW EPA has recently developed a 24-hour on-line credit exchange, to make trading faster and easier (NSW EPA, 2001*b*).

In general, the salinity target (900 EC unit level at Singleton monitoring point and 600 EC units at Denman) has not been exceeded as a result of participant's discharges since the scheme has been in operation. The only occasional exceedances were caused by saline diffuse run-off (NSW EPS, 2001*a*). Notably, exceedances decreased from 33% before the introduction of the scheme to 4% currently (NSW EPA, 2001*b*). The trading scheme operates during high flows. No discharge is allowed during low flows and unlimited discharges are allowed during flood flows. The Department of Land and Water Conservation estimates the total allowable salt discharge at high flows so that the river is below the salinity target.

Trading has allowed major industries such as coal mining and power generation, to discharge saline water on a managed basis. It has also reduced significant costs of water storage or treatment that would otherwise have been incurred by those industries under the previous discharge management system, which included a traditional licensing strategy requiring industries to minimise discharges and discharge a small volume of saline water to the river at all times (NSW EPA, 2001b).

In 1999 and 2000, the number of trades increased, with 31 trades occurring in 2000 (NSW EPA, 2001*b*). Trades between participants can be temporary or permanent. Due to its success, the trading scheme is expected to continue indefinitely. Moreover, the EPA has proposed to establish the scheme through a new specific legislation. The proposed *Protection of the Environment Operations (Hunter River Salinity Trading Scheme) Regulation* will permanently implement the existing pilot trading and place it into a firm legislative framework.

The proposed regulation aims to protect water quality while allowing mining in the area to expand. At present, at least 6 new mines are under consideration in the basin of Hunter River, which generate benefits to the community in terms of employment and an increased financial output. Before the trading scheme was introduced, it would not have been possible to consider such new development. As an indication of indirect benefits to the community, each new mine is likely to employ 300 people and generate an output worth $300 million per year. Under pre-scheme conditions, it is likely that the community opposition to new salt discharges would have stalled most of these projects (NSW EPA, 2001*b*).

The new Protection of the Environment Operations (Hunter River Salinity Trading Scheme) Regulation proposes the following changes (NSW EPA, 2001*b*):

- Create 1000 tradeable salinity credits, of 2 to 10 years, which will be allocated without charge to existing pilot scheme participants.

- The expiry of 20% of the credits every 2 years, and the reallocation of those credits by public auction, with each credit then valid for 10 years. Therefore, the mechanism for allocation,

holding and trading credits shall be altered, moving from administrative allocation of credits to initial allocation based on current holdings (grandfathering) followed by 2-yearly credit auctions. Auctions will ensure new industries can readily access credits.

- Create a new administrative role of "Services Co-ordinator", responsible for providing the river monitoring, modelling and River Register services.

- Create a new stakeholder committee called the *Hunter River Valley Salinity Trading Scheme Operations Committee*, which may take on the role of Services Co-ordinator.

The EPA believes significant benefits will occur from the proposed regulation such as increased certainty that the scheme will continue to function, which provides investors with a longer planning horizon. The proposed changes also aim to increase transparency and accountability with respect to funding and service quality, and avoid non-availability of credits to new market entrants or restriction on eligibility to hold credits (NSW EPA, 2001*b*).

c) Nutrient trading: South Creek Bubble Licence, Hawkesbury-Nepean River (NSW)

Three sewage treatment plants of Sydney Water Corporation (SWC) in the South Creek area of the Hawkesbury-Nepean River are the participants of a ”bubble” licensing scheme with the aim of obtaining improved environmental outcomes at lower cost. The owners of the individual sources within the bubble are permitted to adjust their discharges by trading parts of their nutrient discharge allocations, provided the aggregate limit is not exceeded. The Scheme commenced in 1996 and set nutrient reduction targets until 2004 for both phosphorus (83%) and nitrogen (50%). It is basically a small self-contained emissions trading scheme and it functions within a strong regulatory framework (reviewed by Kraemer and Banholzer, 1999. For key information of this previous review, see Annex).

The NSW EPA conducted a review of the Scheme’s first three years of operation (NSW EPA, 2001c). It concluded that Sydney Water Corporation has complied with the Bubble load limits, while significant reductions in nutrient discharges have been achieved. However, it is yet early to conclude on the environmental response to the discharge reductions, based on the available monitoring data. Both discharge monitoring of the individual treatment plants as well as ambient monitoring is carried out by SWC to measure the impact of nutrients from South Creek on the main reach of the river. Additionally, new scientific information on the impact of nutrients suggests that there may be a need for a further nitrogen reduction. Therefore, further discharge reduction goals should be reviewed in 2002.

The possibility of including non-point sources in the bubble is increasingly discussed and should be further explored (NSW EPA, 2001*c*). The bubble scheme could provide a strong basis for extending trading to incorporate diffuse sources, if further work could provide a basis to quantify the differing impacts of point versus non-point discharges. Point and non-point sources are not considered currently directly comparable, due to the dependence of non-point discharges on weather events. However, including non-point sources in the bubble could be particularly worthwhile if the costs of reducing diffuse discharges were significantly lower than for point sources, after taking into account appropriate trading ratios to reflect their lesser impact. Additionally, any effort to extend the bubble scheme to diffuse sources must recognise the complex array of other initiatives which aim to address water quality problems from diffuse sources. Such initiatives are stormwater management and several integrated catchment management processes (NSW EPA, 2001c).

Overall, the bubble licence scheme is considered successful, since it allows flexibility in capital infrastructure planning by allowing investment in one or two plants opposed to all three, as would occur under uniform concentration limits. Long–term cost savings are estimated to be A$45.6

million (or 37%) compared to requiring plants to meet uniform reduction individually (NSW EPA, 2001c).

3.3.3 New schemes under consideration & Theoretical trading cases

a) Nutrient Trading: Chesapeake Bay

In the early 1980's, research carried out by the US EPA revealed that low dissolved oxygen in the Chesapeake Bay due to nutrient enrichment was a major problem, and the estuary was in need of a collaborative restoration effort. Therefore, the US EPA, the States of Maryland, Virginia, Pennsylvania, the District of Columbia and the Chesapeake Bay Commission singed the 1987 Chesapeake Bay Agreement. According to the Agreement, a 40% reduction of nutrients, compared to 1985 levels, in the Bay is necessary to restore its health by the year 2000 (Wiedeman, 2001). New restoration commitments were adopted under the agreement Chesapeake Bay 2000, which aims to remove the Bay and its tidal waters from the list of impaired water bodies for nutrients by 2010. This would require nutrient reductions far beyond the 1987 40% goal. In the meantime, growth in nutrient load may be expected due to increases in sewage flows and polluted runoff from new development.

Trading to maintain the cap is considered a significant strategy for Chesapeake Bay. Some of the Bay jurisdictions actually began to explore trading on their own. Virginia's legislature enacted the Virginia Water Quality Improvement Act in 1998, which includes a clause requiring trading to be explored as a means of nutrient management (Wiedeman, 2001). In 1998, the Chesapeake Bay Programme formed the Nutrient Trading Negotiation Team, to explore trading in the Chesapeake Bay. The team had to examine the concept of trading in the Bay and develop nutrient trading guidelines to assist States in voluntarily developing State-specific nutrient trading programmes.

The Negotiation Team focused mainly on the 6 following elements, which are vital to the trading framework: the nutrient reduction goals, eligibility of credits to trade, trade administration, accountability, indicators for assessment of the scheme, and stakeholder involvement (Wiedeman, 2001). According to the fundamental principles on trading that the Team formulated, trading will be allowed only within each major Bay tributary among all signatory States to the 1987 Bay Agreement, as well as non-signatory States if they are consistent with the trading guidelines (Nutrient Trading Negotiation Team, 2001). The nutrient trading programme should also be consistent with the Chesapeake Bay Programme's reduction goals, i.e. 40% reduction. To achieve this, trading should be allowed only among "like" sources, which means trading between point and non-point sources is not allowed. The trading programme should set specific nutrient load allocations for each major Bay tributary, a baseline and a cap, as well as allowances for point and non-point sources. The final Nutrient Trading Guidelines of 2001 are available for use by States on a voluntary basis to design their own trading programmes (Wiedeman, 2001). It is considered that many point sources will be able to generate credits for trade, since there are 347 point source wastewater treatment plants in the Bay watershed. Each trade should result in net reduction in nutrient loading and also maintain the tributary nutrient cap. No local water quality impacts are allowed to result from trading. A source may receive credits for reductions in nutrients, through the operation of a facility or the implementation of a BMP (Nutrient Trading Negotiation Team, 2001).

As far as administration is concerned, each State should be responsible for programme oversight and day-to-day management (certification, registration, monitoring, evaluating). A central State co-ordinating office should be established in each State to deal with the administration of trades. Trades should also be governed by a State general regulation under the State's water quality law, and public participation prior to the execution of a trade should be promoted (Nutrient Trading Negotiation Team, 2001).

b) Theoretical analysis of nutrient trading in Minnesota, Michigan and Wisconsin

In the US States of Michigan, Wisconsin and Minnesota, several alternative policy approaches to counteract watershed phosphorus pollution were tested using a modelling framework which analyses their economic and environmental potential. Nutrient trading was examined as an option to achieve phosphorus reductions in three watersheds: Saginaw Bay in Michigan, the Rock River in Wisconsin and the Minnesota River Valley (Faeth, 2000). The three cases were chosen due to the water quality problems of the areas and because the responsible State agencies were interested in exploring trading as a policy alternative. In each of the three States, there has been already significant movement towards the use of trading. In Michigan and Wisconsin, pilot trading programmes are being pursued, and in Minnesota, *ad hoc* trades have occurred.

Concerning the trading background of the three States, Michigan has developed rules for a voluntary State-wide water quality trading programme, which aims to optimise the costs of improving water quality, facilitate Total Maximum Daily Load implementation, and provide economic incentives for non-point source reductions (Great Lakes Trading Network, 2001). Within this context, it conducted a water quality demonstration trading project on the Kalamazoo river, which helps the State and stakeholders in designing a State-wide programme. The fundamental goals of this project were to form partnerships, improve water quality, optimise costs, and provide greater flexibility for a sustained local economy. It was completed in June 2000, and it demonstrated how trading can occur serving as a basis for the State water quality trading programme (Great Lakes Trading Network, 2001). The Water Quality Trading Group in Michigan has drafted an extensive Water Quality Trading Rule to enable it to implement the demonstration programme (Faeth, 2000).

In Wisconsin, after a proposed rule to require all point sources in Rock River to adopt phosphorus controls, a group of point dischargers requested an extension from the Wisconsin Department of Natural Resources to try a nutrient trading programme in the basin. In 1996, a budget was approved to start four pilot programmes on nutrient trading including the Rock River, to also help develop a State-wide framework for trading.

In Minnesota, the Pollution Control Agency established a TDML in 1988, on the lower 25 miles of the Minnesota River. A preliminary study done for the EPA in 1987 identified the basin as a good candidate for a trading programme. In 1997, a trade was initiated which was tied to a new discharge permit for a malting company. This allowed the company to build its own wastewater treatment facility on a tightly regulated stretch of the river in exchange for reducing non-point source discharges upstream from the plant. The reduction of non-point source discharges would be achieved by supporting soil erosion controls, livestock exclusion from waterways, critical area set asides, and wetland systems for nutrient retention. The company receives credits from the above practices which can be applied against the load of its new facility.

Overall, the results of the theoretical analysis in these three watersheds showed great potential for cost savings by applying market-based strategies when compared to water quality regulations and agricultural subsidy approaches. Trading appeared to be dramatically less expensive than conventional approaches while achieving comparable benefits (Faerth, 2000). More specifically, because there is a large differential between remediation costs for conventional approaches and programmes that involve trading, trading has potential in all three considered watersheds.

The proposed strategy is a regulatory mandate on point sources coupled with a flexible trading programme which also directly involves agriculture. It was recognised, however, that although trading has economic potential, there are some uncertainties associated with it (Faeth, 2000). First of all, non-point sources do not produce regular flows of pollution across seasons but rather depend on weather events. Therefore, the reduction from a non-point source may not be equivalent to that from a point source. Liability must also be carefully considered. Namely, water quality

programmes involving trading must provide a legal remedy when someone sells or applies a credit that has no environmental value, just like the regulator can do when a point source does not meet a legal requirement. Due to the above and other issues such as high transaction costs, trading should occur within a regulatory programme where methods are standardised and appropriate review can be cost-effective (Faerth, 2000).

3.4 Lessons learned on effluent trading (pollution control)

Environmental protection measures have succeeded in reducing point source pollution in most OECD Member countries and non-point pollution, coming from diffuse sources and especially runoff from agricultural or residential areas, urban runoff, even bacteria and nutrients from livestock and faulty septic systems, is gaining in importance. As a result, trades between point and non-point pollution would now offer themselves as potential policy measures, especially as reducing non-point pollution is often more cost effective in meeting environmental quality objectives than additional investment in point-source control (see case study on North Carolina). However, experience with this type of trades is limited and the area needs further investigation (James, 1997).

The difficulties in controlling non-point pollution through trade stem from the need to consider a complex array of other measures of non-point pollution control or reduction in the design of a scheme. Problems arise through the uncertainties in estimating and monitoring non-point source loads as well as due to lack of direct comparability of point and non-point pollution, since non-point discharges may depend on weather events, for instance.

4. Potential further applications of water-based tradeable permits

The application of tradeable permits in water resources management deserves to be further investigated especially in countries, where water imbalances (seasonal and/or regional) exist and water availability is or may become an issue of national security. In such cases of scarce, unequally distributed water resources, tradeable permits can be more widely used as a means to achieve fair water allocation.

For instance, a region of potential application of water trade is the Lerma Chapala basin in central Mexico, which is a populated, urban-rural river basin affected by water pollution and scarcity. Five States of capital importance in Mexico's development strategy have a share in the catchment, where 16 million people live and 35% of the industrial GNP is produced (Mestre, 1997). The Lerma River runs through the middle of Mexico, an area of high economic activity. Irrigated agriculture is the main water user, representing 78% of all water presently used. There is however competition between industrial/domestic water supply and agriculture for land and water on a local level (Mestre, 1997). Growing industry and agriculture have led to the overuse of both water from the river and from aquifers, especially since the mid 1980s. Because of untreated waste from both industry and agriculture the river has become very polluted, threatening the water supply of Mexico City and harming downstream stretches, especially Lake Chapalla at the end of the catchment.

Since 1993, a River Basin Council has been established to promote an integrated management framework for the basin's water resources seeking, among others, ways to finance and achieve financial self-sufficiency in the river basin water sector (Mestre, 1997). In order to achieve water use efficiency, water tariffs for irrigated agriculture have been raised close to real costs. Water users have a General Regional Assembly and work committees of a very complex structure, arranged by water use and state. Since 1991, a surface water distribution agreement of clear mathematical rules for water distribution and reservoir operation in accordance with users, has been in force. For aquifers, specific regulation agreements are technically and legally under discussion (Mestre, 1997). In

this search for solutions, water trading could be investigated as a means to contribute to successful integrated water management in Lerma Chapala, by providing incentives to improve water use efficiency and achieve greater water availability, with reasonable quality.

To date, there has been some experience acquired with schemes of interstate trade, especially within the federations of the US and Australia. In the context of transboundary water trade, a significant further development in the use of water abstraction and effluent tradeable permits would be investigating their applicability at the international level. World-wide, at least 214 rivers flow through two or more countries, but no enforceable law governs the allocation and use of international waters (Worldwatch Institute, 1996). Especially in areas of water scarcity, as in the Middle East, transboundary trading schemes, involving the trade of water within an international basin across political borders, can be an attractive means to ameliorate existing and expected water scarcity. Such schemes can be voluntary and involve two or more countries and regions which share a common river basin in terms of hydrology.

In the Jordan river basin, which is shared among Israel, Jordan, Lebanon and Syria, it has been shown in theory that an international trading scheme could increase the welfare of the countries concerned (Zeitouni et al, 1994). Efficient water allocation could be achieved and parties might benefit from auctions of water claims. In the Middle East, in general, there has been some experimentation with market mechanisms at a theoretical level in order to reallocate water resources and solve the Israeli-Palestinian water dispute. A practical application of water trade may be worthwhile, although it would require more work and investigations on the institutional context of such a scheme.

As a general rule, political conflicts need to be resolved first, in order to resolve water disputes. Moreover, in such cases of potential international water trading, it is important to first calculate reasonable entitlements and determine water rights, which satisfy all parties. For instance, an interim agreement was signed in 1995 recognising that the Palestinians have rights to West Bank groundwater, and agreed on an interim water allotment to them (Worldwatch Institute, 1996). However, the larger issue of the permanent allocation of water rights is not yet resolved.

In the case of the Mexico-US conflict over the water of the Rio Grande, an institutional context for water allocation is already existent. More specifically, a treaty between the United States of America and Mexico relating to the utilisation of the waters of the Colorado and Tijuana Rivers and of the Rio Grande was signed in 1944. In the meantime, the transboundary Rio Grande has nearly dried up near its estuary and has been polluted in many parts due to the disposal of industrial effluents, agricultural run-off and urban sewage. Agriculture is the largest regional water user, though growing population and rapid industrialisation, especially in northern Mexico, are bound to place more pressure on the limited water resources in the future. A recent long drought and rising water demands have resulted in the escalation of the water conflict between the two countries (Süddeutsche Zeitung, 2001). In such cases of acute water scarcity, the signing of treaties may not be enough to satisfy the water demands of all parties, and the development of an international trading scheme could serve the purposes of a more efficient water allocation.

Apart from international trade of water abstractions and allotments, transboundary trade of effluents could also be of interest to many countries. It may be worth investigating the setting up of international 'bubbles' for water bodies strongly affected by transboundary pollution, as in the case of the Danube river in Europe. The Danube is one of the longest rivers in the world and numerous nations lie within its catchment. The International Commission for the Protection of the Danube has set up a pollution reduction programme and has been active in identifying ways to combat transboundary pollution and finance pollution control schemes in the catchment. Effluent trading could be investigated as one of the possible approaches against water pollution control in the catchment. Actually, tradeable pollution discharge permit systems have been identified as an

innovative approach with the potential of being financed by the Global Environment Facility (GEF) in the Danube River Basin (Kreditanstalt für Wiederaufbau, 1999).

5. Resistance to water trading

The UN Water Conference in Dublin 1992 declared that "water is an economic good" (ICWE, 1992). This is in contrast to the wording in the European Union's Water Framework Directive, which says that "water is not a commercial product like any other but, rather, a heritage which must be protected, defended and treated as such" (European Parliament and Council of the European Union, 2000). These two definitions are outcomes of international discussions and negotiations and yet convey very different messages. In fact, there are limits to the acceptability of regarding water as a good which should be governed by economic thinking and economic instruments alone.

In this setting, water tradeable permits are a subject of increasing interest, as well as concern and controversy. It is worth exploring the reasons why water trading still faces a lot of resistance in practice, even though it is rather popular in theory:

- Different definitions of water resources in legal and economic terms hinder water trade. It is not unusual to find laws which treat public goods as private and vice versa. This especially holds for cases where groundwater rights are linked to land ownership. Groundwater belongs to the land owner and is private property, even though it is fluid and can easily become an open access/common property resource. Therefore, clear legal definitions that match their economic counterparts are needed.

- The fact that in some countries many different laws and regulations apply to water ownership and allocation, makes it difficult to find out who actually owns the water. Therefore, a clear initial assignment of property claims to water resources is very important but difficult to achieve. The initial step towards a market mechanism needs to come from the government (central planning), which has to grant transparent, clearly-defined property rights in order for a market to evolve. The definition and allocation of tradeable rights also needs to be adaptable to new emerging challenges to be able to effectively adjust allocation of rights and the trading rules (Kraemer and Banholzer, 1999).

- Jurisdictional problems and administrative barriers also make water trading complex. This barrier has been acknowledged, for instance, in the case of interstate water trading within the Murray Darling Basin in the Australian Federation. Clearly defined jurisdictional responsibilities and clear administrative structures would ease water trading in the future.

- Unknown factors due to lack of confirmed scientific information hamper trade, as in the case of effluent trading and the uncertainty in estimating non-point source loads in space and time. In general, water is far from fulfilling the standards of a marketable good of homogenous nature. Its changeable and often unknown nature and consistence make policy-makers reluctant to promote water markets. However, it should be kept in mind that changes in resource availability and water insecurity may actually induce trading. With respect to unknown factors, the case of groundwater markets is rather crucial. In specific, even if groundwater property rights are clearly defined, the design of groundwater markets remains complicated, since aquifers are connected. This can mean that pumping of an aquifer can have unknown impacts in the quality and quantity of neighbouring aquifers.

- Similarly, many countries or regions resist to the introduction of water-based trade because its economic structural effects and economic consequences on individual sectors are unknown or judged as potentially negative. Such uncertainties need to be assessed and

confronted in a pragmatic way. For example, in order to protect potential water markets from monopolistic structures, the introduction of intra- or intersectoral water trade needs to be accompanied by long term planning, 'safety' thresholds, as well as regulations that protect the market from monopolies.

- Effects of trading on other users and ecosystems inhibit the establishment of trading regimes in water management. External environmental costs and social externalities have to be dealt with early in the establishment of a trading scheme to avoid conflicts. The setting up of a good monitoring system can be helpful in resolving such issues and controlling the effects of the trading activity on other users and the environment.

- Social considerations in general increase the "political cost" of an introduction of tradeable permits for policy makers. Such schemes have the ability of exposing implicit and explicit subsidies to use, which often are the answer to legitimate social claims. So, the possibilities of addressing these issues through social/income policies while supporting market mechanisms in water management are limited due to lacking political support.

- Furthermore, water, which is an essential part of natural ecosystems and key to their stability and integrity, is such an important common resource, that it often is a focus of societal organisation. Water management can be linked to the formation of states, such as in Mesopotamia or the Nile valley. Control over water is an important source of power, as can be seen in the history of the US State of California. Central functions of water management, such as flood control, drainage, and irrigation as well as water supply and sewerage are of such importance to human existence, food production, social development and economic activities, that many people come to regard water as "too important to be left to economics". Instead, water is regarded as a resource that should be managed, protected and allocated by collective decision-making, allowing for transparency and building on the participation of all those affected. In effect, there are wide-spread misgivings about allowing water to be allocated by economic mechanisms alone, as would be the case with tradeable water rights. Robert Redford's "Milagro Beanfield War" could be cited as an example.

- In many religions, water plays a prominent role in creation myths and is used in rituals. In the eyes of many, water is provided by nature or a divine power. As a good of such origin, it may then not be appropriated by any one person and denied to others. In some parts of the world, denying water even to your enemy is considered a sin. In others, it is a natural part of hospitality to provide a stranger or visitor with water to drink. Given its position in value systems informed by culture and religion, it is understandable that there can be opposition to charging a price for water, especially if non-payment is then punished by exclusion from the service if water is cut off. Denial of water supply services, in this context, would be regarded as an equivalent to denial of water to a thirsty person. In a milder form, the same sentiment can be seen in opposition to regressive and thus "socially unjust" water tariffs.

Some of the reservations against water pricing and water trading may be overcome, if economic instruments are tempered by participatory decision-making and placed within appropriate governance structures. In particular, distinguishing between "essential use", such as drinking water or water for basic food production, and luxury uses, such as water for swimming pools or the production of non-essential foods (e.g. strawberries in arid regions) may ease concerns. Nevertheless, in the practice of water management, economic principles will usually come second to societal dynamics and politics.

6. Conclusions and outlook

The following conclusions on the use of tradeable permits in water management are based on the findings of this review as well as the previous review by Kraemer and Banholzer (1999). The results of the informal OECD workshop on new areas for the application of tradeable permits in Paris on 6-7 September 2001 have also been consolidated.

- There is a variety of schemes of water tradeable permits already implemented, which allow for only a few general conclusions. In most cases, the schemes of water tradeable permits need to consider the regional physical scale for the development of a market, and bear in mind the specific framework of each culture and region. Therefore, when it comes to water resources, markets are locally adapted, and no global market exists as for greenhouse gases.

- Successful trading regimes tend to be built on pre-existing institutions and are integrated into traditional regulatory regimes. This often includes the combination of tradeable permits with other policy instruments (taxes, charges, environmental quality objectives) in order to form 'instrument mixes' for more effective water management.

- "Good" markets for water tend to be found in irrigation districts, with frequent trades and many market participants. However, therein often lies the danger of developing a monopoly power, when 'insiders' hold a large proportion of the water rights. "Bad" markets, on the other hand, are characterised by low trading activity. This is especially the case when (permanent) rights, rather than the water itself, are traded.

- There is growing opportunity for the application of tradeable permits in water management, by using the structure and established user relationships within the Water User Associations (WUAs), as in Chile and Mexico. In the appropriate context, community based regimes embedded in local customs can be both effective and efficient (Kraemer and Banholzer, 1999). Some form of water markets embedded in local customs have already evolved even at the absence of a formal institutional context. Such markets are often characterised by frequent informal transactions.

- With respect to WUAs, the issue of stakeholder participation in water trade also gains importance. WUAs in Mexico, for instance, trade water but consist of a restricted group of users. The latter implies a concentration of market power in the hands of the few WUAs, leaving other users outside the market and limiting the theoretical efficiency of water trade. Nonetheless, a market for water within WUAs is an initial and valuable experience with trading, which can lead in the future to the participation of other additional stakeholders in the market.

- In view of the unsustainable mining of groundwater (e.g. in Texas, US but also Mexico and Spain), tradeable permits in the right institutional context may become more important to control such developments. This holds also for cases in which groundwater rights are linked to land ownership, and no appropriate abstraction limits or water conservation authorities exist. However, due to the limited experience with groundwater trading schemes, their relevant potential consequences for the environment have not yet been revealed.

- As seen from some case studies in this review, there is an increasing interest in inter-sectoral water trade. In regions where water is scarce and water use is inefficient mainly in connection to agricultural infrastructure, there is a growing need to promote trade between urban and rural users. Towns often appear willing to pay farmers to improve system losses and then trade with them on the available water resources.

- More work, both theoretical and practical, needs to be carried out on the application of water tradeable permits at the international level involving two or more countries. Such an approach to transboundary water management across political borders is important not only for water allocation but also to combat modern transboundary pollution problems, especially in connection to non-point sources of pollution.

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ANNEX 1

In the following tables, key information from the review of tradeable water permits by Kraemer & Banholzer (1999) is included. Facts are mentioned only for the case studies also presented in the present paper to allow the reader to acquire more background information on the trading schemes..

CASES IN TRADE OF WATER ABSTRACTION RIGHTS

	Northern Colorado Water Conservancy District, USA	**California Drought Water Bank, USA**	**Texas, Rio Grande, USA**	**Interstate trading, Murray Darling Basin, Australia**	**New South Wales, Australia**
Place, Initiation date of programme	State of Colorado.	State of California, . 1991	Rio Grande River Basin	Murray Darling River Basin	Murray region; a water exchange (SRIDC) was set up in 1997 (annual transfer existed since 1982 and permanent transfer since 1993)
Object of trade	Annual water shares of the Colorado Big Thompson (C-BT) Project allocated by the District to share owners	Units of water (acre-foot) on a one-year basis	Units of water (acre-foot) annually leased or permanent water rights sold	Parts of each State's entitled water shares	Annual water entitlements
Participants	District irrigators and cities in the region	Public and private users, farmers	Agriculture, mining, municipalities. Agricultural users are mainly sellers, while municipal users are mainly buyers	The States of New South Wales, Victoria, Queensland, South Australia	Murray Irrigation Limited members
Design characteristics (trade management, authority for operation, design, decision making)	The Northern Colorado Water Conservancy District (NCWCD) manages rentals, sales and leases. . Property agencies broker sales of water	Established by the California Department of Water Resources.. 1991 Central Valley Project Improvement Act was passed to allow water contractors to sell water outside their districts	N/A	Under the Murray Darling Basin Agreement, each State has rights to a share of the river water.. The Murray Darling Basin Commission is responsible for water management in the Basin	Scheme set up and managed by the Southern Riverina Districts Council (SRIDC), representative body of the Murray Irrigation Limited shareholders
Number of trades, volumes, prices	Price for permanent rights rose from US$99/acre-foot in 1961 to US$900/acre-foot in 1985. Annual rental price at US$5-7/acre-foot.	In 1991, the Bank bought 825 000 acre-feet at US$125/acre-foot, and sold 435 000 acre-feet at US$175/acre-foot	US$15-20/acre-foot per year for annual leases, US$300-700/acre-foot for sale of permanent water rights	N/A	Market clearing price set at A$45 (500 megalitres traded) in 1997, rising to A$67 in 1998
Evaluation, comment	C-BT water is a supplemental supply which plays an important role in its marketability	The scheme resulted in increased efficiency in water use. California's water supply from storage facilitates transactions		By 1997, the option of interstate sales of water was not available to irrigators	By 1998, small amounts were traded due to short experience of irrigators with trading and the temporary nature of the water market

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ANNEX 2: CASES IN TRADE OF WATER POLLUTION RIGHTS

	Tar-Pamlico River, USA	**Lake Dillon, USA**	**Cherry Creek, USA**	**Interstate salinity trading, Murray Darling, Australia**	**Hunter River, Australia**	**South Creek Bubble Licence, Australia**
Place, Initiation date of programme	Tar Pamlico River Basin, North Carolina,. 1989	Dillon Reservoir of Summit County	Cherry Creek reservoir near Denver	Murray Darling Basin,. 1992	Hunter river of New South Wales, 1995	Hawkesbury-Nepean river, New South Wales
Object of trade	"Shares" to the "bubble" emission limit of 425000kg/year	Discharge rights of phosphorus load	Phosphorus reduction credits	It came into force in as part of the Murray Darling Basin Salinity and Drainage Strategy. Salt credits earned by investing in capital works to manage salt entering the river	Salt discharge credits	"Shares" of nutrient loads within the bubble licence
Participants	Sewage treatment plants and industrial dischargers (Association Members), non-member farmers	Municipal treatment plants, small treatment plants, non-point source polluters	Dischargers of nutrients	Governments of the States of New South Wales, Victoria and South Australia	Coal mines and Pacific Power (operator of two hydropower stations)	Sewage treatment plants belonging to Sydney Water services operator
Design characteristics (trade management, authority for operation, design, decision making)	In 1989, a basin-wide bubble for nutrient pollution was established, administered by the Tar Pamlico Association (also broker for trades). The Association also facilitates transactions among member pollution dischargers and with non-point sources (non-members). A point source can pay fees to a fund used to reduce nutrient loads from non-point sources	The Dillon Water Quality Management Plan set a limit for total phosphorus load. Point dischargers were granted annual discharge rights.. Point/non-point trades were allowed at a 2:1 ratio. Trading is to be approved by the basin management authorities	Point sources allowed to earn phosphorus credits through control of non-point sources.. Non-point sources need to reduce their loads by 50% on their own, before point sources can earn credits	The Murray Darling Basin commission administers the scheme.. Salt credits are tradeable, but each State generally applies them to offset debits from drainage entering the river system	The Environment Protection Agency of NSW developed the trading scheme from the existing salt licensing scheme.. Each discharger is allowed to discharge a specified percentage of the total allowable salt load.. The EPA reserves the right of approving trades and conducts regular audits of the scheme	The Environment Protection Agency of New South Wales developed a "bubble" licence regime to control the aggregate nutrient load from sources in the bubble.. The 2004 target (aiming at significant nutrient reductions) and interim targets developed to ensure environmental improvement
Number of trades, volumes, prices	N/A	Until 1988, no trading was approved since critical loads were not exceeded	Until 1992, no trading existed because economic gro-wth slowed down	N/A	Until 1997, 1 trade had taken place	N/A
Evaluation, comment	Overall discharge of nutrients into the basin reduced by 28%				Initial experience showed conductivity levels remained within target limits	Cost savings expected to be in the order of 10-20%

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Part IV

FUTURE DIRECTIONS

Chapter 8

CONCLUSIONS: OUTSTANDING ISSUES AND AREAS FOR FURTHER WORK

by

Olivier Godard
Laboratoire d'économétrie, Ecole Polytechnique, Paris

The review and assessments of existing domestic TP systems in OECD countries, carried out during the first phase of the current OECD project provided some paradoxical lessons. For instance, the study on the use of the TP concept in water management recommended not to opt for TP schemes conceived to deliver market attributes such as transparent property rights, competitive prices that publicly disclose information on rational decentralised agents. It could result in complete failures. Instead, water rights trading programmes should be grounded into existing institutional arrangements based on local community or water user organisations, even if it involves informal, non-transparent patterns of relationship in water usage.

As for solid waste management in the UK, a seemingly successful TP programme has evolved from a system of packaging recovery notes (PRNs) that was not initially intended by authorities to be a TP programme, but only a mechanism to ensure compliance to obligations under the Extended Producer Responsibility Principle.

In the field of transport, the Austrian Ecopoint programme imposing quantitative restrictions on emissions of truck fleets coming from abroad triggered a great progress in technological performance of trucks although no trading was allowed. Meanwhile, the Zero Emission Vehicle programme in California has not yet achieved its objective in spite of the fact that trading opportunities were fully incorporated into the design.

In climate change policies, for the first time in contemporary history, the Kyoto Protocol provided an internationally agreed framework for developing a TP scheme. The benefits of having internationally harmonised basic rules of the game are to become available. However, some OECD governments prefer to develop separate domestic emission trading programmes framed by specific rules and particularities, in spite of the fact that it would be beneficial to connect and integrate these domestic programmes for the first commitment period under the Protocol (2008-2012).

The UK government is presently setting-up a domestic GHG trading scheme which is rather complicated, combining several features: opportunity given to firms to choose a scheme with absolute or rate-based targets, restrictions of trading between those with absolute targets and those with rate-based targets, and a reverse auction scheme to distribute subsidies to firms to stimulate voluntary participation in combination with a Climate Change Levy. What impacts the complexity of such a scheme would have on the potential efficiency gains of the TP scheme remains to be seen.

These examples show the difficulty of designing TP schemes. It requires the integration of analytical results from economic analyses and the political economy of choice of instruments that are typical of policy process.

TPs are not yet a widespread, familiar policy instrument, and their development could contribute to modernisation of environmental policies in many areas. But the exploitation of this large potential by domestic policy-makers could benefit from further work on specific topics pertaining to TPs, which could build on previous analytical work and general guidelines already available through earlier OECD work. It could also benefit from the guidelines elaborated by several countries for specific programmes to explain how businesses and others can use TPs according to the specific rules chosen for one application. Although the latter guidelines are for users and not for designers of programmes, it would be a useful step to gather and cross-examine them all to highlight the variety of answers given to the same questions and to define benchmarks.

In light of the above, further work by OECD could include several components:

i) improving the methodology of *ex post* evaluations of practical experiences of TP schemes - much can be learned from first attempts of evaluations;

ii) a scoping study on policy tool combinations, looking at the various possible interface of TP schemes with taxes, regulations, voluntary approaches, and other TP schemes - TPs are never used in isolation and do not constitute by far the dominant policy instrument presently in use in environmental policies;

iii) due to the prevalence of global climate policies for the future development of TP schemes, a specific project could be devoted to the integration of domestic GHG emissions trading schemes with regional and international ones; and

iv) a number of topics are recurrent in many discussions on implementation of TPs, demonstrating a need for further clarification papers; these topics (discussed further below) need broad descriptions, key analytical points, various solutions in use or envisaged and their assessments, and recommendations for policy-makers.

1. Ex post evaluations

Ex post evaluations are of critical value for several purposes: (i) improving the performance of existing instruments; (ii) enlightening the design of new TP schemes and future choices of policy instruments; (iii) ensuring a more confident communication between authorities, various stakeholders and the public; and (iv) revealing the gap between *ex ante* expectations derived from the design phase of a policy tool, and what it becomes in practice under implementation.

First, it is important to examine the real impacts of a TP programme on the state of the environment, as only under TP schemes with absolute targets and caps, we can expect certainty over environmental performance. In fact, the importance of focusing on the environmental outcomes in *ex post* evaluations should be recognised for several reasons: (i) TPs may involve a significant geographical redistribution of emissions, then raising a concern for pollution "hot spots" or ecological inequities; (ii) building confidence in the environmental performance of a programme still requires empirical and verifiable information about actual achievements; (iii) some schemes may result in overcompliance during the first years of a programme (noticeably because of provisions for banking) or alternatively in implicit undercompliance, when several provisions (e.g. allocation of bonus allowances, substitution between obligated sources, etc.) alter the rationale of a cap-and-trade scheme; and (iv) any TP scheme based on relative or unit targets is uncertain *ex ante* regarding the overall environmental performance that is to be achieved, making it necessary to verify what has really happened.

Beyond this first point, and on the basis of the proposals made in the paper on *ex post* evaluations[1], five directions for further work can be pointed out:

a. methods for defining counterfactuals/baselines

Carrying out an evaluation implies making comparisons. In the case of *ex post* evaluations, actual achievements have to be compared to something that can be called a counterfactual or a baseline. Defining an agreed counterfactual is a very difficult exercise since, generally speaking, many scenarios could have happened in place of the actual history. In several fields, such as forecasting future energy sector scenarios, it has been shown that it is not possible to trace a unique counterfactual even when there is an agreement to base a business-as-usual scenario on econometric analysis of past trends: results depend on models used to rebuild tendencies and how far back in the past the data series is included. Examining different methods for setting a counterfactual would be a useful exercise, and there is certainly much to take from the debate that has taken place in the context of the implementation of the UNFCCC. One solution that has a pedagogical virtue is to keep in mind that future prospects are neither unique nor given, and thus an evaluation can be made against two or three alternate counterfactuals. Meta-analyses of existing schemes and exploratory work could be undertaken

b. administrative and transactions costs

Administrative and transaction costs are the general category to embrace various costs of running a regulatory regime, for the regulated as well as for the regulatory authorities. They explain a lot about the gap in the results between *ex ante* and *ex post* evaluations of a programme. They are often referred to in the academic literature, but empirical findings are rather scarce and limited in scope. They generally raise the issue of imputability of various costs to a specific programme. A clarification of different types of costs, and of methods to assess each of them would prove useful. Also, the cost of monitoring and enforcement should be considered here. How frequently should reporting and monitoring be done? What types of sanctions (financial, legal, reduced rights/permits in the next period, etc.) should be applied in case of non-compliance? All these add to the overall cost of running a TP programme.

c. how to appraise the efficiency of TP markets

Economic efficiency of a policy instrument cannot be directly observed. Information delivered by prices are useful but cannot be taken as direct indicators of efficiency, i.e. the extent to which social costs are minimised for a given environmental goal. This issue is partly related to the counterfactual issue, but should be documented separately to discourage futile debates and opportunistic interpretations of data on observed prices. For instance, clarifications on measurements of the efficiency gains derived from the stability and predictability of permit prices would be useful for both *ex ante* and *ex post* evaluations.

d. the institutional dimension of evaluations

The institutional context may have a significant influence on the quality of evaluation, but also on the willingness of various stakeholders to accept the findings and incorporate them in policy modifications and refinements. In other fields of public concern, such as food safety and health, individual experts and expert committees have been exposed to strong criticisms, being accused of either being captured by dominant industry lobbies, or imposing their own normative values under the appearance of objective facts and findings. To restore confidence in collective expert bodies for

1. Chapter 2, S. Smith (2001).

evaluations, reforms have been introduced in several countries (e.g. France) or regional institutions (e.g. EU) to ensure high scientific standards, guarantee objectivity and pluralism, and publicly disclose the findings. This is important especially when such an expert body is convened in response to a request by the public authority responsible for what is being evaluated. In parallel, proposals are often made to include representatives of the public as part of an evaluation process by an expert body such as a "steering group", at least to discuss mandates and key directions, but also to ensure that procedures are adequately followed. To the extent that the development of TPs has been controversial in certain circles (e.g. some environmental NGOs) and in the eyes of the public, lessons should be drawn from these recent evolutions in how evaluation processes are arranged. How should the objectivity of an evaluation be ensure, as a collective activity taking place within a specific group, and the acceptability of its results?

e. costs of evaluation and access to information

Related to the issue of the institutional context discussed above, another area of useful work would be to examine the relationship between data accessibility and the amount and type of evaluation research that is undertaken. A team carrying out an *ex post* evaluation commissioned by the authority responsible for a TP programme would naturally benefit from better access to information needed for the study, while independent evaluation research may not have the same level of access to data. Such independent research activities can expand the amount of evaluation evidence available, and may make a substantial contribution to innovation and the development of good practice in evaluation research. One factor that may greatly affect the amount of evaluation work originating from academic and other independent sources is the amount of data about the operation of programmes routinely placed in the public domain.

2. Combinations of policy tools

Combining various policy instruments is the current practice in environmental policies. However, most analyses of policy instruments consider their operations in isolation, which does not shed much light on their joint impacts when they are combined. Thus, a specific study of combinations of policy instruments or policy mixes could be useful. It would have to cover issues such as:

- patterns of relationship: division of labour (by sector, by scale of business), versus integration in one composite instrument for the same population of obligated agents;
- useful complementary aspects and possible negative interference to be avoided;
- possible efficiency and equity distortions resulting from co-existence of multiple instruments;
- borders and linkages/interfaces between instruments;
- specific issues related to TPs and regulations (technology standards, BAT) - are they still necessary when TPs are in place? Do they raise obstacles to the economic benefits of TPs?
- specific issues with TPs and taxes, for cases including tax exemptions granted for voluntary participation in a TP scheme, introduction of taxation as a safety valve when TP prices become excessive, and a mix of TPs and taxes with free allocation of TPs for a majority of emissions, e.g. 80%, and tax for the residual part, e.g. 20%; and
- specific issues with voluntary approaches (VAs) - negotiated or voluntary agreements - may be used to define individual pollution abatement targets, with the possibility to earn tradable credits when the targets are surpassed. Such a solution should address specific constraints on the homogeneity of targets and on monitoring of the VAs. It also raises the issue of fairness

related to the possibility of very unequal efforts embedded in various voluntary agreements and commitments.

This work should be closely integrated with the current work by the OECD on environmental taxes and voluntary approaches.

3. Integration of domestic TP schemes with other regional and international schemes

A more extensive use of TPs at various levels (local, national and international) will presumably multiply cases of interactions between different schemes with different rules. For climate change, it would be the case of the proposed EU-wide emissions trading scheme superimposed on domestic ones, both to be integrated into the international emissions trading programme under the Kyoto protocol for the 2008-2012 period. How is it possible to organise the linkages and restrictions in order to maximise overall advantages from such a pluralistic development of TPs, while avoiding undue escape from local or domestic responsibilities regarding environmental quality? Questions relate to differentiating and articulating domestic systems in different countries as well as 'cap and trade' and 'baseline and credit' schemes. A specific case should be made for electricity generation as a sector affected by both problems. Through the Annex I Expert Group[2], IEA and OECD are carrying out substantial work on this topic, in view of the integration of domestic trading programmes with the Kyoto mechanisms.[3] Further work could include additional issues such as the legal framework and structure of environmental governance required at different levels.

Furthermore, the issues of integration are not limited to GHG emission abatement, but could also be applicable to other areas such as waste management (in light of the EU directives that set common standards or requirements in recycling and landfilling) and river basin management (as water pollution and water resources use are often transboundary issues).

4. Clarifications on specific topics

A number of topics deserve to be considered for further clarifications, to explain the questions raised, review experiences, and present ways and means to tackle the main issues involved. The following suggestions identify some of them and give brief indications on the sub-themes that could be investigated.

(a) Structural change and technological innovations

It is often contended that TPs would not be the best instrument to induce structural change and prepare long-run solutions to environmental problems. Sometimes, it is argued that, by cutting short- term costs, TPs reduce the incentives to introduce more progressive technologies. Along with this argument, TPs need to be integrated within a structural policy using other policy tools (i.e. public spending on R & D, public investments in infrastructures, stringent technological standards, education efforts, etc.) Another view is that a well-designed TP scheme may work as a substitute for planning, specially in areas where technological progress is crucial: the critical point is the long-term planning of

2 . The Annex I Expert Group consists of representatives from countries listed in Annex I of the UNFCCC, i.e. most OECD Member countries, and some countries from central and eastern Europe, and the Commonwealth of Independent States.

3. See for example, E. Haites and F. Mullins (2001), *Linking Domestic and Industry Greenhouse Gas Emission Trading Systems; http://www.iea.org/envissu/epri.pdf.*

an evolution of the environmental target over time. Are TPs really delaying or opposing structural and technological change?

Also, given that existing participants in a TP scheme (assuming to be older pollution sources) generally have sunk costs to bear while new entrants (assuming to be newer facilities) have access to the latest technology, what is a fair treatment between the two types of firms? Answers may have impacts on their investment decisions and the pace of adoption of new technologies. How to treat new entrants is a recurrent issue raised in designing and implementing TP schemes. Although the problem is more serious with cap-and-trade schemes, it is also relevant for baseline-and-credit systems, as different baselines can be used for existing sources and new entrants. With cap-and-trade programmes, a standard answer is that new entrants have to buy TPs from present owners, when the total cap has already been fully allocated to existing sources. Another solution is to reserve a given percentage of permits under a cap for a given period (e.g. one year) for new entrants, and to distribute them for free on the basis of the rule of "first-come-first-served" until the permit reserve is fully allocated. One solution that establishes strict equality between all sources is to organise periodic auctions.

Another consideration is how not to penalise firms that have invested in pollution abatement technology early and therefore have already achieved environmental results prior to introduction of a TP scheme. If initial permit allocation is to be free on the basis of "grand-fathering" based on historic emission levels, those with higher recent and current emission levels due to lack of earlier abatement actions would benefit by being granted more permits. One solution to this would be to use different "grand-fathering" periods as the baseline for free initial permit allocation for those that have invested earlier in pollution abatement.

(b) The political economy of designing TP schemes

TPs have been advocated by environmentalists and economists for their strengths that combine cost-effectiveness and certainty over their environmental performance. Such properties can be materialised only if the programme design addresses monitoring of environmental performance and facilitates trades. However, this has not always been the case. In some situations, public concerns over the environmental performance led to removal of critical aspects of flexibility, such as banking, limiting the ability of TPs to reduce compliance costs significantly. In other cases, the goal of economic efficiency is almost forgotten, and public debate in the policy process becomes captured by interest groups (or stakeholders) over distributive and acceptability issues. Thus, designing a new TP programme is often faced with political and social realities that could interfere with its potential advantages in economic efficiency and environmental effectiveness. Therefore, the policy process in developing a TP scheme, as well as the design options, should be adapted to the socio-political contexts in which TPs are intended to be applied. In this regard, guidelines in designing TP schemes should not be monolithic. Is it possible to draw some guidelines on which case should be applied in which contexts?

Especially for cap-and-trade programmes, one controversial issue is deciding on the initial permit allocation method, due to its distributive implications. Advantages and drawbacks of different solutions have to be scrutinised by considering the following points:

- Different combinations of allocation of rights and obligations
- Free initial allocation versus auctioning of rights
- Different types of auctions
- Various criteria for free initial allocation for absolute quotas and relative credits

- for existing businesses
- for new entrants

- Duration of permits

- Compatibility with domestic, regional (e.g. EU) and international laws limiting public subsidies and state aids in the context of competition regulatory regimes.

(c) Market structures and rules for TPs

TPs are market instruments in their most developed forms involving external transfers. Their environmental effectiveness and contribution to overall economic welfare depend on the good working of the markets involved: an efficient allocation would result from flexible and competitive markets. Policy-making should consider both the structure of the market for TPs (e.g. sulphur emission allowances), and the markets for products generated by targeted activities (e.g. power generation). How the markets operate determines how revenues are distributed among participants. Some market structures allow some participants to capture rents at the expense of others. This is the reason why it is generally claimed by economists that market structures are relevant to TP design decisions on aspects such as initial allocation of permits (i.e. for free or by auctioning, in what amounts?) and temporal flexibility (e.g. unlimited banking could allow some firms to accumulate market power). Moreover, interactions between the permit markets and the product markets should be considered, since limitations of transactions of TPs may in some circumstances improve the efficiency of the products market. New developments of TPs that are focused on increasing the level of some activity (e.g. recycling) instead of decreasing pollution also add new dimensions to consider. It would be useful to develop clear benchmarks by which policy-makers could design a new TP scheme, taking market structures into account.

(d) Voluntary participation and "opting-in"

Generally speaking, firms prefer voluntary programmes to compulsory ones. This is reflected in the development of voluntary agreements in many countries and, more specifically, in the UK domestic GHG emissions trading scheme. However, efficiency, fairness, and credibility of such voluntary approaches are still to be clarified. A contractual approach whereby public authorities offer several options at different compliance costs may be useful, by having decentralised agents reveal information about their marginal abatement costs through self-selection. However, such an approach would require authorities to organise and manage several regulatory instruments. Voluntary entrance and possible opt-in provisions for a TP scheme may not guarantee the overall emission cap to be met, and add a factor of unpredictability in the equilibria and price trends in the TP market.

(e) Adaptability vs. stability

While it is important to keep the design of a TP programme flexible so as to allow adaptability and learning, it is often advocated that predictability and stability of the rules of the scheme is important to assure confidence of the regulated agents. Therefore, the design of a TP programme needs to be flexible and adaptable, yet stable and predictable.

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