



Tradeable Permits

**POLICY EVALUATION, DESIGN
AND REFORM**

Tradeable Permits

Policy Evaluation, Design and Reform



ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT

ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT

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FOREWORD

In 2003, the OECD Environment Directorate organised a workshop on the 'Ex Post Evaluation of Tradeable Permits'. The workshop brought together both economists who have been responsible for undertaking policy evaluations of tradeable permit systems, as well as policymakers responsible for the design and implementation of the programmes themselves. In addition to reviewing the efficiency of existing tradeable permit systems, it also sought to shed light on the evaluation process itself, as well as on more general issues associated with the links between policy evaluation and the public policy process. This publication brings together the reports presented at the workshop, as well as a general review of the recent secondary literature on the evaluation of tradeable permits, and on policy evaluation generally.

Tradeable permits were chosen an especially suitable focal point for examining the wider issues associated with policy evaluation since their use is increasing rapidly (including, but not limited to, the requirements created by the Kyoto Protocol), and as such the demand for insights derived from the evaluation of actual experience is great. In addition, tradeable permits have been controversial and are somewhat unfamiliar to most decision-makers, and this adds to the need for solid information upon which policymakers can base their decisions. And finally, tradeable permit systems are also somewhat unique in that in the normal course of their operation they reveal information about the costs of compliance and estimated emissions, and can even provide motivation for significantly improving the quality of the data necessary for sound policy evaluation. This can significantly reduce the cost and increase the reliability of ex post evaluations.

In this report we examine how tradeable permits have been applied in a variety of different contexts. These include controlling air pollution, the degradation of wetlands, agricultural pollution, water scarcity and fisheries depletion. In addition to focussing on a wide variety of natural resource and pollution applications, the case studies cover a variety of geographic locations (Australia, the Netherlands, New Zealand and the United States). Moreover, existing evaluations have employed a wide variety of methodologies, assessing different evaluation criteria, and undertaken under different institutional arrangements. As such, the workshop discussions, the case studies and the review of the literature highlighted the diversity of uses to which evaluations can be put, as well as the diversity of means by which they are undertaken and integrated with the policy framework more generally.

Tom Tietenberg (Department of Economics, Colby College, Maine) and Nick Johnstone (National Policies Division, OECD Environment Directorate) were responsible for the preparation of the publication. Special thanks is extended to the academics responsible for the preparation of the case study evaluations, as well as the government officials who presented overviews on their national policy evaluation systems, and provided valuable inputs to the workshop discussions. Carrie Delecourt provided invaluable assistance both in the organisation of the workshop and in the preparation of the final manuscript. The book is published under the responsibility of the Secretary-General of the OECD.

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Chapter 1

EX POST EVALUATION OF TRADEABLE PERMITS: METHODOLOGICAL ISSUES AND LITERATURE REVIEW

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I. Introduction

The workshop on the ‘Ex Post Evaluation of Tradeable Permits’ held at the OECD on January 21st–22nd, 2003 had multiple objectives. It sought not only to add to our stock of knowledge about how well tradeable permit systems work and the design features that contribute to desirable outcomes, it also sought to use detailed analysis of case studies to shed light on the evaluation process itself, as well as on more general issues associated with the links between policy evaluation and the public policy process.

Alongside environmentally related taxes, tradeable permits (TP) are increasing in importance in OECD Member countries as governments seek to introduce more flexible market-based instruments to address environmental problems. However, unlike taxes they are quantity-based, rather than price-based, environmental policy measures. A quantitative limit is introduced either as a maximum ceiling for “cap and trade” schemes, or as a minimum performance commitment for “baseline and credit” schemes. Moreover, these targets can be either in absolute terms or in relative terms. And finally the permits can be denominated either in terms of ‘bads’ (i.e. pollution emission) or ‘goods’ (i.e. natural resources). Amongst economists, and increasingly amongst the environmental policy community in general, it is felt that TP schemes can provide significant benefits in terms of economic efficiency and environmental effectiveness (see OECD 2002 for a full definition).

The workshop was designed to build on previous work at the OECD on the use of tradeable permits (OECD 1999a, Godard 2001, OECD 2002), as well as the ex post evaluation of environmental policy instruments (OECD 1997a and Smith 2002). Tradeable permits were seen an especially suitable focal point for examining the wider issues associated with policy evaluation for a number of reasons:

1. The views expressed in this chapter are those of the authors and do not necessarily reflect those of the institutes with which they are affiliated, or the OECD.

- Since their use is increasing rapidly (including, but not limited to, the requirements created by the Kyoto Protocol), the demand for insights derived from the evaluation of actual experience is high;
- The fact that tradeable permits have been controversial and are somewhat unfamiliar to most decision-makers has added to the need for solid information upon which to base decisions;²
- Tradeable permit systems are also somewhat unique in that in the normal course of their operation they reveal information about the costs of compliance for affected firms through permit price trajectories, facilitating evaluations; and,
- The process of assuring compliance generates information on estimated emissions, and can even provide motivation for significantly improving the quality of the emissions data,³ also reducing the cost of conducting an evaluation.

In this report we consider how tradeable permits have been applied in a variety of different contexts. These include controlling air pollution, the degradation of wetlands, agricultural pollution, water scarcity and fisheries depletion. In addition, the case studies focus not only on a variety of natural resource and pollution applications, but a variety of geographic locations as well (Australia, the Netherlands, New Zealand and the United States).

Variations in experience can be exploited to isolate a wider range of lessons for both the design and use of tradeable permits and the design and use of ex post evaluation. Providing a wider variety of contexts allows us to begin to see precisely how context matters. Moreover, the case studies employed a wide variety of methodologies, assessing different evaluation criteria, and undertaken under different institutional arrangements. The workshop discussions and the case studies highlighted the diversity of uses to which evaluations can be put, as well as the diversity of means by which they are undertaken and integrated with the policy framework more generally.

Finally, while this workshop dealt explicitly with ex post evaluations, the extent to which the discussions drifted so easily and so frequently into other forms of evaluation indicates the degree to which all forms of evaluation are ultimately linked. Ex ante analysis is frequently a stepping-stone for ex post analysis, and ex post evaluations themselves can serve very different functions. While evaluations performed early in the process may not be particularly helpful in informing definitive judgements on how well the programme ultimately works, they can be very helpful in providing the feedback necessary to shape the evolution of the system over time. Indeed, one central message from these case studies is that the programmes rarely emerge fully-formed. They evolve over time and interim feedback can be particularly valuable.

This introductory chapter is divided into four sections. Section I reviews the criteria which can be used for the evaluation of environmental policy instruments in general, and tradeable permit systems in particular. Section II looks at general methodological issues associated with undertaking policy

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2. Consider just three examples. In air pollution control a legal challenge was brought in Los Angeles during June 1997 by the Los Angeles-based Communities for a Better Environment (Tietenberg 1995). In fisheries a challenge was brought against the halibut/sablefish tradeable permits system in Alaska (Black 1997) and the US Congress imposed a moratorium on the further use of a tradeable permits approach in US fisheries (National Research Council Committee to Review Individual Fishing Quotas 1999). Though both legal cases were ultimately thrown out and the moratorium has now been lifted, these legal challenges illustrate the controversies that can arise.
 3. This was particularly true for example in the US sulfur allowance programme where the programme required continuous emissions monitoring for the first time (Ellerman et. al. 2003).

evaluations, while Section III provides a brief review of ex post evaluations of tradeable permit systems, including those discussed at the workshop. And finally, there is a brief concluding section.

The remainder of the report is made up of the six case studies presented: the American SO₂ allowance program; Los Angeles's RECLAIM program, the Dutch nutrient quota trading system, individual transferable quotas for fish stocks in New Zealand, the American wetlands credit trading scheme, and water trading in Australia. In all cases the results are presented, along with a discussion of methodological issues and policy implications. The report concludes with a discussion of the more general issues associated with the management of ex post evaluations and their integration with the policy process.

II. Criteria for the Ex Post Evaluation of Tradeable Permits

OECD (1997a) lists a number of criteria by which environmental policy instruments can and should be addressed:

- *Environmental effectiveness* – the extent to which the policy meets its intended environmental objective. For pollution emissions this might refer to overall emission levels, while for natural resource use this might refer to overall levels of resource exploitation. However, spatial and temporal effects, as well as interactions with other environmental impacts should also be considered.
- *Economic efficiency* – the extent to which the policy can achieve its stated objective at minimum cost. Thus, for a given level of emissions or resource use a policy instrument is economically efficient if it results in minimum compliance costs across all affected parties.

These two are the principal criteria by which any environmental policy should be assessed. However, there are a number of additional elements which could be considered as well. Implicitly some of these criteria are incorporated within the criterion of economic efficiency, but they are of sufficient importance to warrant separate identification. These may include:

- *Dynamic effects* – the extent to which the policy generates incentives for optimal rates and direction of technological innovation, as well as increased diffusion of such innovations and temporal efficiency in investment patterns;
- *Administration costs* – the cost to public authorities for the management of the policy regime. This includes monitoring and enforcement costs, as well as the cost of initially implementing the policy;
- *Public finance* – since some measures generate government revenue (i.e. taxes or auctioned permits) and others necessitate government expenditures (i.e. environmentally-motivated subsidies) it is important to assess impacts upon public finance;
- *Wider economic effects (i.e. distribution and competitiveness)* – while not the explicit objective of any environmental policy measure, given their importance in public policy in general, such impacts can be of sufficient importance to warrant explicit consideration; and
- *Soft effects (i.e. managerial change)* – while exceedingly difficult to quantify, different instruments can have very different implications for public attitudes, bureaucratic culture and behavioural responses with respect to the mitigation of environmental damages.

It has been widely argued that tradeable permits perform well on the basis of these criteria, and as such they have been advocated as an efficient and effective means to manage environmental resources for over three decades. Following on from the insights of Coase, they represent a quantity-based means of rationing access, through the creation of property rights.

Table 1.1 provides an overview of the more specific criteria which can be used to assess environmental policy instruments, as well as some of the relevant examples related to tradeable permits. It is important to note that some criteria explicitly subsume other criteria within them. For instance, under the heading of economic efficiency, if a programme is evaluated to determine whether it is Pareto-optimal, then implicitly the outcome will indicate whether it is also at least as cost-effective as all other alternatives. Similarly, if a programme is found to be cost-effective, then the market for permits themselves must be efficient. However, a cost-effective programme is not necessarily Pareto-optimal (it may not even pass a benefit-cost test), and nor does an efficient permit market mean that the programme is more cost-effective than any alternatives.

Applying these criteria, the case for the use of tradeable permits has generally been made on the following specific grounds:⁴

- Incentives for abatement cost equalisation;
- Positive technological innovation and diffusion impacts;
- High degree of environmental certainty;
- Relatively low administrative costs; and,
- Flexibility to address distributional concerns.

All of these arguments in favour of the use of tradeable permits are, of course, subject to important qualifications depending upon the particular conditions under which they exist as well as the manner in which they are applied. For instance:

- abatement cost equalisation will not be economically efficient if damages from sources differ depending upon time or place of emission (see Tietenberg 1995);
- some means of using permit allocation mechanisms to address distributional concerns may have adverse impacts on the economic efficiency of the programme, qualifying the separability argument (see Cramton and Kerr 1998);
- tradeable permits may not lead to optimal rates of environmentally-beneficial technological innovation if there are important failures in technology markets, such as finance for research and development (see Kemp 1997); and,
- the start-up costs for cap-and-trade systems and the running costs of many baseline-and-credit schemes can be exceedingly high, particularly if negotiating costs are considered.

4. See Smith (2001) and Tietenberg (2002) recent discussions.

Table 1.1: Policy Evaluation Criteria and Potential Benefits of Tradeable Permits

	General Nature of Criteria	Specific Implications for Tradeable Permits
Economic Efficiency		
Economic optimality	The level of stringency of the target is optimal and the instrument chosen reaches this target at lowest-cost relative to all other alternatives.	The optimal level of the cap is chosen, and the tradeable permit regime results in the equalisation of marginal benefits with marginal costs across all sources.
Cost effectiveness	The magnitude of savings to reach a given environmental target relative for the instrument chosen relative to some alternative instrument.	Equalisation of marginal abatement costs for a given level of emissions. Analogously, for tradeable natural resource quotas, permits will be supplied by those who receive lesser value from their use to those who receive relatively greater value.
Market efficiency	The efficiency of the market – this is not usually relevant for other types of environmental policy instruments.	Potential gains from trade within the permit market are fully exploited – i.e. market power and transactions costs are not significant.
Environmental Effectiveness		
Certainty of aggregate cap	The certainty with which a given environmental target is reached.	A binding constraint on the use of the natural resource or the level of emissions through the cap.
Monitoring accuracy	The extent to which the regulator is able to ascertain whether a given environmental target has been met.	The installation of continuous monitoring systems is required in order to ensure that permit use reflects actual emissions or resource exploitation.
Compliance and enforcement	The likelihood that the regulator will ensure that transgressions are penalised.	Penalties for excessive resource use or pollution emissions are enforced, ensuring that the cap is not breached.
Local or temporal impacts	The extent to which the policy addresses the heterogeneity of impacts by space and time.	If resource use or pollution emissions have different environmental consequences these are reflected within the permit system, such as through trade restrictions.
Soft Effects		
Data accuracy	The extent to which the policy affects the likelihood of having reliable data.	When setting up a baseline-and-credit system, reliable data is gathered on existing emission levels or resource use.
Bureaucratic culture	The extent to which the policy results in more pro-active management of environmental concerns in private and public bureaucracies.	Encouraging firms to see environmental management as analogous to management of financial asset.
Dynamic Effects		
Rate of innovation	The extent to which the policy encourages a rate of innovation which is optimal.	Providing continuous incentives for innovation in environmentally- preferable technologies.
Direction of innovation	The extent to which the policy encourages a direction of innovation which is optimal.	By allowing firms full flexibility in determining the most efficient technological means of mitigation.
Administrative costs		
Start-up costs	The cost of putting in place the programme in the first instance.	Since many baseline-and-credit schemes arise out of existing regulatory systems, can be introduced at little cost.
Running costs	The cost of overseeing and maintaining the programme during the course of its lifetime.	By using a decentralised market as the means of implementation, can be parsimonious with respect to costs for central authorities.
Social Impacts		
Distributional impacts	The extent to which the policy results in adverse (regressive) impacts.	Separation of distributional effects from efficiency effects through the permit allocation mechanism.
Participation	The extent to which the policy allows for broad stakeholder involvement.	By allowing any agent to purchase permits, can encourage broad participation in meeting the environmental objective.

More generally, while many of the arguments in favour of the use of tradeable permits may hold in theory, in practice there can be significant political barriers to their use. As such, it is not sufficient to advocate the case for tradeable permits on the basis of theoretical arguments. For this reason, empirical ex post evaluations of the effects of the use of tradeable permits in practice are very beneficial. Indeed, the case studies included in the workshop addressed these qualifications - and many others - in considerable detail.

Fortunately much of the data required to assess policies relative to at least some of these criteria are gathered routinely in the normal operation of the tradeable permit programmes. Indeed, this is one of the great strengths of tradeable permit programmes relative to other policy measures. Due to their very nature – in particular the use of a tradeable private commodity as the mechanism for implementation of the policy - information and data are generated that facilitates subsequent evaluation. Through data on permit prices and trade levels it is possible to assess many attributes of the workings of a tradeable permit system. (Table 1.2 provides some examples of data required for the evaluation of different types of criteria.)

Table 1. 2: Data Collection Requirements and Evaluation Criteria

Evaluation Criteria	Examples of Data Required
Transaction and Search Costs	Permit Price Dispersion; Time Required to Consummate a Trade
Market Power	Number of Market Participants; Market Entry and Exit
Environmental Impacts	Exceedences of Permit Cap; Spatial Pattern of Trade
Administration Costs	Number of Person-Hours for Administration; Budget of Enforcement Agency
Temporal Efficiency	Permit Price Volatility; Banking and Borrowing Levels
Cost Effectiveness	Abatement Cost Heterogeneity; Unrealised Gains from Trade

III. Methodological Issues in Ex Post Evaluation

Evaluations are a technical exercise, albeit one which is ‘housed’ inside a policy evaluation and management framework. In this section, some of the technical issues associated with undertaking an evaluation are examined.

Ex Ante vs Ex Post Policy Evaluation

The workshop and the case studies presented focused on ex post evaluation of tradeable permits. However, it is useful to distinguish between ex post evaluations and ex ante evaluations, both in terms of their methodologies and their purpose. Most obviously, while ex ante evaluations are undertaken in order to assess whether or not a particular policy instrument is to be introduced, ex post evaluations are undertaken in order to assess whether or not a policy is functioning well, and if not what reforms should be undertaken to improve its performance.

Thus, they are likely to be chronological complements and assuring that the results of ex ante analyses are directly comparable with the results from ex post evaluations would be very useful. This implies methodological consistency through time. If this is assured comparison of results would provide considerable opportunity to learn from the differences. For example, do ex ante studies reveal any systematic tendency toward excessive optimism about the outcome of the programme? Some evidence suggests, for example, that programme administrators tend to underestimate ex ante costs and those being regulated may tend to overestimate them (Harrington et al., 1999). Despite its apparent usefulness, comparability remains the exception rather than the rule.

In methodological terms, the distinction between the two types of evaluation is also striking. In general, ex ante evaluations involve hypothetical simulations - often assuming optimising behaviour on the part of market participants - while ex post evaluations seek to assess the policy on the basis of the revealed behaviour of those affected by it (OECD 1997a). This distinction is important, and is further discussed below.

However, in practice in many cases there may be relatively little to learn from ex post evaluations that could not be ascertained ex ante. For instance, data weaknesses or methodological complexity may prevent an ex post evaluation from obtaining significant additional insights on questions such as distributional impacts beyond those which could have been obtained from a well-developed ex ante evaluation. Nonetheless, it does appear that there are some significant areas where ex post evaluations do 'add value':

- transaction costs associated with permit markets;
- administrative costs associated with programme implementation;
- temporal efficiency by market participants;
- effects on technological innovation;
- the local environmental impacts of specific spatial patterns of trade; and,
- soft effects associated with managerial and organisational change.

To one extent or another the case studies presented provide evidence in all of these areas. Moreover, it appears that many of the impacts would not have been considered in ex ante evaluations, let alone accurately evaluated.

The Choice of Evaluation Methodology

One insight that emerged from the workshop is the need to broaden the definition of policy evaluation. Not surprisingly, the economics profession often places cost/benefit analysis at the pinnacle, with all other alternatives viewed as partial or intermediate. In fact, the studies indicate that many other forms of evaluation can be very useful, not only as stepping stones to more fully-fledged policy evaluations, but also as ends in themselves to assess policies relative to particular evaluation criteria.

Indeed, to a great extent, the choice of evaluation criteria to be assessed will play a determinant role in the choice of methodology. If one hopes to determine whether or not a particular policy is 'optimal' in a formal sense, then clearly a cost-benefit analysis is required. However, this is not always the case. The case of the Kyoto Protocol is perhaps instructive. A government with foresight will have undertaken a cost-benefit analysis of carbon emission reductions prior to entering into negotiations and agreeing to binding limits. However, once the government has agreed to meet a binding environmental target in the context of such an international agreement, it is likely to be more concerned with undertaking other types of evaluations which cast more light on instrument choice and programme design. The value of undertaking a full cost-benefit analysis once the agreement has been struck is limited unless it feels there is some possibility of either rescinding or reforming the agreement.

Even if one is concerned with a particular element of a programme, there may be alternative methodologies which can be used to evaluate its relative importance and likely effects. For instance, in order to evaluate transaction costs associated with the permit market, it may be appropriate to use statistical evidence to determine price dispersion, semi-structured interviews and surveys of participants

to obtain information on the search and other costs associated with trading, or institutional analyses to assess the bureaucratic hurdles involved in consummating an exchange. All provide useful insights.

For instance, Young (2004) points out that using semi-structured interviews with market participants is very helpful in determining what transaction costs are and even whether participants understand what is being traded. Shabman (2004) examined the characteristics of the programme from an institutional perspective to determine whether or not there was any 'grit' in the market. On the other hand, Ellerman (2004), Kerr (2004) and Wossink (2004) use data on price dispersion of quota prices to evaluate whether or not the 'law of one price' holds in practice. All are valuable exercises, and the evidence is reviewed below.

Similarly, if the concern is with market power, the economics profession provides a host of alternative methodologies for evaluation. Some of these are rather simple reviews of various concentration ratios, such as the Herfindahl Index. Pedersen (2003) looked at concentration ratios in the Danish CO₂ programme. In other cases, more formal econometric work such as event studies, persistence-of-profit studies, and structure-conduct-price studies can be applied. While more formal empirical evidence of this sort is likely to be more illuminating, it is also more data-intensive and more difficult to undertake.

In effect, these insights suggest that evaluators must not only incorporate methodologies that go beyond economics, but also incorporate other parts of economics that may traditionally be less involved in a formal evaluation process (i.e. institutional economics, industrial organization, etc.). Additionally, it may be important to use a combination of such methodologies when undertaking an evaluation. Young (2004) provides examples of the different types of evaluations that have been used to evaluate the Australian water entitlements trading system, including surveys of attitudes of market participants toward the programme, surveys of actual transactions, and comparisons with a wide variety of counterfactuals such as existing government policy, ideal market conditions, and some notion of 'best practice' in a technological sense.

The choice of methodology is also linked to the point at which it is being undertaken in relation to programme implementation. More targeted and less formal evaluations may be appropriate if the results are to be used for mid-term corrections early in the programme's life. The more limited evaluations may be focused on one or a limited number of criteria that are thought to be potentially problematic for the tradeable permit scheme in question. A full evaluation of the programmes as such (i.e. a cost-benefit analysis) may be valuable prior to the introduction of a policy or after a certain number of years, but is unlikely to be helpful soon after implementation since the evidence suggests important learning-by-doing effects are associated with the introduction of tradeable permit schemes.

If a quantitative evaluation is deemed to be appropriate, data availability and financial cost, as well as evaluation criteria and point of implementation will play a role in determining the choice of methodology to be applied. Thus even if a cost-benefit analysis (CBA) is thought to be appropriate, in some cases it may not be feasible for reasons of data availability or financial cost (OECD 1997b). In such cases, it may be necessary to adopt alternative methodologies such as:⁵

- cost-effectiveness analysis
- compliance cost assessments
- risk analyses
- budget analyses (i.e. to determine implications for government budgets)

5. For a full discussion, see Deighton-Smith (1997).

- business impact analyses (i.e. to determine impacts on productivity or competitiveness)
- consequence analysis
- socio-economic analyses (i.e. to determine distributional impacts)

In some sense, all of these can be understood as ‘partial’ CBA’s, distinguished by their comprehensiveness relative to a fully-fledged CBA. For instance, in a cost-effectiveness analysis cost are evaluated (as in a CBA), but benefits are not monetised. In a compliance cost assessment, benefits are left unevaluated altogether. In a budget analysis or a business impact analysis only some elements of the cost side are included (see Viscusi 1997).

Table 1.3 provides an overview of the elements which are ideally to be included using different methodologies. The greatest distinction is, of course, those which focus on potential benefits and those which concentrate on costs. Only cost-benefit analyses and cost-effectiveness analyses address both costs and benefits, with the latter doing so in terms which are incommensurable.

Table 1. 3: Elements Evaluated under Different Quantitative Evaluation Methodologies

	Costs				Benefits		
	Administration	Budgetary	Private (P.E.)	Private (G.E.)	Private	Public Good	Monetised Public Good
Cost-Benefit Analysis	*	*	*	*	*	*	*
Cost-Effectiveness Analysis	*	*	*	*	*	*	
Compliance Cost Analysis	*	*	*				
Budget/Fiscal Impact Analysis	*	*					
Business Impact Analysis			*	*			
Consequence Analysis					*	*	
Risk Analysis					*	*	

P.E. indicates partial equilibrium effects and G.E. indicates general equilibrium effects.

In some senses, evaluation methodologies can be seen in developmental terms, with governments progressing toward the wider application of CBA’s, in which there are greater and greater data collection and analysis requirements. However, given the value of alternative means of evaluation it is important not to make the ‘ideal’ the enemy of the ‘good’, since initially it may be difficult for reasons of data availability, technical capacity or resource constraints to undertake a full CBA (Boyle 1993).

Moreover, when undertaking a CBA a number of simplifying assumptions are frequently made which have the effect of understating the potential importance of some elements of a programme. For instance, on the basis of the evidence reviewed below it appears that transaction costs are a significant factor in many actual tradeable permit schemes. However, it is unlikely that a CBA will include detailed information on the effects of different levels of transaction costs, unless that is the explicit objective of the evaluation.

OECD (1997b) provides an overview of some of the different types of methodologies which have actually been applied for environmental and other policies in different OECD countries in the past. While most countries apply a broad mixture of methodologies depending upon particular circumstances, there are some differences. Some countries (i.e. United States and Canada) placing a strong emphasis on the wide application of CBA’s, while others focus on other types of studies such as compliance cost

assessments (United Kingdom), business impact studies (Australia) or budget/fiscal impact analyses (Spain). However, it must be emphasised, that this is continuously evolving in all countries.

Counterfactuals and Baselines

Accurately defining the counterfactual is key to undertaking a successful ex post evaluation (Smith 2002 and OECD 1997a). There are three related possibilities: a pre-intervention 'business-as-usual' scenario; a counterfactual based upon some alternative policy; and some optimum such as Pareto-optimality. The latter is rarely undertaken in practice for ex post evaluations, although it is common for ex ante evaluations. Moreover, the first two options can be more closely related than might be initially assumed.

In Ellerman's (2004) study of the SO₂ allowance programme, a clear distinction is made between a counterfactual which is based upon a pre-intervention 'business-as-usual' scenario and a counterfactual which is based upon the introduction of an alternative(s) policy. In addition, the interdependence of the programme of interest with other related programmes makes it difficult to disentangle the unique effects of a tradeable permit policy and to draw implications for how the policy might work in a rather different policy environment. (This issue is discussed at greater length below.)

'Backcasting' the technological trajectory that would have taken place in the absence of the introduction of the policy in question (or under the introduction of an alternative policy) is clearly important. For example, Ellerman (2004) points out that the study undertaken by a team at the Massachusetts Institute of Technology (MIT) of the emission reductions achieved by the sulfur allowance programme is based on the assumption that the heat input observed at affected units in each year did not change from the pre-policy rates at those units. This counterfactual assumption has the effect of making the estimated emission reduction insensitive to changes in demand, either at individual units or in the aggregate.

To the extent that other environmental regulations, or changes in relative fuel prices, would have caused the emission rate at affected units to fall during the period of evaluation, the effect of the SO₂ allowance programme would be over-estimated. Indeed, Ellerman (2004) estimates that most of the difference in estimated cost savings between the MIT study and that undertaken by Carlson et al. (2000) can be attributed to differences in assumptions concerning changes in the variable costs of scrubbers, with the latter assuming much higher variable costs than the former.

Similarly, developing counterfactuals about costs involves a considerable amount of subjectivity since they depend directly on secular trends assumed in the counterfactual. Forecasts of trend changes within the affected sectors are, of course, a key component in the results of evaluations. For instance, RECLAIM participants experienced a very large unanticipated demand for power that could only be accommodated by older, more polluting plants. Permit prices soared in a way that was never anticipated (Harrison 2004). Conversely, in the Australian water entitlements trading case, a counterfactual which reflects secular decline in the agricultural sector seems to be more appropriate (Young 2004). Wossink (2004) points out that earlier evaluations of the Dutch nutrient quota programme did not reflect the changes in the livestock sector arising from the Common Agricultural Policy. This is a very different counterfactual than would have been produced by simply extrapolating historic trends.

Policy endogeneity is also important. For instance, the outcome of the SO₂ allowance programme has been heavily influenced by the markedly enhanced rail availability of low sulphur coal from the western United States by rail. Would these events have occurred in the absence of the programme (and therefore should be reflected in the counterfactual) or were they the result of the programme itself (and thus left out of the counterfactual)? For instance, while unlikely, it is possible that the prospect of the introduction of the SO₂ allowance programme played a role in encouraging the deregulation of rail travel, since the benefits of such deregulation increased appreciably with tradeable permits relative to what

would have been the case under the previous direct forms of regulation. Ultimate conclusions about the effectiveness of this programme depend on the answers to such questions.

In light of these difficulties, some evaluation exercises apply many different counterfactuals. For instance, Young (2004) provides examples of the different counterfactuals that have been used to evaluate the Australian water entitlements trading system:

- Comparison with a hypothetical baseline such as a perfectly competitive market;
- Simulation models referenced against a formally defined counterfactual;
- Comparison with a benchmark defined by government policy; and,
- Comparison with industry and other benchmarks of "best" practice.

All of these comparisons can serve useful purposes in and of themselves. However, when used in combination they are likely to give a much richer perspective on the relative merits of a given policy.

The Role of Uncertainty: Natural Risk and Policy Change

Ex post evaluations also have to recognise the role of natural risk (meaning the effects of changes in the environment within which the policy operates) or policy uncertainty (meaning the effects of changes in policy framework itself) in the operation of these markets. Firstly, different programmes may have different implications for the degree of risk to resources or participants, and changes in risk matter in any comprehensive evaluation. Secondly, how well a programme performs may be directly related to the way in which it creates or handles natural risk and policy uncertainty. Evaluations that fail to consider these dimensions will be incomplete, and potentially misleading.

For example, even if the apparent schedule of targets under a tradeable permit system is equivalent to that which would have been the case under direct regulation, the two policies may behave very differently in the face of unexpected changes, such as policy shocks. For instance, in the face of shocks which result in increased emission rates the cap is binding in a way that may not be the case for other policies. This has been particularly true in the RECLAIM programme (Harrison 2004).

In addition, both the New Zealand ITQ and Australian water entitlements trading programmes showed how fundamental uncertainty about the "right" level of the cap can lead to serious problems. For instance, in the New Zealand case (Kerr 2004) a lack of understanding of the biology of the orange roughy led to a cap that permitted unsustainable harvests. In the Australian water entitlements trading case (Young, 2004) excessive withdrawal could cause substantial increases in salinity. However, due to the nature of the programmes, the latter can be attributed to the policy itself, but the former is set externally. This difference should be reflected in any evaluation of the programmes.

Policy uncertainty can play an analogous role. For instance, in the context of the Dutch nutrient quota trading system, policy uncertainties arose for two main reasons: (1) uncertainty over the continuance of the quota system; and (2) uncertainty concerning the introduction of future constraints on quota use. A survey in 1997 showed that Dutch livestock farmers in general perceived policy uncertainty as very significant, and of equivalent importance to the uncertainty from production and markets (Wossink 2004).

This uncertainty can serve as a deterrent on investment, with implications for long-run costs, and as such these adverse effects need to be reflected in any evaluation of the programme. In general, it is widely argued that once the basic policy design has achieved a certain rule stability, it is more important to make the remaining uncertainty transparent and measurable than it is to try to eliminate it entirely.

Bounding the Subject of Evaluation

Tradeable permits are not usually implemented in a vacuum. They frequently complement and interact with other public policies, including other environmental policies. For example:

- The SO₂ allowance programme operates within the more general framework of sulphur oxide regulation established by the National Ambient Air Quality Standards.
- The RECLAIM programme operates within the context of a rather volatile electric deregulation programme in California that has put unusual, but intense pressure on the program.
- The Dutch nutrient quota trading programme operated within the framework of the European Union's Common Agricultural Policy.

The RECLAIM programme provides an interesting example. As noted, soon after the programme was introduced, the electricity supply industry in California was deregulated. This had important implications for the permit market. However, RECLAIM also had impacts upon the electricity sector, with permit prices responsible for a significant proportion of the rise in wholesale electricity prices (Joskow and Kahn 2002). As such, the effects of the two policies can not be separated easily and the inter-linkages between the two must be accounted for in any evaluation undertaken.

Thus, disentangling the effects of the tradeable permit programme from other areas of public policy is not straightforward. This is perhaps even truer when tradeable permits interact with other environmental policies. While Harrison (2004) emphasises that policy makers must be careful not to try to use two policy instruments to achieve one policy objective, in practice they do so quite frequently (see Johnstone 2003 for a discussion). As such, as a practical exercise evaluators are required to evaluate policy packages, not just single policies.

Bounding the limits of a policy can also be important for ecological reasons. Focusing an ex post evaluation only on the resource or pollutant being controlled directly by the tradeable permit system might miss some important spill-over effects. Pollution emissions have technological links (i.e. joint products in generation and abatement) and ecological links (i.e. synergistic effects in the generation of secondary pollutants) which need to be assessed. Similarly, there can be important links between a resource which is directly affected by the tradeable permit programme and other resources (i.e. predator-prey relationships).

In the Australian water case Young (2004) demonstrates the links between water quantity and the environmental damage caused by salinity. Kerr (2004) points out the interdependence between marine species that are controlled by distinct ITQ markets. The RECLAIM market for NO_x co-exists with direct regulations for volatile organic pollutants, a synergistic pollutant in the generation of urban smog. Ignoring the links between these different pollutants and policies would result in fundamentally flawed evaluations.

III. A Review of Ex Post Evaluations of the Impacts of Tradeable Permit System

Earlier OECD reports present a rather detailed list of possible programmatic aspects that can usefully be covered in an ex post evaluation (see OECD 1997a and Smith 2002). In fact, it was found that the actual list of aspects currently covered by ex post evaluations is somewhat smaller. And the information on some topics is invariably more limited than others. Nonetheless some useful and interesting insights emerged from this workshop. These will be addressed under a number of broad headings, following the evaluation criteria set out above.

Economic Efficiency

Ex post evaluations that purportedly tackle the question of economic efficiency typically examine some or all of four rather different concepts: economic (or Pareto) optimality, cost effectiveness relative to what would have been the baseline in the absence of intervention, comparison to some other counterfactual benchmark such as “what would have happened otherwise” under a different policy, and permit market efficiency. Since these are in fact quite different concepts, studies relying on them could come to quite different conclusions, even if they are examining the same programme.

Economic Optimality

The principle of Pareto optimality (the maximisation of net benefits) is a means of assessing whether or not the policy derives all the net benefits from the resource use that are possible. Naturally this requires a comparison of the costs of the programme with all the benefits achieved, including the value of reduced pollution or conserved resources. Conducting this kind of evaluation is time- and information-intensive and, the experience with tradeable permit systems is rare.⁶

An alternative is simply to compare the present value of net benefits for the programme with the net benefits from some predefined alternative policy with the same environmental objective. This is discussed below. However, one important insight that did emerge from the workshop was that while evaluations typically treat the environmental target as predetermined and exogenous, several of the studies seem to indicate that the target may be affected by the choice of the policy instrument (see Ellerman 2004 and Kerr 2004). To the extent this is true the target become endogenous rather than exogenous and this blurs the distinction between studies of economic optimality (in which the target itself is an element for consideration) and cost-effectiveness studies analysis (in which the target is taken as given).

Cost Effectiveness

As noted above, a more common evaluation approach assesses cost-effectiveness, particularly for ex ante studies. This approach typically takes a predefined environmental target as given (such as an emissions cap or a total allowable catch) and examines whether the programme minimises the cost of reaching that target. Another form is to compare the cost of reaching the target with the programme to the cost of reaching the programme with the next most likely alternative policy. This approach, of course, compares the programme not to an optimal benchmark, but rather to some pragmatic benchmark.

While the evidence on environmental consequences is mixed (especially for fisheries), in terms of cost effectiveness the results are somewhat less ambiguous, with the introduction of such schemes often results in lower costs relative to direct regulation alternatives. In the presence of adequate enforcement tradeable permits do appear to increase the value of the resource (in the case of water and fisheries) or lower the cost of compliance (in terms of emissions reduction).⁷

Amongst the case studies presented, the SO₂ allowance programme is the only evaluation which has been subjected to a formal evaluation of this kind. Ellerman (2004) estimates the cost savings relative to

6. None of the studies in this workshop attempt this type of evaluation.

7. It was interesting to note in these case studies that it is not always possible to compare ex ante and ex post cost savings. Harrison (2003) reports, for example that when the RECLAIM programme was being developed, cost savings were estimated to be about 40 percent compared to the cost of achieving the same emission levels using the traditional command-and-control approach. However, no ex post estimates of actual cost savings have been made.

a continuation of existing policies as equal to approximately \$US 350 million in the first years of the programme. He also cites results from ex post evaluations which assume other counterfactuals – such as the introduction of an ‘ideal’ – but highly unrealistic due to information requirements - command-and-control policy. Savings in the latter case are negative.

Other studies have found evidence of gains from trade. In the area of air pollution control, in a number of cases tradeable permit schemes do seem to have generated considerable savings in meeting the pollution control targets (Hahn and Hester 1989, Tietenberg 1990, Smith 2002). The case of water pollution tradeable permit schemes is less clear, with many programmes generating few trades, and thus few cost savings. However, it must be emphasised that even if very few trades are actually undertaken, cost savings can be considerable relative to a policy alternative in which permits are not tradeable (see Woodward 2000).

In the case of water, trading of entitlements has been found to increase welfare by transferring the resources from lower valued to higher valued uses (see Easter et al. 1998). In fisheries, potential benefits include not only the higher profitability from more appropriately scaled capital investments (resulting from the reduction in over-capitalisation), but also from the fact that ITQs frequently make it possible to sell a more valuable product at higher prices (fresh fish rather than frozen fish) (National Research Council Committee). One review of 22 fisheries found that the introduction of ITQs increased wealth in all cases (OECD 1997c, p. 83).

Market Efficiency

A number of evaluations focus on whether the market is efficient or not. In the absence of an initial allocation that happens to mimic the cost-effective allocation, transactions costs and market power can inhibit trade and prevent a market from achieving the target at minimum cost.⁸ Gains from trade are not fully realised. A number of the studies in this workshop (Ellerman 2004 Kerr 2004, Wossink 2004, Young 2004 and Harrison 2004) do examine market efficiency. They use both qualitative and quantitative assessment methodologies to do so.⁹

Interestingly, while many ex ante studies have traditionally focussed on market power, the ex post studies cast more light on transaction costs. This is borne out by the studies reviewed in this workshop, which have rather more to say on transaction costs than they do on market power. The long tradition of modelling market power combined with the fact that suggestive data are available ex ante (i.e. number of players, market share, etc.) may explain the bias for ex ante evaluations toward the analysis of market power, while the theory concerning transaction costs is relatively less interesting, and the evidence of it only emerges once the market commences operation.

Moreover, it appears that transaction costs are, in fact, of considerable importance. For instance, under the original EPA Emissions Trading Programme, it was found that transaction costs could exceed the value of the emission reduction credits (OECD 1997a). Kerr and Maré (1998) estimated that transaction costs associated with participation in the American Lead Phasedown program were sufficient to result in a loss of potential gains from trade in the region of 10%-20%.

A number of the case studies reviewed (Wossink 2004, Shabman 2004 and Young 2004) also find that transaction costs are high. This is explained in part by programme design. However, in many cases it

8. For a theoretical treatment of the role of transactions costs in permit markets see Stavins (1995).

9. Both Wossink (2003) and Young (2003) reveal information of an "anecdotal" nature about transaction costs. Wossink (2003) shows how the nutrient programme explicitly provided support to help participants understand the nature of the market, based upon anecdotal evidence that small farmers were having trouble functioning in the market.

appears that they tend to fall through time. This is due in part to 'learning by doing'. This point was also made by Gangadharan (2000) in a study of the RECLAIM programme. However, falling transaction costs are also due to endogenous institutional changes, such as the growth of secondary markets. In addition, technology is now making an entrance in water markets (the 'Water Links' electronic water exchange in California, for example) to lower transaction costs (OECD 1999a).

Market power can be a particular problem for markets which are spatially-constrained due to the localised nature of the environmental impact. See, for instance, a discussion of market power in a system of transferable development rights in McConnell et al. (2003). For similar reasons this was also a significant concern in many of the water pollution programmes initially introduced in the United States (see Woodward 2000) and is mentioned as a significant concern in wetlands credit trading (Shabman, 2004).

However, the evidence indicates that generally market power has not been a significant issue in most other markets despite some tendencies toward the concentration of quota. In Fisheries this is due in part to accumulation limits that have been placed on quota holders and the fact that these are typically not markets in which accumulation of quota yields significant monopoly-type powers.¹⁰ In fisheries some concern has been expressed (Palsson 1998) that the introduction of ITQs will mean the demise of the smaller fishers as they are bought out by larger operations. The evidence does not seem support this concern.¹¹

Conclusions

The evidence with respect to the economic optimality of tradeable permit systems is limited, due in large part to the fact that most studies take the environmental target as given and focus on cost-effectiveness. Such studies generally indicate that tradeable permit schemes can result in cost savings. However, the functioning of the market is key, with many studies indicating that transaction costs can result in a significant loss in potential gains from trade. Market power – despite being the focus of many ex ante studies – may be less of a concern.

Environmental Effectiveness

The demonstration that the traditional regulatory policy is not generally cost-minimising has two corollaries. It either implies that the same environmental goals could be achieved at lower cost or that better environmental quality could be achieved at the same cost. We have already discussed the issue of costs. In this section we look at its environmental counterpart. As a general point, however, it is worth noting that many of the early air pollution control programmes were designed to exploit the first implication (cost reduction), later programmes attempted to produce better air quality *and* lower cost.¹²

Unarguably, one of the great benefits of tradeable permit systems (at least cap-and-trade schemes) is their ability to deliver a degree of environmental 'certainty' which is not true of other instruments. Since direct regulations (even if expressed in terms of absolute levels of resource use or

10. In many fisheries, for example, the relevant markets are global with many different sources of supply. In air pollution the number of participants is typically quite high.

11. An OECD review concludes "there was very little evidence to support the hypothesis that small scale fishers would be eliminated." (National Research Council Committee, 1999, p. 84)

12. In an interesting analysis of the cost and emissions savings from implementing an emissions trading system for light-duty vehicles in California, (Kling (1994) finds that although the cost savings from implementing an emission trading programme (holding *emissions* constant) would be modest (on the order of 1% to 10%), the emissions savings possibilities (holding *costs* constant) would be much larger (ranging from 7% to 65%).

pollution emissions) are targeted at the level of the individual economic agent, they can not directly deal with changes in levels of economic activity except by means of additional ad hoc constraints. Even relative to taxes, permits provide greater certainty since any uncertainty regarding compliance costs does not affect the attainment of the environmental target. However, there are many senses in which this argument needs to be qualified, and the empirical evidence casts light on just how important these qualifications are in practice.

Meeting the Cap

As noted above, one common belief about tradeable permit programmes is that their environmental effects are secured purely by the imposition of the aggregate limit. Hence, it is believed, the counterfactuals should hold the environmental outcome constant and examine only the costs associated with how that fixed target is reached. This is an oversimplification because several aspects of the programme (including the feasibility of implementing the programme and the magnitude of the cap) may depend on the choice of policy instrument. Any ex post evaluation that fails to capture these endogenous effects will miss an important programme characteristic.

For instance, in air pollution the tradeable permit programmes have typically had a very positive effect on reducing targeted emissions. In both the lead phase out and ozone-depleting gas programmes the targeted pollutants were eliminated, not merely reduced. The SO₂ allowance programme has resulted in substantial reductions in sulphur emissions over time. This is due in part to the fact that the anticipation of lower costs offered by trading were used in initial negotiations to secure more stringent pollution control targets (acid rain program, ozone depleting gases, lead phase-out and RECLAIM) or earlier deadlines (lead phase-out programme) (see OECD 1997a).

Under RECLAIM the air quality effects from more stringent limits were reinforced by the use of adjusted offset ratios for trades in non-attainment areas. Offset ratios were required to be greater than 1.0 (implying a portion of each acquisition would go for improved air quality). In addition environmental groups have been allowed to purchase and retire allowances (SO₂ allowance programme). Retired allowances represent emissions that are authorised, but not emitted. The Dutch nutrient quota programme contained a similar rule (Wossink 2004).

In fisheries the institution of ITQs has sometimes, but not always, resulted in lower (more protective) TACs. In the Netherlands, for example, the plaice quota was cut in half over time (and prices rose to cushion the income shock) (Davidse 1999). What have been the effects on biomass? The evidence has been mixed. In the Chilean squat lobster fishery the exploitable biomass has rebounded from a low of about 15,500 tons (prior to ITQs) to a level in 1998 of between 80,000-100,000 tons (Bernal and Aliaga 1999). The herring fishery in Iceland has experienced a similar rebound (Runolfsson 1999).

On the other hand, one review of 37 ITQ or IQ fisheries, found that 24 experienced at least some temporary declines in stocks after instituting the programmes. These were largely attributed to a combination of inadequate information on which to set conservative TACs, and illegal fishing activity resulting from ineffective enforcement. Interestingly, 20 of the 24 fisheries experiencing declines had additional command-and-control regulations such as closed areas, size/selectivity regulations, trip limits, vessel restrictions, etc. (OECD 1997c). These additional regulations were apparently also ineffective in protecting the resource.

In baseline-and-credit schemes (rather than cap-and-trade schemes) the issue of environmental effectiveness is considerably more tendentious since it is a function of the definition of the baseline and the administrative procedure by which credits are created. There is, therefore significant potential for there to be few environmental benefits, particularly if the technological trajectory is such that environmental impacts are decreasing. Unfortunately, there have not been any systematic ex post evaluations of the environmental effectiveness of baseline-and-credit schemes. However, it has been argued that generous

baseline definitions under the UK Emissions Trading Scheme resulted in an over-allocation of credits, creating 'hot air' (ENDS 2003). A systematic evaluation has yet to be undertaken.

The Endogeneity of Compliance and Enforcement

Regardless of how well any tradeable permit system is designed, non-compliance can prevent the attainment of its economic, social, and environmental objectives. Non-compliance not only makes it more difficult to reach stated goals, it sometimes makes it more difficult to know whether the goals are being met.¹³

Although it is true that any management regime raises enforcement issues, tradeable permit regimes raise some special concerns. One of the most desirable aspects of tradeable permits for resource users - their ability to raise income levels for participants - is a double-edged sword, because it also raises incentives for non-compliance. In the absence of an effective enforcement system, higher profitability could promote illegal activity. Insufficient enforcement could also result in failure to keep a tradeable permit system within its environmental limit.¹⁴

Do enforcement costs rise under tradeable permit programmes? The answer depends both on the level of required enforcement activity (greater levels of enforcement effort obviously cost more) and on the degree to which existing enforcement resources are used more or less efficiently. Higher enforcement costs are not, by themselves, particularly troubling because they can be financed from the enhanced profitability promoted by the tradeable permit system.¹⁵ Indeed, there are many examples in which the rent from transferable permit programmes are used to finance superior monitoring and enforcement systems.

In the American SO₂ allowance programme, for example, the environmental community demanded (and received) a requirement that continuous emission monitoring be installed (and financed) by every covered utility. Coupling this with the rather stringent penalty system has meant 100% compliance. In the Danish CO₂ system (Pedersen 2003), which does not rely on continuous emission monitoring, the electricity producers pay an administration fee of 0.079 DKK per tonne of CO₂ allowance to the DEA to cover the administration costs (verification of emissions, distribution of allowances, operation of the registry, monitoring of trading, development of the scheme etc.).

The rents generated by ITQs in fisheries have also provided the government with a source of revenue to cover the costs of enforcement and administration. In many of the ITQ fisheries in Australia, Canada, Iceland, and New Zealand, the industry pays for administration and enforcement with fees levied on quota owners. Not all uses of tradeable permits, however, offer as convincing a solution for the

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13. In fisheries, for example, stock assessments sometimes depend on the size and composition of the catch. If the composition of the landed harvest is unrepresentative of the actual harvest due to illegal discards, this can bias the stock assessment and the total allowable catch that depends upon it. Not only would true mortality rates be much higher than apparent mortality rates, but the age and size distribution of landed catch would be different from the size distribution of the initial harvest (prior to discards). This is known in fisheries as "data fouling".
 14. Prior to 1988, the expected positive effects of ITQs did not materialize in the Dutch cutter fisheries due to inadequate enforcement. Fleet capacity increased further, the race for fish continued, and the quotas had to be supplemented by input controls such as a limit on days at sea (National Research Council Committee, 1999).
 15. Not only has the recovery of monitoring and enforcement costs become standard practice in some fisheries (New Zealand, for example), but funding at least some monitoring and enforcement activity out of rents generated by the fishery has already been included as a provision in the most recent amendments to the US Magnuson-Stevens Act. The sulphur allowance programme mandates continuous emissions monitoring financed by the emitting sources.

monitoring and enforcement problems. With respect to fisheries one comprehensive earlier review found that “higher enforcement costs and/or greater enforcement problems occurred in 18 fisheries compared to five that experienced improvements. Enforcement proved particularly difficult in the high value fisheries, in multi-species fisheries, and in transnational fisheries” (OECD 1997c).

In addition to the obvious potential for quota busting that all tradeable permit approaches face, fisheries can also face problems with poaching (harvests by ineligible fishermen), unreported high-grading (discarding low valued fish to make room in the quota for higher valued fish) and by-catch discards (non-targeted species caught and discarded).

Whether these problems are intensified or diminished by the implementation of a tradeable permit programme depends (in part) on the economic incentives confronting participants. The incentives for high-grading, for example, depend on the magnitude of price differentials for various types and sizes of targeted species. As the price premium for fish of a particular size and type increases, the incentive to use quota for especially valuable fish increases along with the incentive to discard less valuable fish (Anderson 1994). On the other hand, implementing an ITQ regime may favour some technologies over others, with ambiguous implications for by-catch and high-grading.

As Shabman (2004) points out, reviews of the wetlands credit trading programme have identified several specific failures to secure the programme’s “no net loss” objective. Some reviews have found that the ecological functions, especially for wildlife and habitat, of avoided wetlands and on-site wetlands offsets are compromised by polluted runoff and adverse changes in hydrologic regimes. In some cases ecological failure resulted from poor construction techniques. In other cases, a promised restoration project may not have been undertaken at all. In general the failure to prevent these compromises to the programme could be traced back to limited agency resources available for enforcement.

However, Shabman (2004) does report one clear environmental benefit of wetlands credit trading in terms of enforcement incentives, at least relative to the system of in-lieu fees which it supplements. The introduction of private firms in the market lead to greater demands for ‘quality assurance’ on the part of regulatory authorities, and thus tighter enforcement of the ‘no net loss’ rule which underpins EPA’s policy on wetlands.

The Endogeneity of Monitoring

Closely related to issues of compliance and enforcement, is the issue of monitoring. Reliable ex post evaluation of tradeable permit programmes depends on the quality as well as the quantity of data related to emissions, resource use, etc. In order to assure the accuracy of reported data, it is sometimes necessary to build a number of safeguards into the programme.

In fisheries, proper control procedures include both onshore and at-sea components. An onshore system of checks would normally include a requirement that sales only be made to registered buyers and that both buyers and quota shareholders co-sign the landing entries. These measures create an audit trail that can be electronically monitored for an instance in which a comparison of processed product weight and recorded purchases suggests suspiciously high product recovery rates. The at-sea component would include both on-board observers, where the fishery is profitable enough to bear the cost, and random checks at sea by the appropriate authority (or perhaps by video monitoring). In the absence of these measures the catch data could be contaminated by unreported off-shore discards and inferences drawn from them in ex post evaluations could be biased.

In theory, these requirements should not differ according to instrument choice. However, monitoring technology may also be endogenous to the choice of instruments. Indeed, tradeable permits seem to have resulted in greater demand for accurate monitoring. Moreover, with technological change in monitoring costs such as the continuous emission monitoring and web-base reporting in the SO₂

allowance programme (Ellerman 2004), these expenditures may come down further. Therefore, an interesting question for ex post evaluation is the degree to which technological progress in monitoring and enforcement is exogenous and the extent to which it is promoted by tradeable permit schemes. To the extent that it is endogenous, ex post evaluation schemes that treat it as exogenous will be biased.

One characteristic of some of the tradeable permit programmes, for example, has been the development of computer-based hardware and software. Key to a smoothly implemented tradeable programme is ensuring that all data are input to an integrated computer system that is accessible by eligible users on a real-time basis. Such a system would not only provide up-to-date information on permit use to both users and enforcement agencies, it would also provide a wealth of data for ex post evaluations. It would ideally also allow short-notice transfers, such as when a vessel heading for shore has a larger than expected by-catch and needs to acquire additional quota for the by-catch species before landing. Facilitating this kind of flexibility would reduce the enforcement burden considerably by giving permit holders a legal alternative to illegal discarding without jeopardising the objectives of the program.

The computer system should also provide easy data entry. Card swipe systems, such as those used in the Alaska halibut and sablefish ITQ fisheries, automatically input all the necessary identification data so that only landings (and hence permit use) need to be recorded. It is also possible to have the harvest level recorded directly from the scales (with appropriate adjustments for "ice and slime" or the degree to which the fish are already processed). Entry terminals that are connected to the master computer system should be available at all authorised landing sites.

Technology has also played an important role in monitoring of the SO₂ allowance programme (Kruger et al. 1999). Both the collection and dissemination of the information derived from the continuous emissions monitors is now handled via the web. Special software has been developed to take individual inputs and to generate information both for the public and for EPA enforcement activities. According to Kruger et al. (1999) the development of this technology has increased administrative efficiency, lowered transactions costs and provided greater environmental accountability.

And finally, one rather unexpected point that emerges from ex post evaluation is the degree to which the number of errors in pre-existing emission registries are brought to light by the need to create accurate registries for tradeable permit schemes (Wossink 2004, Pedersen 2003 and Hartridge 2003).¹⁶ Although inadequate inventories plague all quantity-based approaches, tradeable permits seem particularly effective at bringing them to light and providing incentives for emitters to help to improve the quality of the inventories.

Spatial Differentiation of Impacts

The myth perpetuated by the theory of tradeable permit markets is that the commodity being traded is homogeneous. However, the case studies show that the commodity is frequently not homogeneous - or rather the environmental damages associated with the commodity are not homogeneous - and this lack of homogeneity has to be taken into account in ex post evaluations. For instance, since marginal damages can differ depending upon the place or time of emission, equalisation of marginal abatement costs across sources would not be economically optimal. Any ex post evaluation which assumes that marginal abatement cost equalisation is optimal is effectively discounting the importance of such differences and will provide misleading estimates of the benefits.

One source of heterogeneity stems from spatial considerations, specifically the fact that for some types of tradeable permits systems the location of the emission release or resource use matters (Tietenberg

16. This was also true in the tradeable permit scheme set up to control pollution in Chile. See Montero et al. (2002)

1995). Theory typically treats trades as if they affect only the cost of compliance, not the environmental consequences. Any cost-effectiveness analysis that does not account for the heterogeneity may be defining “effectiveness” incorrectly. In effect, since the environmental outcome will depend upon the pattern of trade, it may not be possible to hold the ‘benefit’ side constant when undertaking comparisons with different instruments.

Temporal differentiation of environmental impacts are also important. In some cases the time of emission is relatively unimportant. This is certainly true of greenhouse gases. It is less true of sulphur dioxide emissions, but temporal impacts are still relatively unimportant and for this reason, banking of permits is allowed within the SO₂ allowance programme. However, urban smog – the target of the RECLAIM programme – is a flow problem, and it is partly for this reason that only very limited ‘banking’ is allowed, through overlapping denomination of the permits. Irrespective, if the time of emission has important consequences for the level of environmental damages, this must be addressed in any evaluation in two ways. On the one hand, on the benefits side the evaluation should not value damages equally. On the other hand, on the cost side the evaluation should examine the implications of constraints on permit trade and / or use.

These issues are addressed in different ways in different programmes. One strategy used in US air pollution control policy to resolve the spatial externality problem is regulatory tiering. Regulatory tiering implies applying more than one regulatory regime at a time. Sulphur oxide pollution in the United States is controlled both by the regulations designed to achieve local ambient air quality standards as well as by the SO₂ allowance trading program. All transactions have to satisfy *both* programmes. Thus *trading* is not restricted by spatial considerations (national trades are possible), but the *use* of acquired allowances is subject to local regulations protecting human health via the ambient standards. The second regulatory tier protects against the harmful spatial clustering of emissions (by disallowing any specific trades that would violate the standards), while the first tier allows unrestricted trading of allowances. Because the reductions in sulphur dioxide are so large and most local ambient standards are not likely to be jeopardised by trades, few trades have been affected by this provision. Yet its very existence serves to allay fears that local air quality could be in jeopardy (see Johnstone 2003).

Unfortunately, the case studies do not provide solid evidence to assess the efficacy of alternative means of addressing spatial heterogeneity of impacts within tradeable permit systems. Ellerman (2004) does find that degradation in local conditions has not arisen in the SO₂ case despite some fears, but it must be noted that the programme involved such a large reduction in aggregate emissions that such an outcome would have required a particularly skewed pattern of trade. Nonetheless, Ellerman (2004) does forward arguments to support the view that for economic reasons the pattern of trade tends toward environmentally-beneficial directions.

The case of RECLAIM (Harrison 2004) is perhaps more interesting since in this case spatial issues were dealt with within the trading system itself. The RECLAIM programme restricts sales from the coastal zone to the inland zone for this very reason (Harrison 2004). Unfortunately, the trade data is not disaggregated spatially in a way which would allow for the evaluation of whether or not the constraints on trade have played a significant role in preventing the deterioration of local environmental conditions.

In the Dutch nutrient quota trading programme the transfer of phosphate quota was allowed within regions and from a surplus region into a deficit region, but prohibited from a deficit region into a surplus region. In addition to phosphate quota, farmers willing to expand animal production in the surplus region had to acquire ammonia rights (Wossink 2004). Trade in ammonia rights was only allowed within a county and hence was even more spatially restricted than trading in phosphate quota. In the American wetlands credit trading programme, spatial constraints were also placed on trade in an effort to ensure comparability of environmental outcomes (Shabman, 2004).

Ancillary Impacts and Effects on Other Resources

As noted above, evaluations of tradeable permit programmes must take “external” effects into account. The resource controlled by the permit programme is frequently not the only resource affected. For instance, in water trading programmes, one significant problem has been the protection of “instream” uses of water (Young 2004). In the United States some states only protected private entitlements to water if it was diverted from the stream and consumed. Recent changes in policy and some legal determinations have afforded more protection to these environmental uses of water.

In air pollution control several effects transcend the normal boundaries of the program. Devlin and Grafton (1994) illustrate how the use of tradeable permits for one pollutant can result in technological change which will have implications for any other pollutants which are joint products. These effects can be negative or positive. Under any tradeable permit systems for climate change program it is widely recognised that the control of greenhouse gases will result in substantial reductions of other pollutants as a side effect (see Ekins 1996, OECD 2000, Hartridge 2003).

As noted above, in fisheries the two main effects have been by-catch and high-grading. By-catch is a problem in many fisheries, regardless of the means of control. The evidence from fisheries on how the introduction of ITQs affect by-catch and high-grading is apparently mixed. However, there is clearly some potential for ITQs to result in changes in fleet technologies, perhaps due to concentration of quota holdings amongst larger vessels. Since by-catch and high-grading are largely a function of gear choices, the introduction of ITQs can result in significant changes in by-catch and discard rates, whether positive or negative. Two reviews found that by-catch and high-grading may either increase or decrease in ITQ fisheries depending on the fishery (OECD 1997c, National Research Council Committee, 1999).

Conclusion

Thus, the evidence suggests that treating environmental effects as exogenous in ex post evaluations is an oversimplification for several reasons. First, whether it is politically possible to set an aggregate limit may be a function of the policy used to achieve it. Second, both the magnitude of that limit and its evolution over time may be related to the policy. Third, the choice of policy regime may affect the level of monitoring and enforcement and non-compliance can undermine the achievements of the limit. Fourth the policy may trigger environmental effects that are not covered by the limit.

Administration Costs

Ex post evaluations should examine not only private compliance costs, but public administrative costs as well. Few published case studies shed much light on administrative costs. Smith (2002) gives some evidence for Michigan’s VOC trading programme. OECD (1997a) provides similar information on the CFC phase-out. However, the case studies in the workshop provide more evidence in this area. They demonstrate how the amount and nature of public administration tasks change with the adoption of a tradeable permits approach.

One recurring theme during the workshop suggests that the administration of tradeable permit systems not only involves fewer administrative person-hours (McLean 2003), but also results in significant changes in bureaucratic functions performed (McLean 2003, Harrison 2004). The rather different types of tasks required of administrators in a tradeable permit system have rather definite implications for the nature of the skills required by administrators. Administrators who can monitor and enforce compliance replace engineers who seek: (1) to identify the correct control strategies for sources to adopt and (2) to negotiate permit exemptions. Incidentally, this further strengthens the view that monitoring, compliance and enforcement should be considered to be endogenous to instrument choice.

The studies also point out that the *type* of tradeable permits system seems to affect administrative costs. Credit-based programmes, such as the Emissions Trading System in the United Kingdom (Hartridge 2003) must necessarily keep a large element of the previous administrative infrastructure in place. In addition, baselines must be determined often at the level of the plant. Programmes with regulatory pre-approval (i.e. wetlands credits and water trading) have even greater implications for administrative costs, since each individual transaction implies an administrative burden for the regulatory authorities.

In addition, other specific design features, such as the opt-in in the SO₂ allowance programme (Ellerman 2004) and the use of relative targets in the UK's emissions trading scheme (Hartridge 2003) also add considerably to administration costs. Since the design features vary so much from programme type to programme type, it is difficult to generalise insights about administrative costs across programmes. However, it does seem that cap-and-trade systems may have higher start-up costs (due to intensive lobbying), but baseline-and-credit schemes have higher running costs. Evaluations are likely to improve our information on how these design features influence administrative costs.

In addition, ex post evaluation also has begun to reveal how tradeable permits, particularly cap-and-trade permit systems, change the fundamental nature of regulation (Ellerman 2004, Harrison 2004, McLean 2003). With tradeable permits bureaucrats are no longer in charge of defining the appropriate way to meet the goal. Rather they are in charge of assuring that the private firm manages to meet the goal.

Soft Effects

In some cases tradeable permit systems may have effects on how firms manage environmental concerns internally. For instance, discussions at the workshop revealed a substantial amount of anecdotal evidence about how tradeable permit programmes are changing the manner in which environmental risk is being treated within firms (Hartridge 2003, McLean 2003). Historically the environmental risk manager in most firms has not been involved in the most fundamental decisions about product design, production processes, selection of inputs etc. Rather the risk manager was simply confronted with decisions concerning product design, etc. made elsewhere within the firm, and his/her responsibility was merely to ensure that the firm was in compliance with environmental regulations. Relegating risk management to the tail end of the process prevents the incorporation of one major avenue of risk reduction - pollution prevention (see Johnstone 2001).

Because tradeable permits put both a cap and a price on environmental risks, creating markets for these permits provides an incentive for financial officers to become involved in the management of environmental risk. Some evidence on the endogeneity of management systems to instrument choice is provided and is discussed in the case studies, as well as more broadly in the literature. For instance, in a review on the effects of the SO₂ allowance trading programme undertaken by researchers at the Environmental Law Institute, interviews with representatives from affected firms indicated that with the introduction of the program there had been a shift in responsibility for environmental matters from engineers in dedicated environmental departments to financial officers in more central locations within the firm (see Swift, 2001).

The introduction of ITQs in fisheries has had implications for compensation systems for crew, processors and fishing communities. Traditionally in many fisheries crew have co-venturers in the fishing enterprise, sharing in both the risk and reward. In some cases the shift to ITQs has shifted the risk and ultimately shifted the compensation system from a share of profits system to a wage system. Though this has not necessarily lowered incomes, it has changed the culture of fishing (McCay et al. 1989, McCay and Creed 1990). However, the evidence on the extent of organisational and management changes that might be initiated by tradeable permits garnered at this workshop should be treated more as a hypothesis to be tested than a firm result. As such it provides an interesting target for further ex post evaluations that begin to inquire about how policy reforms affect decision-making within organisations.

On the negative side, the case studies in this workshop also point out that participants frequently require some experience with the programme before they fully understand (and behave effectively) in the market for permits. This suggests that particularly in the earlier years, ex post evaluation of market efficiency can be crucial not only in understanding the impediments to smoothly operating markets, but also to how those barriers could be reduced (Pedersen 2003, Hartridge 2003, Wossink 2004, and Young 2004). There may be a lag in the development of appropriate management responses.

This is consistent with earlier results. For instance, Kerr and Maré (1998) found that 'first trade' costs were very high in the market for lead permits. Firms took some time to learn how to operate in the market, and this resulted in lost gains from trade in the early stages of the market (Smith 2002). If, as these studies suggest, a private "learning by doing" effect is associated with the tradeable permit market itself, this would affect the optimal timing of ex post evaluations. Early evaluations would be helpful in promoting learning, but not in evaluating ultimate success or failure of the programme since participants will still be high up on their learning curve.

The ex post evaluations have also revealed the degree to which inefficient rent-seeking by firms can be affected by the introduction of tradeable permit systems. Discussions among participants suggested that the choice of policy instruments could affect the degree to which rent-seeking activity takes place. In one sense the apparent transparency and lack of administrative discretion that characterises smoothly operating tradeable permit regimes discourages firms from lobbying for changes in regulations at the level of the individual plant or firm. Changes are worked out in the market, not the bureaucracy. This view is also supported by McConnell et al (2003) who found that rent-seeking behaviour fell once a system of transferable development rights was introduced since the benefits of lobbying for zoning variances were lost. The same point has been made by Sunstein (1988) and Ackerman and Stewart (1991) in reviews of the early American experience with tradeable permit systems.

On this point participants drew a sharp distinction with traditional environmental permit systems (based on performance or technology standards). In traditional direct regulation, environmental permitting inevitably involves a significant amount of negotiation between the firm and the regulatory agency no matter how uniformly applied the system may appear to be in principle. The case studies also suggest, however, that a different form of rent-seeking can, and does, take place in tradeable permits systems. For instance:

- Under a cap-and-trade system firms will lobby for a favourable initial allocation of permits (Ellerman 2004, Harrison, 2004).
- Under any baseline-and-credit system firms will lobby ex ante to ensure that they receive credit for any undertaken reductions (whether they are "additional" to the base case or not) (Shabman 2004, Hartridge 2003); and
- Under any tradeable permits system that requires regulatory pre-approval of trades firms will lobby for approval (Shabman 2004, Wossink, 2004, Young 2004).

The overarching conclusion from the case studies seems to be that the amount of rent-seeking in tradeable permit systems is positively related to the amount of administrative intervention in the operating systems.

Dynamic Effects and Temporal Efficiency

One of the major theoretical expectations concerning tradeable permits is that they will promote more environmentally-beneficial technological progress. Given the central role of technological change in addressing environmental problems, it is important to test this hypothesis, difficult though it may be to do

so. Most obviously, the difficulties associated with defining a technological counterfactual have been discussed above.

The case studies provide ambiguous support for technological change being promoted, though the examples all fall far short of a ringing endorsement of the strong expectation generated by the theory. For instance, Wossink (2004) identifies some changes in livestock feeding practices that came into being after the introduction of the Dutch nutrient quota program. However, these may be more attributable to regulations on manure, than to the quota system. Ellerman (2004) reveals how fuel mixing technologies and newer, less expensive sulphur scrubbers have appeared in the SO₂ allowance programme. This work complements new insights that are emerging elsewhere in the literature suggesting that technological progress is endogenous (Carlson et al 2000, Popp 2001, and Kerr 1997).

Dynamic efficiency is not, however, merely a function of technological change. General issues of temporal efficiency are key. For instance, Kerr and Newell (2001) found that the American lead phasedown allowed high-cost firms to avoid the loss of sunk costs through delayed technology adoption, while low-cost compliance firms moved early. The case studies also reveal the importance of banking schemes in providing firms with temporal flexibility. Existing tradeable permit schemes differ considerably in how they treat banking and/or the role of forward markets. And the message that emerges from these studies is that this temporal flexibility can be quite important.

Ellerman (2004) discusses the considerable role that both banking and forward markets have played in the SO₂ allowance programme. Harrison (2004) shows how the price spikes in the RECLAIM programme (which does not have banking) were probably intensified by the absence of this flexibility, but also how helpful even the limited temporal flexibility provided by the programme through overlapping permit validity. Pedersen (2003) also mentions the importance of temporal flexibility for climate change investment in the Danish case.

The introduction of an ITQ system may also result in different patterns of entry and exit than would have been the case otherwise, with the inevitable result that the composition of resource users changes. When the New Zealand ITQ system was introduced consolidation occurred, with many "artisanal" fishers getting out of the sector altogether. While this had no appreciable effect on market concentration, it may have led to changes in fishing technology. In particular since by-catch and discards are subject to imperfect monitoring, technological change in the fleet may trigger environmental implications that either improve or worsen the situation. By providing artisanal fishermen a secure asset that they could not previously transfer and liquidate, ITQ fisheries may have directly encouraged artisanal exit and therefore indirectly affected the fishing technologies in use.

The Dutch nutrient quota system also affected composition of resource users. Wossink (2004) suggests that many of the sales of quotas in the "surplus" region were from farmers who left farming altogether or shifted to the "deficit" area. This rather large turnover in quota holders could potentially have quite large environmental consequences, not only in terms of where the quotas were being used, but also in terms of the change in practices that could result from the change in ownership.

And finally, dynamic issues associated with the scheduling of targets were discussed. Based on her analysis of the Dutch system, Wossink (2004) suggests that tradeable permits work best when the targets are not too stringent. On the other hand, if we are to derive lessons from the operation of the market, permits must not be sufficiently lax either. Scarcity is necessary to provide adequate incentives for emitters or resource users to seek alternatives to the status quo (through changes in technology or trading).

Defining a sufficiently stringent cap to ensure that scarcity arises has not always been the case. In RECLAIM, for example, the programme was imparting little information of value to market participants for the first few years due to the relative laxity of the cap. The desire to implement the programme led to a very generous definition of the cap in the earlier years. Clearly some balance must be established between

the desire to gain political support for the programme and the desire to demonstrate rapid progress toward environmental goals.

IV. Conclusions

In their most successful applications tradeable permits have been able to protect environmental resources at relatively low cost. However, such programmes remain relatively few in number. Nonetheless, their use is now sufficiently widespread to allow for some general conclusions on their potential uses, as well as the role of evaluation in ensuring that their potential benefits are maximised.

Placing Tradeable Permits in their Historical Context

The record revealed by the literature on tradeable permits seems to indicate that until quite recently resorting to a tradeable permits approach usually only occurred after other, more familiar, approaches had been tried and failed. In essence the costs of implementing a new system with which policy administrators have little personal experience are typically perceived as incurring such large costs that they can only be justified when the benefits have risen sufficiently to justify the transition (Libecap 1990).

For instance, most fisheries managers that have turned to these policies have done so only after a host of alternative input and output controls have failed to stem the pressure being placed upon the resource. A similar story can be told for air pollution control. The offset air pollution control policy, introduced in the US during the 1970s, owes its birth to an inability of any other policy to reconcile the desire to allow economic growth with the desire to improve the quality of the air.

As such, tradeable permits have often been used in areas in which other instruments have not performed as well as had been hoped. The RECLAIM case is a perfect example. On the one hand, this means that tradeable permits are often being used to address some of the most problematic environmental areas, which should be borne in mind when assessing their relative merits. On the other hand, this also means that the counterfactual against which they are often being assessed is frequently one which has been recognised as a failure. This, too, should be borne in mind.

It is also clear from the historical record that not every attempt to implement a tradeable permit approach has been successful. In air pollution control initial attempts to establish a tradeable permits approach failed in Poland (Zylicz 1999) and Germany (Scharer 1999). Initial attempts also failed in United Kingdom (Sorrell 1999). In a general sense, programs targeting water pollution control have generally not been very successful (Hahn and Hester 1989). On the other hand it does appear that the introduction of new Tradeable permit programmes becomes easier with familiarity. Learning-by-doing exists within individual programmes, as well as across programmes through time. In the United States, following the very successful lead phase out programme new supporters appeared and made it possible to implement the SO₂ allowance programme.¹⁷

The design features of the programmes are not stable over time; they evolve with experience. The earliest use of the tradeable permit concept in the United States, the emissions trading programme, overlaid credit trading on an existing regulatory regime and was designed to facilitate implementation of that programme. Trading baselines were determined on the basis of previously determined, technology-

17. It is frequently suggested that new programmes should be of the “cap and trade” type because they reduce transaction costs. While they may reduce transactions costs, it is less clear that “cap and trade” programmes can always achieve the political will to be implemented without gaining familiarity though the more heavily controlled credit programmes.

based standards and created credits could not be used to satisfy all of these standards. For some the requisite technology had to be installed.

More recent programmes, such as the SO₂ allowance and RECLAIM programmes largely replace, rather than complement, traditional regulation. Allowance allocations for these programmes were not based on pre-existing technology-based standards. In the case of RECLAIM the control authority (the South Coast Air Quality Management District) could not have based allowances on predetermined standards even if they had been inclined to do so. Defining a complete set of technologies which offered the necessary environmental improvement (and yet were feasible in both an economic and engineering sense) proved impossible. Traditional regulation was incapable of providing the degree of reduction required by the Clean Air Act.

Reviewing Tradeable Permits Relative to Evaluation Criteria

The lessons obtained from the case studies of the literature review support many of the conclusions reached in previous OECD work in this area (see Smith 2002 and OECD 1997a), but provide important new insights in a number of areas. For instance, the results do seem to indicate that existing tradeable permit systems - even some of those whose design has been clearly imperfect - have resulted in significant gains from trade. The costs of meeting given environmental objectives can be reduced through increased flexibility, and if the programme is well-designed these gains can be considerable.

The case studies provided new and interesting evidence in the area of market efficiency. Using a variety of different methodologies, a number of the studies found that transaction costs can be quite high and that this can affect potential gains from trade. This is an area where ex post evaluations are particularly helpful. Perhaps more significantly, such studies also provide evidence on the types of factors which are likely to lead to higher transaction costs. In particular, regulatory pre-approval of trades stands out as one important barrier. However, more general issues associated with the complexity of administrative procedures may be equally important.

In addition, it appears that the administrative costs from introducing tradeable permits are not excessive and may be considerably less than under alternative forms of regulation. That said, the case studies do reveal a clear distinction between cap-and-trade schemes and baseline-and-credit schemes in this area. While the former may have relatively higher start-up costs, they are likely to result in significant savings in terms of administrative costs over the longer term. Thus, a number of the case studies and the more general secondary literature provided support for the view that the administrative costs associated with tradeable permit systems are no greater, and perhaps considerably less, than those associated with other policy instruments.

Moreover, the studies also revealed that the nature of administrative burdens have changed with less emphasis on administrative procedures which have uncertain effects on environmental outcomes (eg. negotiating permit exemptions). This can have important spin-off benefits, such as improved enforcement. Perhaps more significantly, tradeable permit schemes reduce the potential benefits associated with rent-seeking behaviour. Once again, however, a distinction between cap-and-trade and baseline-and-credit schemes must be emphasised since the latter can even result in increased incentives for rent-seeking behaviour relative to direct forms of regulation.

The widespread belief in the environmental effectiveness of tradeable permits is also generally supported by the case studies. Placing an absolute limit on emission levels or resource use is important, particularly in the face of uncertain economic conditions (i.e. marginal abatement costs or future economic growth rates). Moreover, this has played a key role in quelling the fears of those who are sceptical about the potential merits of tradeable permit systems. The extent to which tradeable permits can address spatial and temporal heterogeneity of environmental impacts in an efficient manner is clearly an important area in which further policy evaluations are required.

However, the case studies have cast new light on the relative merits of tradeable permit systems in terms of environmental effectiveness in a broader sense. It would appear, that the level of the cap, the accuracy of data for monitoring, and the degree of compliance and enforcement are endogenous. Thus, ex post evaluations which seek to compare 'like with like' by assuming equivalent environmental outcomes for different instruments may be misleading. In general, all of these factors strengthen the case for the use of tradeable permit systems, certainly in relation to direct forms of regulation.

Perhaps most interestingly, the case studies also provided new information in the area of 'soft effects' associated with the introduction of tradeable permits. It appears that it takes time for permit market participants to become accustomed to trading in the market, and even to fully understand the nature of the commodity which is being traded. At the early stages of policy implementation this can result in thin markets, price volatility and other phenomena which can undermine the development of the market. In addition, the 'soft' effects of tradeable permit schemes on internal management systems within firms is also an area worthy of further analysis.

While the theoretical case for the important positive role of tradeable permits with respect to innovation effects is very strong, the empirical evidence remains limited. The most that can be said is that there is some evidence of increased diffusion of available technologies, and more efficient use of existing technologies. However, this in itself may be revealing, since the greatest benefits of tradeable permits in the early stages of their implementation may be due to the relaxation of regulatory constraints which have been preventing the application of simple, but more efficient, technologies which are readily available.

A more general factor which emerged clearly from the case studies is the potentially adverse consequences of placing severe restrictions on their operation as a way to quell administrative fears about undesirable and unforeseen outcomes. As Shabman (2004) points out this is precisely the state the US wetlands credit programme is currently experiencing. Although with increased familiarity (and comfort) these restrictions tend to disappear over time, they do tend to severely diminish the early accomplishments of the programmes, and in some cases may so cripple it that it never evolves into a smoothly operating program.

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Chapter 2

EX POST EVALUATION OF THE RECLAIM EMISSIONS TRADING PROGRAMMES FOR THE LOS ANGELES AIR BASIN¹

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I. Introduction

The emissions trading programme developed for the Los Angeles air basin—the Regional Clean Air Incentives Market (“RECLAIM”)—provides perhaps the most complex experience thus far of any established emissions trading programmes. Begun in 1994 after a three-year development effort, the RECLAIM cap-and-trade programmes for nitrogen oxides (“NO_x”) and sulfur dioxide (“SO₂”) include participants from numerous sectors (in contrast to most other existing trading programmes, which focus on a single sector). Several project-based programmes that provide credits for reductions from mobile and area sources supplement the cap-and-trade programmes. Indeed, this basic structure is similar to that envisioned for other emissions trading programmes including those for greenhouse gas (“GHG”) emissions, in which trading under national or regional caps would be supplemented by project-based programmes (Joint Implementation and Clean Development Mechanism). Notwithstanding important differences between RECLAIM and other potential programmes, the experience with RECLAIM promises to provide some insights into how other trading programmes might function.

The NO_x RECLAIM programme has had a tempestuous history. The significant developments include:

- In the summer of 2000, the prices for NO_x Reclaim Trading Credits (“RTCs,” the basic unit of trade) increased dramatically. From a range of about \$1,000-\$4,000 per ton in 1994-99, the average RTC price increased to more than \$45,000 in 2000 (with individual trades reported for more than \$100,000 per ton).

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1. Parts of this paper draw upon a joint paper with Denny Ellerman and Paul Joskow (Ellerman, Joskow and Harrison 2003). I am indebted to both co-authors for their insights as well as to Nick Johnstone and other participants on the Workshop on Ex Post Evaluation of Tradable Permits organized by the Organization for Economic Cooperation and Development and Sue Lieu and Jill Wynot of the South Coast Air Quality Management District for helpful comments on a previous draft and to Warren Herold and James Patchett of NERA for research assistance. In the past, I have served as a consultant regarding RECLAIM to the South Coast Air Quality Management District, the California Council for Environmental and Economic Balance, and the Regulatory Flexibility Group. Despite these debts and affiliations, all views expressed by are mine and I alone am responsible for any errors or omissions this paper might contain.
 2. The views expressed in this chapter are those of the author and do not necessarily reflect those of the institutes with which the author is affiliated or the OECD.

- This increase in RECLAIM NO_x prices both was a reflection of the “California electricity crisis” of 2000—which involved dramatic increases in wholesale electricity spot prices—and was itself a significant contributing factor to the higher wholesale electricity spot prices.
- As a result of the high NO_x prices and the electricity circumstances, NO_x emissions exceeded the RECLAIM cap for 2000 by about 6 percent (after taking advantage of limited banking/borrowing options). RECLAIM provisions call for these excess emissions to be reflected in reduced future RTC allocations.
- In May 2001, the agency responsible for RECLAIM, the South Coast Air Quality Management District (“SCAQMD”) passed major changes to RECLAIM that have in effect suspended participation in it by electricity generators and returned the control of electricity generators’ emissions at least temporarily to a command-and-control programme. Generators submit compliance plans for the installation of best available retrofit control technology (“BARCT”) and pay a mitigation fee of \$7.50 per pound (\$15,000 per ton) for excess emissions that is used by the SCAQMD to fund project-based emission reductions; generators’ future RTC allocations are reduced if excess emissions are not made up for by the SCAQMD.
- Electricity generators have complied with the May 2001 requirements by submitting compliance plans and paying mitigation fees for excess emissions in 2000 and 2001. The SCAQMD has contracted for emission reductions from marine and other sources.
- RTC prices have declined considerably since their peaks in 2001, so that by the end of 2002, RTC prices for NO_x 2003 RTCs were in the neighbourhood of \$6,000 per ton.

These experiences can suggest lessons for how emissions trading works “in practice,” particularly with regard the treatment of price volatility and the interaction of emissions trading and restructuring of electricity markets. In considering such lessons, however, it is important to compare RECLAIM’s performance not only to an “ideal” emissions trading programme but also to the alternative “command-and-control” programme that otherwise would exist.

The key conclusions regarding the recent RECLAIM performance are the following:

1. *With regard to environmental performance, RECLAIM has generally been successful in achieving its emission goals; even for 2000, the net effect of the dramatic shifts in RTC prices was to shift a relatively small percentage (6 percent) of NO_x emission reductions from 2000 to future years.* Moreover, there is no reason to believe that a command-and-control alternative would have performed better under the circumstances. Indeed, since emission *rates* would have been regulated—rather than overall emissions—the command-and-control alternative would likely have resulted in the same or greater emissions increases without the compensating measures taken as a result of exceeding the NO_x RECLAIM cap.
2. *With regard to cost savings, the large number of RTC transactions suggests that trading has reduced the overall cost of meeting emissions targets; indeed, even in 2000, the RECLAIM market behaved as a market should.* As the demand for NO_x RTCs increased and their supply decreased in 2000, NO_x prices increased as they should have. Moreover, the prices of an important product “using” NO_x RTCs, wholesale electricity, also increased. This should have provided signals to affected sources to invest in emissions controls as well as signals to consumers to reduce consumption of electricity.

3. *The major “problems” of RECLAIM were due primarily to flaws in California’s newly deregulated electricity markets rather than to serious flaws in the RECLAIM programme itself.* Had the structure of California’s electricity industry remained composed of regulated monopolies, or if the transition to new competitive market structures had been done more competently and with greater sensitivity to the interactions between NO_x RTC prices and spot electricity prices, the impact of higher NO_x RTC prices on the average price of electricity would have been much more modest. And if the electricity crisis had not occurred, electricity generators likely would not have been removed—at least temporarily—from the NO_x RECLAIM programme.
4. *RECLAIM could be improved by providing more tools to manage extreme price volatility effectively.* The experience of RECLAIM in 2000 in 2001 shows that short-term and unexpected fluctuations in demand can lead to significant volatility in allowance prices, which can in turn lead to significant volatility in prices of goods whose production involves significant “use” of emissions allowances. The major tool used in other programmes to reduce price volatility—banking of excess emissions reductions—is largely unavailable under RECLAIM. Other options include expanding the range of emission-reduction alternatives in RECLAIM—beyond the relatively few additional emission reduction alternatives made available from project-based credits or voluntary opt-ins thus far—and maintaining the mitigation fee or developing a “safety valve” that would directly limit the maximum price.

II. Overview of RECLAIM and Initial Experience

Development of the RECLAIM Programme

Regulators in the Los Angeles air basin were developing RECLAIM in the early 1990s at the same time that the national Acid Rain Programme was being developed for electricity generators (see Chapter 3). RECLAIM was significant both in some of its provisions and as the first major example of a tradable permit programme developed by a local jurisdiction, rather than a federal authority.

The SCAQMD approved the RECLAIM programme in October 1993 after a three-year development programme, and the programme began operation in January 1994 (South Coast Air Quality Management District 1993). RECLAIM was developed as an alternative and less expensive means of achieving the emission reductions of nitrogen oxides (NO_x) and SO₂ called for by a set of command-and-control measures designed to bring the Los Angeles Basin into compliance with National Ambient Air Quality Standards. The Los Angeles Basin consists of a 6,600 square mile area with a combined population of about 13 million inhabitants in 1990. The SCAQMD had struggled with the task of complying with federal air quality mandates for many years. At the time that efforts began to develop an emissions trading programme, air emissions were governed by the 1989 Air Quality Management Plan (AQMP). The 1989 AQMD was a massive plan—consisting of 130 individual control measures affecting every sector in the region—that was designed to bring the area into compliance with federal air quality standards by 2010. The cost of the control strategy was enormous, estimated to be about \$13 billion (1988 dollars) per year, or about \$2,200 annually for every household in the region (Harrison 1988).

Interest in emissions trading was due in large part to a desire to develop a less expensive means to achieve air quality targets and to avoid incentives for businesses to leave the region. The SCAQMD devoted substantial resources to the development and evaluation of the RECLAIM programme; prominent activities included hosting workshops, producing many documents including a full-scale feasibility study, and setting up an Advisory Committee to assist in the development of details for the programme that became RECLAIM.³ Studies done at the time indicated that RECLAIM could reduce

3. The author was a member of the RECLAIM Advisory Committee.

the costs of meeting emission targets by about 40 percent relative to the command-and-control alternatives in the AQMP (Harrison and Nichols 1993, Johnson and Pekelney 1996).

Key Features of RECLAIM

The major elements of RECLAIM were cap-and-trade programmes for major stationary source emitters of nitrogen oxides (NO_x) and sulfur dioxide (SO₂). (The SCAQMD had considered developing a cap-and-trade programme for volatile organic compounds (“VOCs”), but decided not to do so.⁴) Sources subject to the cap—which were issued RECLAIM Trading Credits (“RTCs”)—included those emitting more than 4 tons per year, although there are provisions for additional sources to opt-in to the programme. RECLAIM also allowed sources to gain additional RTCs from reductions in emissions from mobile sources, primarily through scrapping existing motor vehicles.

Under RECLAIM, the caps for both NO_x and SO₂ were set higher than expected emissions in the initial years—which reflected in part the effects of a recession in the Los Angeles region—but the overall caps were reduced steadily over time so that by 2003, emissions from the covered sources would be reduced by about 50 percent below early-1990s emission levels and about 80 percent below allowable emissions. From 2003 on, the caps remain constant. The final caps were equivalent to the levels of emissions expected under the set of command-and-control regulations that would otherwise have applied to these sources.

Several features of the RECLAIM programme distinguish it from other emissions trading programmes. First, the programme covers a heterogeneous group of participants including power plants, refineries, cement factories, and other industrial sources. Second, because of new source review concerns, the RECLAIM programme distinguishes between emissions in two geographic zones.⁵ Since emissions in the Los Angeles Basin generally drift inland from the coast, sources located in the inland zone are allowed to purchase RTCs from facilities in either the inland or coastal zones, but sources located in the coastal zone can only purchase RTCs issued for facilities in the coastal zone. A third distinctive feature of the RECLAIM programme is that it does not allow banking because of concerns that use of banked emissions might lead to substantial increases in some future year, and thus delay compliance with ambient air quality standards.⁶ RECLAIM does provide limited temporal flexibility, however, by grouping sources into two 12-month reporting periods, one from January through December and the other from July through June, and by allowing trading between sources in overlapping periods. This provision effectively allowed the programme as a whole to bank or borrow RTCs six months before or after a given compliance period. (Individual participants can purchase RTCs in any vintage and thus effectively bank or borrow across any period.)

The initial allocation of RTCs was the most contentious part of the planning process, although eventually an allocation plan acceptable to the wide range of affected facilities was developed (Harrison 1999a). As was the case with the Acid Rain Programme, RTCs were allocated free to

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4. VOCs were not included in RECLAIM apparently because of concerns among environmental groups that toxics exposure might be affected and because of concerns among industry groups that cost saving would be relatively modest and not justify the setting of a cap on overall emissions. See Harrison 1999b.
 5. The original RECLAIM proposal included 38 separate trading regions, corresponding to the regions used for the offset programme. This detailed geographic division was abandoned as a result of the plausible fear that the trading markets would be too thin. See Harrison (1999b).
 6. RECLAIM also requires facilities to “true up” their emissions—that is, show that they have RTCs equal to their emissions—every quarter as a means of avoiding being in a deficit position at the end of the compliance period.

incumbents and distributed many years prior to when they could be used for compliance.⁷ The final set of formulas for allocating RTCs departed considerably from the simple formula initially proposed by the SCAQMD, and it was the result of literally dozens of proposals, many of which were exhaustively studied by the SCAQMD (and no doubt by the affected firms as well). Despite threats by several firms and sectors to oppose the programme if their formulas were not chosen, the final result was an administratively feasible and politically salable cap-and-trade programme.

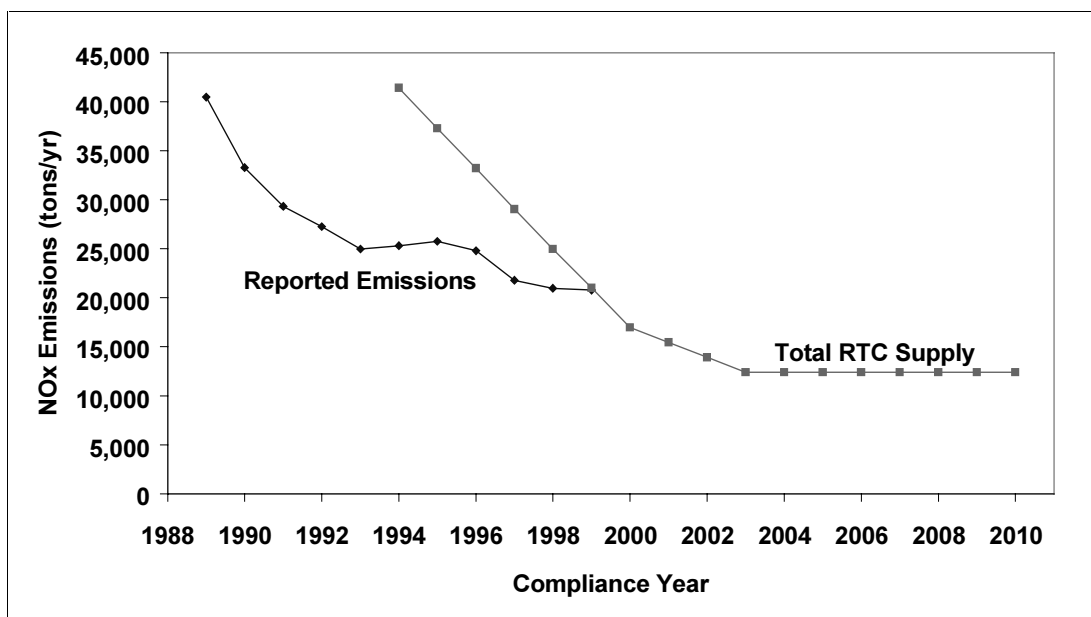
Initial Experience with RECLAIM

The experience with RECLAIM over the first six years, from 1994 to 1999, was relatively uneventful and positive—emissions caps were not exceeded, brokerage and other institutions developed to facilitate trades, some initial monitoring difficulties were worked out, and large volumes of RTCs were traded. There were, however, anticipations of future difficulties.

Emissions and Caps

Emissions for both NO_x and SO₂ RECLAIM sources were below the annual caps throughout the period from 1994 to 1999, although the 1999 NO_x emissions were virtually equal to the cap. Figures 2.1 and 2 show emissions and the annual caps for NO_x and SO₂, respectively.

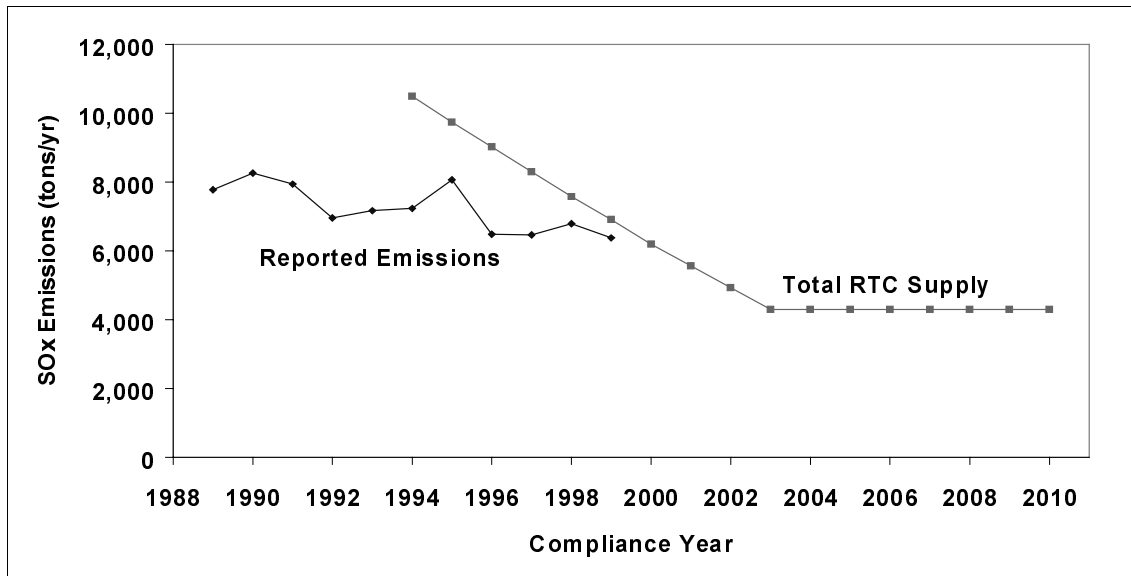
Figure 2.1. NO_x Emissions and Available RTCs (1994-1999)



Source: South Coast Air Quality Management District (2001c).

7. SO₂ allowances in the Acid Rain Programme are distributed for thirty years forward on a rolling basis, while RTCs are distributed indefinitely into the future.

Figure 2. 2. SO_x Emissions and Available RTCs (1994-1999)



Source: South Coast Air Quality Management District (2001c).

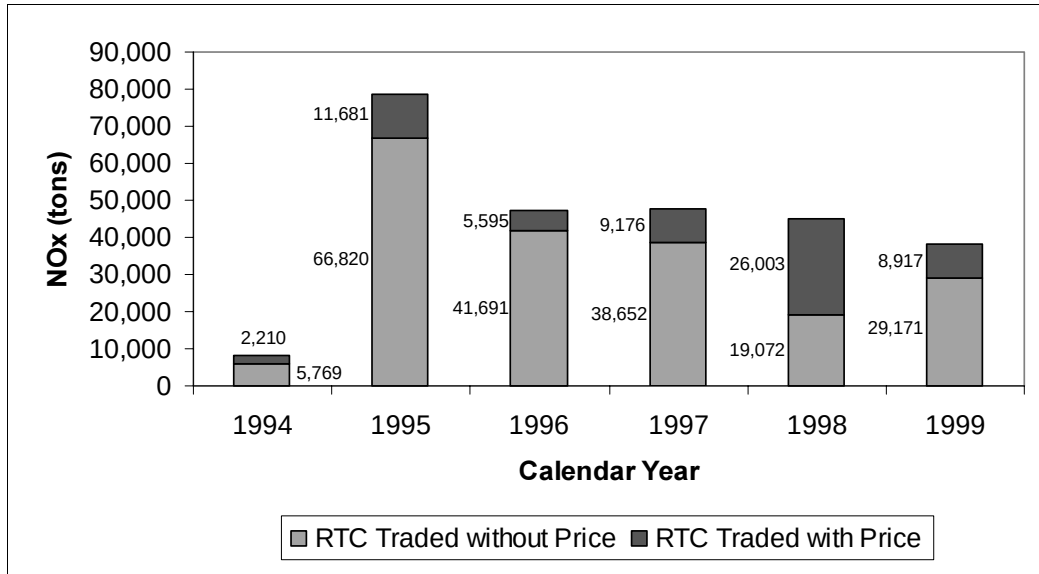
These figures also suggest that the relatively lenient caps in the early years of RECLAIM did not result in emissions increases. Some critics of RECLAIM have complained about the lax early caps, although it is important to remember that many of the command-and-control regulations that RECLAIM replaced had not gone into force when RECLAIM began. Indeed, the emphasis in the 1989 AQMP was on compliance with ozone requirements in 2003—an important milestone for showing federal compliance—which may explain why the final cap levels were set for 2003.

The number of sources changed somewhat over time, although the changes in RTCs were relatively modest. **Error! Reference source not found.** shows inclusions, exclusions and shutdowns over the period from 1994 to 1999. Shutdowns do not affect the RTC cap.

Trading Volumes and Compliance Costs

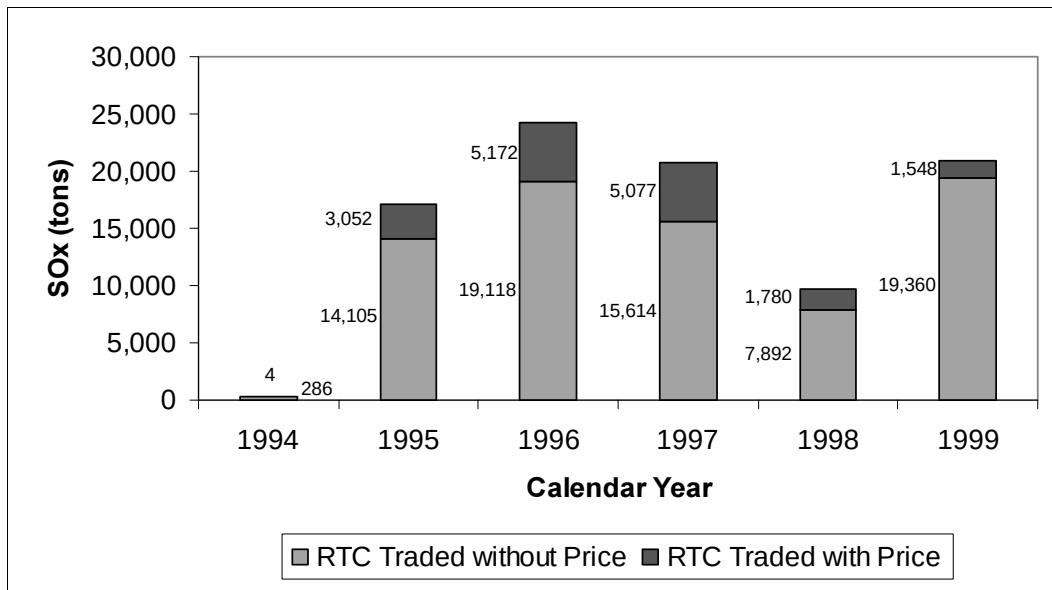
Figures 2.3 and 2.4 show the volumes of trading in SO₂ and NO_x RTCs, respectively, over the first six years of RECLAIM. The figures are reported separately for RTCs without a price—which includes transfers from sellers to brokers, transfers within facilities with common ownership, and transfers as part of other transactions (e.g., sale of a facility including its RTCs)—as well as the number of RTCs traded between firms at various prices.

Figure 2.3. Total Tons of NO_x RTCs Traded (1994-1999)



Source: South Coast Air Quality Management District (2002a).

Figure 2.4. Total Tons of SO_x RTCs Traded (1994-1999)



Source:

South Coast Air Quality Management District (2002a).

The figures show the total numbers of RTCs of all vintages “traded” in each calendar year; because the figures are not reported for individual vintages, it is not possible to compare total volumes of trades with total RTCs for the current compliance year. The fact that the volume of trade in a given trading year typically exceeds the current compliance year RTCs implies that firms were trading various vintages. As of the end of 1999, RTC permits for about 265,000 tons of NO_x and about 93,000 tons of SO₂ had been traded. Since the aggregate NO_x and SO₂ caps were not binding in some years and because the volumes traded in virtually every year exceed that year’s cap (often by several multiples), the presumption is that most of these trades are in future vintages. Moreover, the trend of

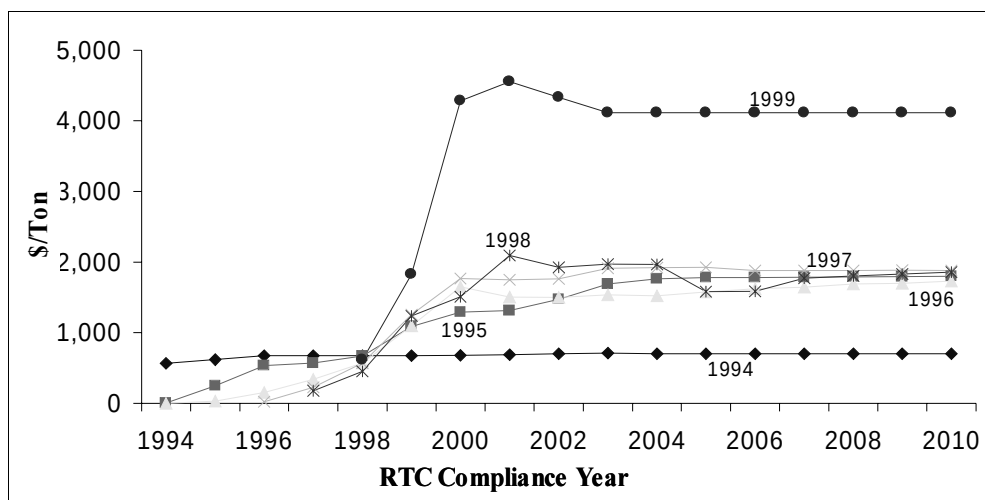
less trading over time, especially for NO_x, suggests that future vintages were bought, sold, and transferred ahead of time, in keeping with plans to install the required abatement equipment to meet the final cap in 2003 and thereafter.

These conclusions regarding the volume of RTC transactions must, however, be tempered by complexities in the reporting of RTC trades. As Burnside and Eichenbaum (1996) point out in an early review of RECLAIM, there is considerable double counting of trades because RTCs are recorded as sold for a zero price when they are transferred to a broker for possible sale, and recorded again when they are sold or returned to the seller if no buyer is found.

RTC Prices

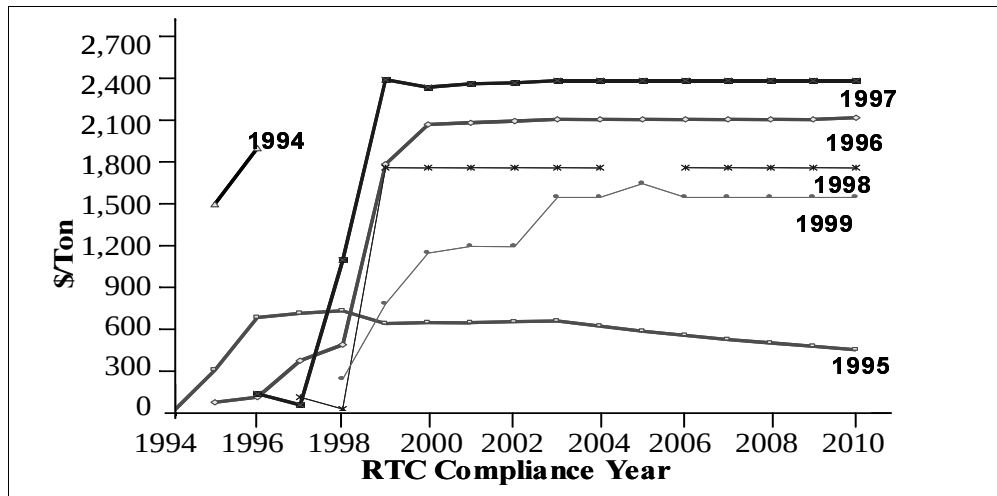
The average prices for RTCs over the six-year period from 1994 to 1999 are provided in Figure 2.5 and 6 for NO_x and SO₂, respectively. The graphs are complicated because they summarize prices for different vintage RTCs in the various calendar years. The horizontal axis in Figure 2.5 represents the vintage, i.e., the year in which the RTC can be used, and the lines in the graph show the average prices for current and future vintages in successive calendar years. The curve labelled 1999, for example, shows the average price for transactions in calendar year 1999. This curve shows that the price in 1999 of the current year vintage (1999 vintage year) was substantially lower than the prices of future vintages (2000, 2001, 2002, etc.), which are about equal. In addition, between 1994 and 1998, the prices of the NO_x RTCs for vintages 2000 and beyond remained relatively stable, ranging from \$1,500 to about \$2,000 per ton.

Figure 2.5. Yearly Average Prices by Vintage for NO_x RTCs (1994-1999)



Source: South Coast Air Quality Management District (2000).

Figure 2.6. Yearly Average Prices by Vintage for SO_x RTCs (1994-1999)



Source: South Coast Air Quality Management District (2000).

Prices for SO₂ RTCs are generally substantially lower than for NO_x RTCs and show a different pattern. Prices generally have fallen over time, presumably in line with expectations about the likely marginal costs of reducing SO₂ emissions from RECLAIM sources. Note, however, that the prices for SO₂ RTCs are substantially higher than for national SO₂ allowances under the acid rain trading programme (see Chapter 3), which have generally been in the range of \$100-\$200 per ton, about one-tenth of the value for SO₂ RTCs. This difference reflects the more costly controls required to comply with the SO₂ cap in RECLAIM compared to the SO₂ cap in the acid rain trading programme.

Transaction and Monitoring Costs

Numerous brokers and other intermediaries have emerged to facilitate these trades and to provide other services to participants, such as pricing information and some derivatives to manage price risk. The current RECLAIM website lists more than 100 individuals who have registered to engage in RTC sales, many of whom are affiliated with brokerage firms rather than RECLAIM facilities. Transaction costs for RTCs appear to have been relatively low and there can be no doubt that RECLAIM markets have been active.

Like the SO₂ cap-and-trade programme, RECLAIM required that the largest sources use CEMs to verify their emissions as a means of providing assurance that the data were valid. Table 2.1 summarises the monitoring requirements for RECLAIM sources. When the programme first was implemented, there were technical difficulties with some of the CEMs, and some facilities could not rely upon CEMS data for all of their submissions (South Coast Air Quality Management District 1998). (The early problems with CEMs occurred because of technical malfunctions of the CEMs equipment; see Burnside and Eichenbaum 1996.) These difficulties (and the higher costs of CEMs) prompted requests that the large sources be allowed to use the less expensive monitoring options allowed for smaller sources, a request that was denied by the SCAQMD because of concerns expressed by the U.S. Environmental Protection Agency ("EPA"). Eventually the technical difficulties with these CEMs were overcome and virtually all of the emissions data from large sources now are based upon the CEMS information.

Table 2.1. Monitoring Requirements for RECLAIM Sources

<i>Source Category</i>	<i>Major Sources (NO_x and SO_x)</i>	<i>Large Sources (NO_x only)</i>	<i>Process Units and Rule 219 Equipment (NO_x and SO_x)</i>
Monitoring Method	Continuous Emission Monitoring System (CEMS)	Fuel Meter or Continuous Process Monitoring System (CPMS)	Fuel Meter and/or Timer
Reporting Frequency	Daily	Monthly	Quarterly

Source: South Coast Air Quality Management District (2002a).

III. More Recent Experience with RECLAIM

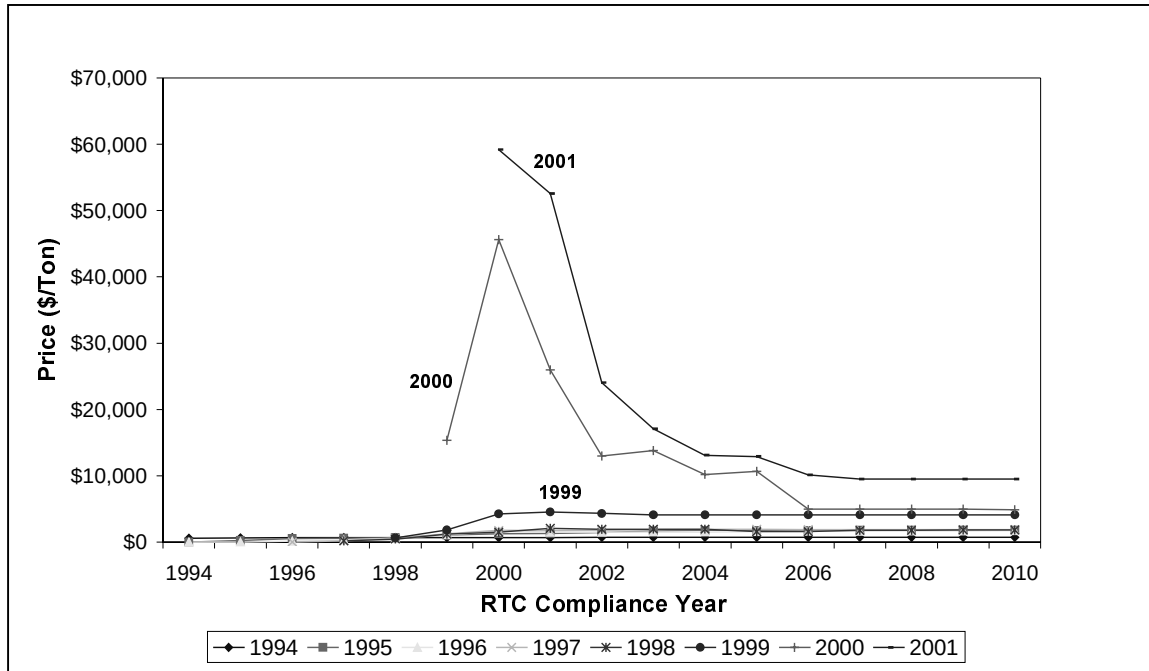
The difficulties experienced by the RECLAIM programme began in the summer of 2000 as an implication of restructuring of California's electricity sector and culminated in May 2001 in significant changes in RECLAIM, including at least a temporary exclusion of electricity generators from RECLAIM and introduction of command-and-control requirements.

RECLAIM Price Spikes

Price volatility manifested itself most visibly in the dramatic spikes in 2000 in the prices of NO_x RTCs for 2000 vintages and those surrounding it. This trend continued into 2001, until the electricity generators were removed (at least temporarily) from RECLAIM.

Figure 2.6 shows average NO_x prices for various vintages over the history of the programme from 1994 to 2001. In 2000, the prices for all "near term" vintages of allowances jumped significantly (the points on the top line labelled "2000" in the figure), with the largest price increase exhibited for the 2000 vintage allowances, tapering off quickly for later vintages. (Note that the price for the 1999 vintage allowance also increased, because RTCs from the 1999 July cycle could be traded - used to cover emissions through June 2000.) The price for 2000 NO_x RTCs increased from an average of about \$4,300 per ton for trades in 1999 to almost \$45,000 per ton for trades in 2000, a ten-fold increase. The average monthly price of 2000 vintage NO_x RTCs reached in the peak month in 2000 was more than \$70,000 per ton, with the highest single price reported equal to more than \$100,000 per ton. The price increases—in 2000, relative to 1999—for 2001 and later vintage allowances (the points on the top line) were smaller and taper off for farther-out vintages. The increase in NO_x RTC prices continued into 2001, with the same pattern as 2000 regarding the much larger increases for near-term (2000-2002) vintages relative to later vintages, whose prices would be limited by the costs of emission reduction alternatives.

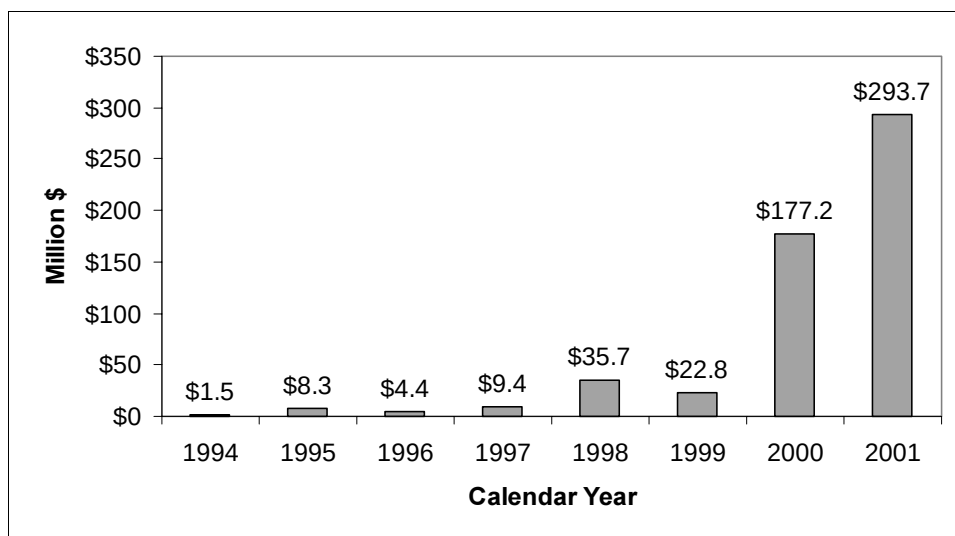
Figure 2.6. Yearly Average Prices for NO_x RTCs (1994-2001)



Source: South Coast Air Quality Management District (2002a).

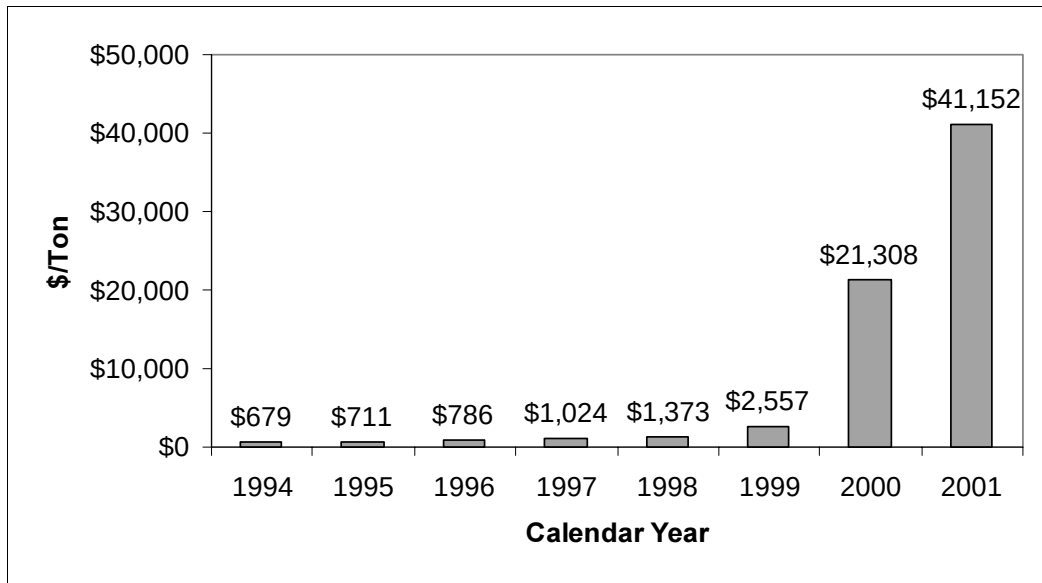
The dramatic increase in NO_x RTC prices in 2000 and 2001 was reflected in the overall dollar of trading volume and the average value for NO_x RTCs. Figure 2.7 shows the total value for NO_x RTC trades by trading (calendar) year. The total volume of NO_x RTC transactions increased from a high of \$35.7 million in the 1994-99 period, to \$177.2 million in 2000 and \$293.7 million in 2001. The average value per ton for all NO_x RTC transactions ranged from around \$1,000-\$2,500 in the 1994-99 period, to more than \$21,000 in 2000 and more than \$41,000 in 2001. (Note that these average values reflect transactions for all RTC vintages.)

Figure 2.7. Total Value of NO_x RTC Trades (1994-2001)



Source: South Coast Air Quality Management District (2002a).

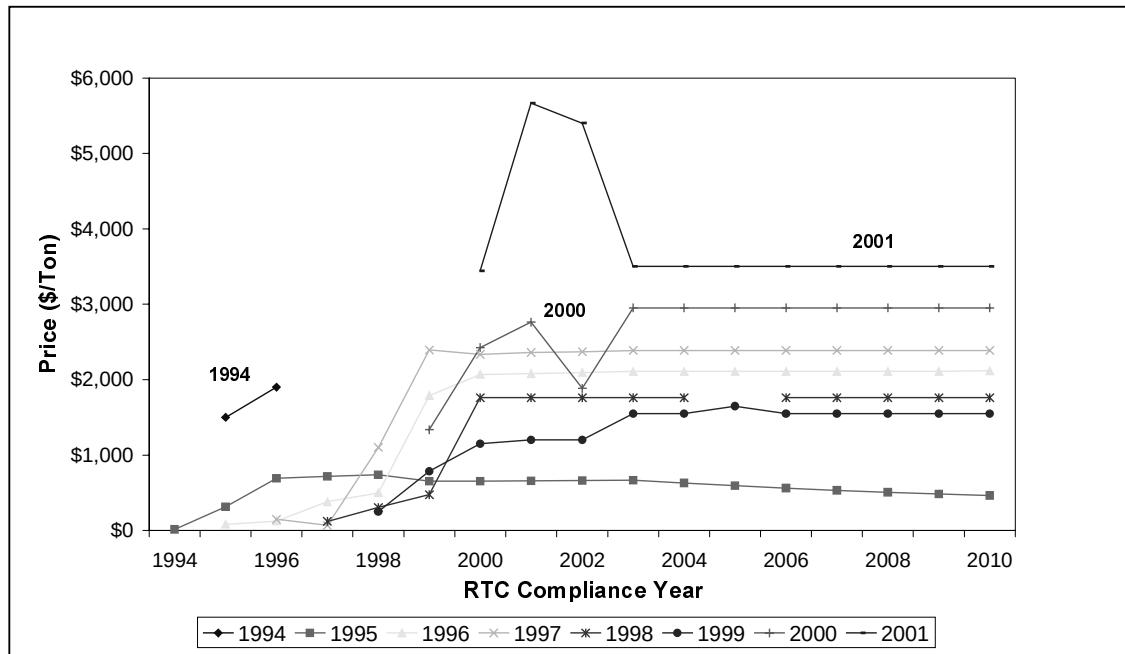
Figure 2.8. Average Price per Ton for NO_x RTC Trades (All Vintages) (1994-2001)



Source: South Coast Air Quality Management District (2002a).

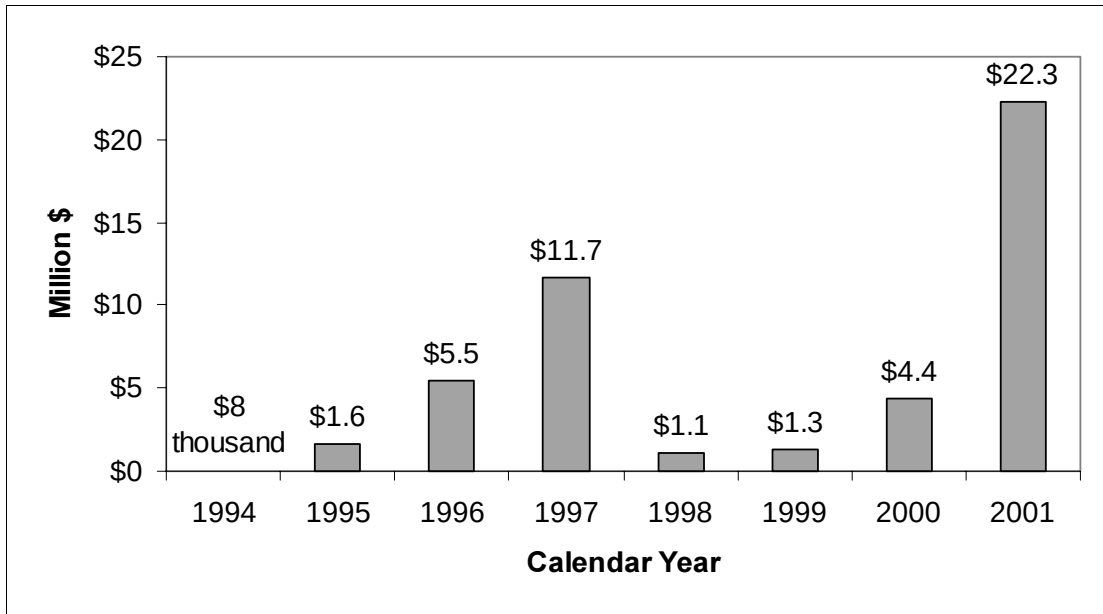
The increases were much less pronounced for SO₂ RTCs than for NO_x RTCs. Figure 2.9 shows the price history for SO₂ RTCs. Prices increased substantially, particularly in 2001, although the increases were much smaller than for NO_x RTCs. Indeed, as Figures 2.10 and 2.11 indicate, the total value of SO₂ trades and the average price per SO₂ RTC were actually greater in 1997 than in 2000, although the values were greatest in 2001.

Figure 2.9. Yearly Average Prices for SO_x RTCs (all vintages) (1994-2001)



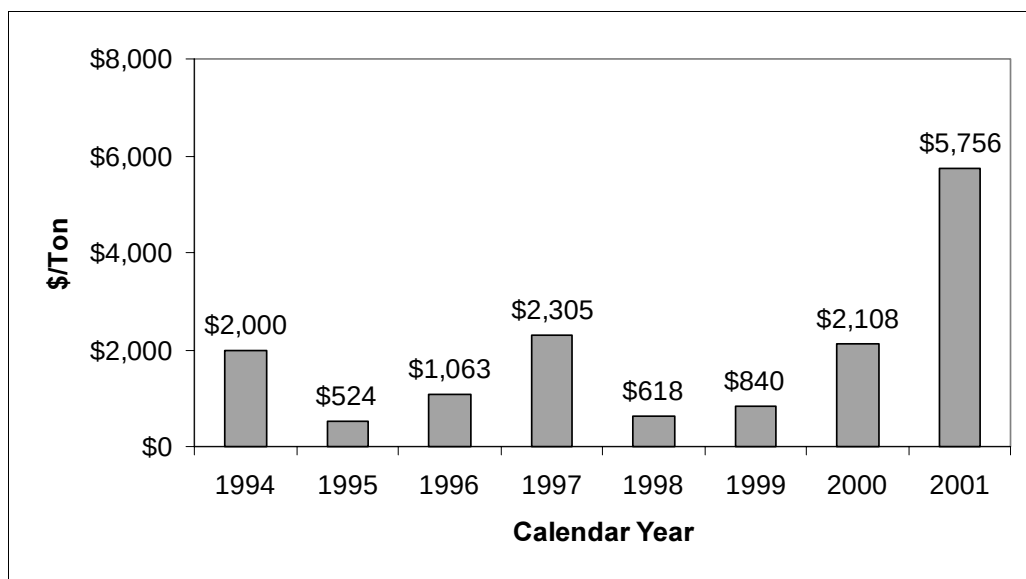
Source: South Coast Air Quality Management District (2002a).

Figure 2.10. Total Value of SO_x RTC Trades (all vintages) (1994-2001)



Source: South Coast Air Quality Management District (2002a).

Figure 2.11. Average Value of SO_x RTC Trades (all vintages) (1994-2001)



Source: South Coast Air Quality Management District (2002a).

Interactions Between RECLAIM Market and California Wholesale Electricity Market

The dramatic increase in the cost of NO_x RTCs in the summer of 2000 was caused by a substantial increase in the demand for RTCs on the part of electric generators. The demand for electricity soared in California during the summer of 2000 while the availability of imported power from other states declined (Joskow 2001). The increased demand had to be met by running the large fleet of old in-state gas-fired generating units more intensively than in the recent past. Few of these old plants had yet

installed NO_x emissions controls and no new plants were completed until the summer of 2001. As a result, the demand for NO_x RTCs and their prices increased significantly during summer 2000 as generation from the in-state gas fired power plants increased to balance supply and demand.

The high price for NO_x RTCs was one of several factors leading to the high wholesale electricity prices in California during 2000 (see Joskow and Kahn 2002 and Borenstein et al. 2002). The generating units subject to NO_x RECLAIM requirements became the marginal units for a significant number of hours, thus setting prices for wholesale electricity throughout California. Moreover, a large fraction of electricity demand under California's new wholesale market institutions was supplied through the spot market, where demand was effectively insensitive to prices, rather than through long-term contracts. As a result, the huge spikes in spot market prices for electricity, caused in part by the spike in NO_x RTC prices, became an enormous financial burden on distribution utilities, consumers, and the state of California.

There has been speculation that unregulated electricity suppliers manipulated the NO_x RTC credit market as well as the electricity and natural gas markets. Indeed, there is some preliminary evidence that electricity generators used the NO_x RTC credit market to earn greater profits in the electricity market (Kolstad 2002) and there is currently substantial litigation regarding the behaviour of electricity and natural gas traders during this period. It seems too early to develop clear conclusions on these issues of market power and market manipulation.

What does seem clear, however, is that the inadequacies of the California electricity restructuring contributed substantially—perhaps definitely—to the dramatic increase in NO_x RTC prices (see Joskow 2002). Had the structure of California's electricity industry remained composed of regulated monopolies, or if the transition to new competitive market structures had been done more competently and with greater sensitivity to the interactions between NO_x RTC prices and spot electricity prices, the impact of higher NO_x RTC prices on the average price of electricity would have been much more modest.

This is the case because regulated prices would have been based on the average costs of generating electricity, including the average costs of NO_x RTC credits rather than the marginal cost of the generating units with the highest marginal spot fuel and RTC costs that cleared the spot electricity market. In addition further price increases resulting from unregulated generators exercising market power (Joskow and Kahn 2002, Borenstein, Bushnell and Wolak 2002) would not have occurred under regulation. Finally, as Joskow (2001) points out, the disruptions caused by divestiture, the hasty movement to a set of complex new wholesale market institutions, inelastic short-term electricity demand, and the excessive amount of electricity demand supplied out of the spot market, probably undermined rational forward contracting and investments in NO_x controls by the new owners of most of the fossil-fired power plants in California and shifted the burden of short-term price volatility to economic agents (distribution utilities and retail consumers) who did not have the ability to manage it.

RECLAIM NO_x Emissions

The dramatic increases in demand for NO_x RTCs—leading to price spikes in 2000—also were reflected in emissions above the NO_x cap set for 2000. The excess emissions were due to excess emissions both from power facilities and non-power facilities. Table 2.2 summarises NO_x emissions and allocations for Compliance Year 2000, the last year for which information is available. Overall, compliance year 2000 emissions were in excess of Compliance Year 2000 allocations by about 3,300 tons, or about 19 percent of the Compliance Year 2000 allocation. (Figure 2.12 shows these results graphically.)

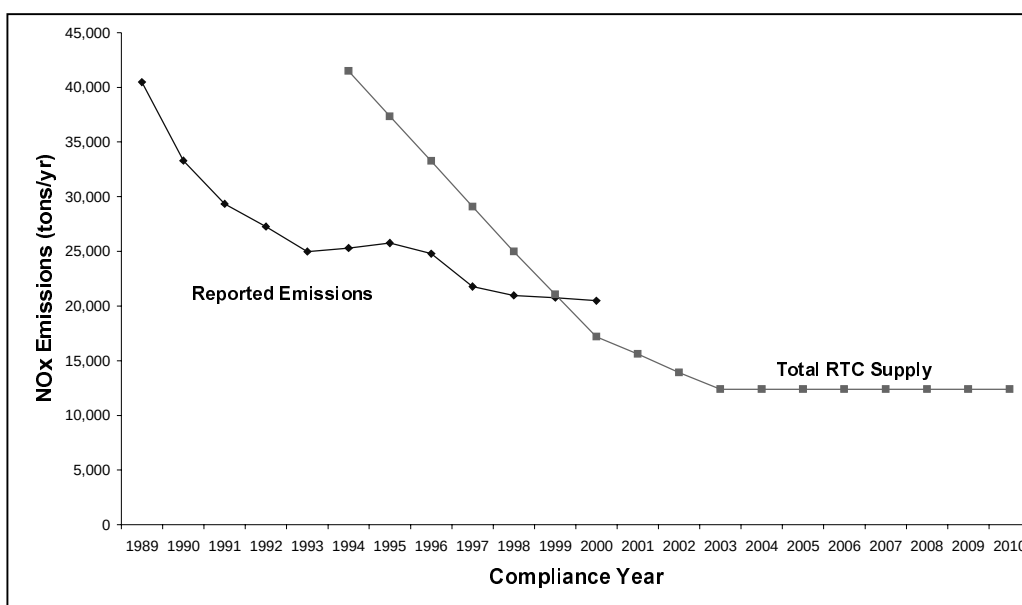
The role of power plant demand on the overall demand for NO_x RTCs is clear—emissions from power facilities were almost 6,800 tons, approximately 300 percent of their initial allocation. Although power facilities purchased more than 2,500 tons of NO_x RTCs, these purchases were not sufficient to put them into compliance, leading to a non-compliance excess of more than 1,900 tons. Indeed, as a group, non-power facilities sold more NO_x RTCs than they were allocated, yielding a non-compliance excess of more than 1,300 tons.

**Table 2.2. Impact of NO_x Emissions from Power Producing Facilities
on the Overall NO_x Allocations for Compliance Year 2000**

	Non-Power Producing Facilities (a)		Power Producing Facilities (b)		All Facilities (a) + (b)
	RTCs Held in CY 2000	Initial Allocations	RTCs Held in CY 2000	Initial Allocations	
Allocations (tons)	12,345	14,895	4,852	2,302	17,197
Emissions (tons)	13,703		6,788		20,491
Difference (Exceedance)	(1,358)	1,192	(1,936)	(4,486)	(3,294)

Source: South Coast Air Quality Management District (2002a).

Figure 2.12. NO_x Emissions and Available RTCs



Source: South Coast Air Quality Management District (2002a).

The temporal flexibility provided in RECLAIM—limited though it was—did, however, result in a substantial reduction in the actual excess emissions from the 19 percent level reported in Table 2.2. The data in Table 2.2 compares emissions in Compliance Year 2000 only to allocations issued by compliance year 2000. But RECLAIM has two overlapping 12-month cycles. This structure creates overlapping six-month periods so that RTCs from different compliance years can be used to offset emissions. In the case of 2000, sources could use RTCs from compliance years 1999 and 2001 as well as 2000.

Based upon the emissions monitoring and compliance reports that were submitted by RECLAIM sources, the amount of excess emissions totalled 1,089 tons, only about 6 percent above the Compliance Year 2000 allocation (South Coast Air Quality Management District 2002). Thus, as a result of the limited temporal flexibility in RECLAIM, the excess emissions were reduced by more than a factor of three—from about 19 percent to 6 percent.

Although the effect on emissions appears to be modest, the high NO_x RTC prices may have led to improper activity on the part of a prominent RTC brokerage firm. The SCAQMD issued a violation notice against one of the major RTC brokers for alleged violations of the emissions credit trading regulations (South Coast Air Quality Management District 2002b). The notice follows lawsuits charging the broker, Automated Credit Exchange (“ACE”), and its president with not providing RTCs that were paid for by their clients (Jacobs 2002). The SCAQMD notes, however, that these alleged problems have not compromised the environmental performance of the RECLAIM programme.

RECLAIM Modifications in May 2001

The high NO_x RTC prices triggered a backstop provision in RECLAIM that requires the Executive Officer of SCAQMD to submit an evaluation and review within six months of the determination that the average RTC price has exceeded \$15,000 per ton. The Executive Officer also is required to propose that the Governing Board amend the programme as appropriate to address specific problems. On January 11, 2001, the SCAQMD released a White Paper evaluating the increases in NO_x RTC prices and presenting near-term and long-term options to stabilize NO_x RTC prices (South Coast Air Quality Management District 2001a). This document reflected public input from an Advisory Committee. The staff concluded that increased demand by electric generation sources was the major cause of the abrupt change in NO_x RTC prices. The authors of the White Paper concluded that substantial lower-cost control alternatives were available for RECLAIM sources, but that these controls would take some time to be put in place. The staff recommended various changes to the RECLAIM programme, most of which were adopted by the governing board.

The governing board of the SCAQMD on May 11, 2001 approved the following changes to RECLAIM (South Coast Air Quality Management District 2001b):

- Power plants are separated temporarily from the rest of RECLAIM until at least December 31, 2003, after which they could be brought back into the programme;
- Power plants must submit by September 1, 2001, compliance plans that involve installation of pollution control equipment (equal to “Best Available Retrofit Control Technology” or “BARCT”) over the next two to three years;
- Power plants are allowed to pay into a mitigation fund for any excess emissions at the rate of \$7.50 per pound (\$15,000 per ton), with the SCAQMD using the mitigation fees to reduce pollution from other sources (e.g., diesel trucks and equipment at the port);
- Non-power facilities with emissions greater than 50 tons per year must submit by September 2001 plans either to demonstrate compliance with their NO_x allocations for years 2001 to 2005 based upon their 2000 production rate or to comply with BARCT by January 1, 2003;
- Temporary credit programmes will be established, including an Air Quality Investment Programme for mobile and area source credits that can be used by new power plants and other buyers;

- Additional and more timely information will be required on RTC trades, including a requirement that trades be reported jointly by the buyer and seller within five days of the trading transaction; and
- The SCAQMD will study the merits of a centralised trading market to replace the decentralised system involving individual RECLAIM companies, brokers or agents, and auctions.⁸

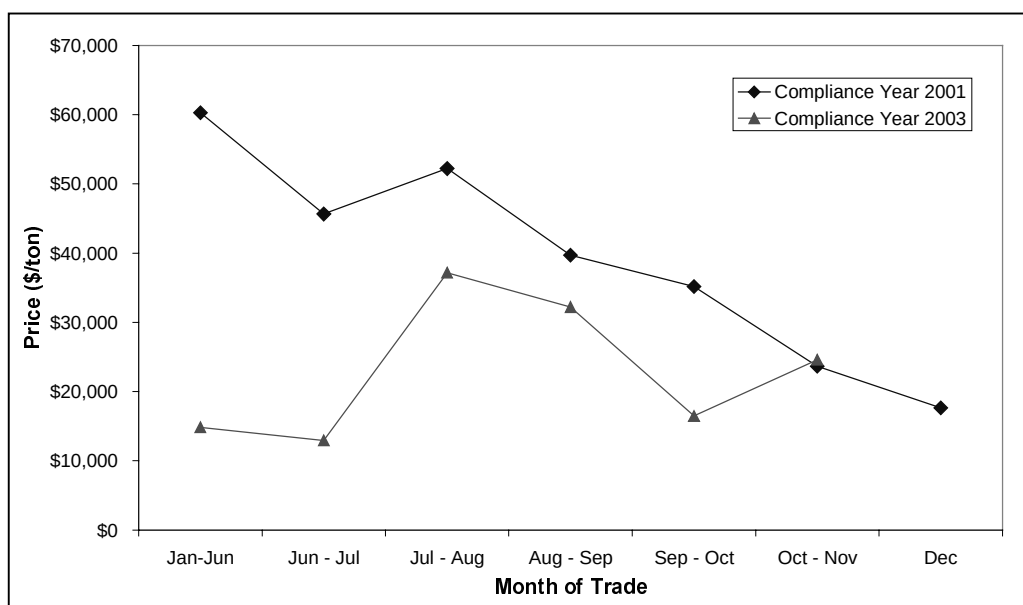
Experience After May 2001 Changes

Since the May 2001 changes were put in place, the RECLAIM market has been much less eventful -- prices have declined and emission targets have been achieved, albeit at least temporarily due in part to a command-and-control regime. Moreover, the crisis in the California electricity market that largely precipitated the extreme price RTC price volatility has subsided (see Joskow 2002).

RTC Prices

The May 2001 changes—particularly the bifurcation of power facilities from the RECLAIM programme—appear to have resulted in substantial suspension reductions in RTC prices for the remainder of 2001. Figure 2.13 shows monthly average RTC NO_x prices for the first half of 2001 (January-June) compared to monthly average prices for the last six months of 2001. Prices are shown for two vintages, 2001 and 2003. Prices for 2001 vintage NO_x RTCs dropped substantially over the last six months of 2001, from more than \$60,000 per ton in the first six months of 2001 to less than \$20,000 per ton by December 2001.

Figure 2.13. Changes in Prices for 2001 and 2003 NO_x RTCs by Month in Calendar Year 2001



Source: South Coast Air Quality Management District (2002a).

8. The SCAQMD concluded that a central market to supplant existing arrangements was not desirable.

Prices for NO_x RTCs have continued to fall in 2002 and 2003, presumably reflecting the introduction of emission control equipment and the lack of demand from power generators. Information from Cantor-Fitzgerald indicates that prices for 2003 NO_x RTCs were about 6,000 per ton at the end of 2002 and the beginning of 2003 (Cantor-Fitzgerald 2003).

Compliance Plans

The compliance plans required by the May 2001 RECLAIM changes represent at least a temporary shift to a command-and-control regime intended “to ensure timely installation of emission controls” (South Coast Air Quality Management District 2001); All power facilities were required to submit a plan by September 1, 2001 to demonstrate plans to install NO_x controls equal to BARCT. Deadlines for compliance with BARCT were set for January 1, 2004 for turbines used as peaking units and by January 1, 2003 for all other units. Non-power facilities still in RECLAIM emitting more than 50 tons per year also were required to submit a plan by September 1, 2001 to either demonstrate compliance with their NO_x allocations between years 2001 and 2005 (based on 2000 production rates) or comply with BARCT by January 1, 2003. Together, the facilities required to submit compliance plans accounted for about 90 percent of RECLAIM NO_x emissions in compliance year 2000.

The required compliance plans—14 by power producers and 42 by non-power producing facilities—were submitted by the due date of September 1, 2001. Table 2.3 shows the projected emissions reductions from the compliance plans, and Table 2.5 shows comparisons of projected emissions for the power producing facilities based upon the compliance plans with their RTCs, and the resulting emissions reductions necessary from the mitigation fees programme. Based upon the compliance plans, power facilities are projected to have NO_x emissions below their RTC allocations by Compliance Year 2003. Moreover, SCAQMD staff have recommended emission reductions beyond those called for in the compliance plans; according to the SCAQMD, these changes would yield additional NO_x emission reductions of 460 tons, 291 tons and 195 tons in Compliance Years 2003, 2004 and 2005, respectively.

Table 2.3. Projected Emission Reductions from RECLAIM Compliance Plans

Facility Category	Compliance Year				
	2001	2002	2003	2004	2005
Emission Reductions from Power Producing Facilities (tons)	2,559	3,414	4,896	5,373	5,494
Emission Reductions from RECLAIM Facilities with annual emissions greater than 50 tons of NO _x emissions (tons)	6,855	6,949	7,539	8,122	8,147
Total (tons)	9,414	10,363	12,435	13,495	13,641

Source: South Coast Air Quality Management District (2002a).

Table 2.4. Comparison of Projected Emissions and Available RTCs at Power Producing Facilities

<i>Compliance Year</i>	<i>Projected NO_x Emissions in Tons</i>	<i>RTCs Available in Tons</i>	<i>Emission Reductions Necessary from Mitigation Fees Programme</i>
2001	4,010	2,735	1,275
2002	3,155	2,371	784
2003	1,673	2,037	None
2004	1,196	2,165	None
2005	1,075	2,156	None

Source: South Coast Air Quality Management District (2001d).

Mitigation Fees and RTC Credits

The net result of the compliance plans and the SCAQMD projections is that SCAQMD expects emissions to be below RTC levels for all future compliance years. As reported by the SCAQMD, emissions in 2001 are lower than originally expected because of lower than expected electricity demand during the second half of calendar 2001. Thus, the SCAQMD projects little need for RTC credits to be generated from the mitigation fee payments.

Table 2.5 shows SCAQMD records regarding the RTCs that were added as a result of conversion from mobile and area sources over the history of RECLAIM, from 1994 to 2002. Over the entire period, a total of 598 NO_x RTC tons were added. This total is a small percentage of the average annual NO_x RTCs over the period. These figures indicate that project-based credits had a relatively small effect on the RTC market, and did not provide much of a “cushion” to absorb the increase in demand that occurred in 2000 and 2001.

Table 2.5. NO_x RTCs Converted from Mobile and Area Source Credits, 1994-2002

<i>Year</i>	<i>Amount (tons)</i>	<i>Source of Credits*</i>
1994	33	Rule 1610
1995	36	Rule 1610
1996	36	Rule 1610
1997	4	Rule 1610
1999	50	Rule 1612
2000	150	Rule 1612
2000	68	Rule 2506
2000	13	Rule 1620
2001	72	Rule 1612
2001	68	Rule 2506
2002	68	Rule 2506

*Rule 1610 – Old Vehicle Scrapping

Rule 1612 – Credit for Clean On-Road Vehicles

Rule 2506 – Area Source Credits for NO_x and SO₂

Source: Lieu, Dabirian, and Dawson (2001).

The SCAQMD has developed several projects to generate emission reductions to offset excess power plant emissions. In 2001, the SCAQMD contracted to reduce NO_x emissions from marine vessels by 175 tons per year over the four-year period from 2002 to 2005.

Table 2.6 summarizes the SCAQMD’s most recent estimates of the likely market conditions for NO_x RTCs in various future compliance years, taking into account emissions from power plant (Rule 2009), non-power plants required to submit compliance plans or show compliance (Rule 2009.1), and other RECLAIM facilities. According to these projections, the NO_x RTC market would achieve a surplus (excess of RTCs over emissions) after the 2003 compliance year.

Table 2.6. Projected Emissions and Available NO_x RTCs for all RECLAIM Facilities

Compliance Year	Projected NO_x Emissions (Tons)					Available RTC (Tons)	Excess RTCs (Tons)
	Rule 2009	Rule 2009.1 Option 1	Rule 2009.1 Option 2	Other Facilities#	Total		
2001	4,010	7,633	1,269	2,509	15,421	15,266	0 – (195)
2002	3,155	7,539	1,269	2,509	14,472	13,833	0 – (639)
2003	1,673	6,949	1,269	2,509	12,400	12,396	0 – (4)
2004	1,196	6,366	1,269	2,509	11,340	12,396	1,056
2005	1,075	6,341	1,269	2,509	11,194	12,396	1,202

Source: South Coast Air Quality Management District (2001d).

IV. Conclusions Regarding RECLAIM Experience

The nine-year experience with RECLAIM—including its relatively uneventful early years, its tumultuous years of 2000 and 2001 and its recent return to normalcy—provides a basis for tentative conclusions regarding its economic and environmental performance.

Cost Savings Relative to Command-and-Control

When the programme was being developed, cost savings were estimated to be about 40 percent compared to the cost of achieving the same emission levels using the traditional command-and-control approach (see Harrison and Nichols 1992 and Johnson and Pikelney 1996). No *ex post* estimates of cost savings have been made. It is possible, however, to speculate on the cost savings under RECLAIM.

Volume of RTC Trades

The high volume of trading in the RECLAIM programme implies significant cost savings relative to the command-and-control alternative that it replaced. Although there is some double counting of trades—because of the RECLAIM accounting rules—it is clear that substantial cost-reducing RTC trades have taken place over the nine years the programme has been in place.

Use of Mobile and Area Source Reduction Credits

The use of mobile and area source reduction credits in theory should add to the cost savings, because it introduces the possibility of lower-cost control options. In practice, however, these credits have accounted for a very small share of overall emissions allowances. Indeed, some criticise EPA for not approving mobile and area source credit rules and advocate expanding these alternatives to increase potential cost savings (see Wyman 2002).

Compliance Plans for Electricity Generators

The May 2001 requirements for electricity generators to adopt BARCT controls reflected a concern that RECLAIM participants had not taken proper measures to reduce their emissions, relying too much on the market to purchase RTCs. This logic disregards the unusual circumstances that precipitated the dramatically increased demand for RTCs in 2000 and 2001. Whether the required compliance plans have led to cost-effective emission reductions that should have been adopted or instead represent excessively costly controls has not been studied.

Environmental Performance

The environmental performance of RECLAIM can be measured along several dimensions, including the ability of the programme to meet the increasingly-stringent cap, the likely comparison with the command-and-control alternative, and the accuracy (and cost-effectiveness) of monitoring information.

Emissions Relative to Caps

Notwithstanding the relatively lenient early caps and the extreme pressure of prices that increased ten-fold in the course of a few months of 2000, RECLAIM has been successful in achieving the caps that were set in 1993 when the programme was passed. Based upon compliance annual compliance audits for 1994 to 2000, the cap was exceeded only in 2000 for NO_x. Moreover, when the

effects of the limited banking/borrowing that is allowed are taken into account, the excess emissions in 2000 represent only a six percent increase. In addition, because this excess is reflected in fewer RTC allowances in future years, the emissions increases will be “made up” in the future; in essence, the 2000 excess represents a borrowing of emissions allowances rather than a net increase in overall programme emissions.

Comparisons to Command-and-Control Alternatives

There is no reason to believe that a command-and-control alternative would have performed better than RECLAIM did under the circumstances. Indeed, since command-and-control mandates typically regulate emission *rates*—rather than overall emissions—a command-and-control alternative would likely have resulted in the same emissions increases but without the compensating measures taken as a result of exceeding the NO_x RECLAIM cap. Put another way, the command-and-control alternative does not have a natural mechanism to provide for borrowing (or banking) of emissions reductions.

Accuracy of Emissions Monitoring

Assessments of the environmental performance of RECLAIM depend in part on assessments of the accuracy of the monitoring data. Under RECLAIM, the largest emission sources were required to install expensive CEMs in order to provide credible and accurate emissions data. The early experience with CEMs was problematic; many of the initial data could not be used because of CEM malfunctions. But these technical issues were resolved, and the CEMs data currently provide reliable emissions values. It is possible, however, that similarly reliable data could be obtained with less expensive monitoring (although perhaps with less public acceptability).

Other Considerations

The RECLAIM experience suggests several additional considerations beyond the likely effects on emissions and costs.

Links Between Emissions and Electricity Markets

Perhaps the most important lesson from the 2000 experience with RECLAIM is that the “problems” were due primarily to flaws in California’s newly deregulated electricity markets rather than to serious flaws in the RECLAIM programme itself. RECLAIM behaved largely as it should have. Demand for NO_x RTCs increased, their supply decreased and their prices increased as they should have, and the prices of an important product, electricity, that required NO_x credits also increased. This should have provided signals to affected sources to invest in emissions controls, as well as signals to consumers to reduce consumption of electricity. Had the structure of California’s electricity industry remained composed of regulated monopolies, or if the transition to new competitive market structures had been done more competently and with greater sensitivity to the interactions between NO_x RTC prices and spot electricity prices, the impact of higher NO_x RTC prices on the average price of electricity would have been much more modest. And if the electricity crisis had not occurred, electricity generators likely would not have been removed—at least temporarily—from the NO_x RECLAIM programme.

Administrative Costs and Broker Activity

Administrative costs under RECLAIM appear to be relatively modest, contributing to the overall cost savings. Developing the regulatory programme was costly, but in all likelihood substantially less costly (and time consuming) than developing all of the command-and-control regulations that

RECLAIM supplanted. Moreover, the ongoing administrative costs of RECLAIM are almost certainly less than those for the command-and-control alternatives, which require ongoing assessments of technical alternatives.

Brokers have participated actively in RECLAIM and no doubt have helped to promote active markets in RTCs. Moreover, there is no evidence that a recent scandal involving a prominent RTC broker—which involves allegations of substantial misconduct—has had an effect on either the economic or environmental performance of RECLAIM.

Means of Addressing Price Volatility

Despite the general, solid performance of RECLAIM, its experience in 2000 and 2001 make it clear that short-term fluctuations in emissions levels can lead to significant volatility in allowance prices. Allowance price volatility in turn can lead to significant short-term volatility in prices of goods whose production involves significant “use” of emissions allowances. Accordingly, the recent experience with RECLAIM suggests that it is important to give emissions market participants the necessary tools to manage extreme price volatility effectively. Assuming the use of a marketable permits programme—rather than an emissions tax approach—several mechanisms could remedy price volatility problems (Ellerman 2001). These include expanding the ability to bank/borrow RTCs, expanding the range of sources participating in the programme, and providing a mitigation fee or (if damages roughly linearly related to emissions) a “safety valve” that would cap RTC prices.

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Chapter 3

THE U.S. SO₂ CAP-AND-TRADE PROGRAMME

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I. Introduction

The U.S. SO₂ cap-and-trade programme was established as a result of the enactment of the 1990 Clean Air Act Amendments (1990 CAAA) under the authority granted by Title IV, which included several measures to reduce precursor emissions of acid deposition.² The SO₂ component consisted of a two-phase, cap-and-trade programme for reducing SO₂ emissions from fossil-fuel burning power plants located in the continental forty-eight states of the United States. During Phase I, lasting from 1995 through 1999, electric generating units larger than 100 MW^e in generating capacity with an annual average emission rate in 1985 greater than 2.5 pounds of SO₂ per million Btu of heat input in 1985 (hereafter, #SO₂/mmBtu) were required to reduce emissions to a level that would be, on average, no greater than 2.5 #SO₂/mmBtu. In Phase II, beginning in 2000 and continuing indefinitely, the programme was expanded to include fossil-fuel electricity generating units greater than 25 MW^e, or virtually all fossil-fuel power plants in the United States. Emissions from these affected units are limited, after accounting for any allowances banked from Phase I, to an annual cap of 8.9 million tons, or about half of total electric utility SO₂ emissions in the early 1980s. The Phase II cap is equivalent to an average emission rate of 1.2 #SO₂/mmBtu, when divided by the mid-1980s level of heat input at fossil-fuel burning power plants.

This cap on national SO₂ emissions was implemented by issuing tradable allowances—representing the right to emit one ton of SO₂ emissions—equal in total to annual allowed emissions from affected units in each year after 1995, and by requiring that the owners of these units surrender an allowance for every ton of SO₂ emitted. Allowances not used in the year for which they are allocated can be carried over or banked for future use by the original owner or by any party to whom the banked allowance is sold. Allowances are allocated to owners of affected units free of charge for

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1. The views expressed in this chapter are those of the author and do not necessarily reflect those of the institutions with which the author is affiliated, or the OECD.
 2. The most important of the other measures reduced NO_x emissions by two million tons by imposing technology-based, maximum average annual NO_x emission rates on affected sources. In meeting these standards, utilities were allowed to average emission rates among the units they controlled, but not to trade NO_x emissions among utilities.

the next thirty years, generally in proportion to each unit's average annual heat input during the three-year baseline period, 1985-87. A small percentage (2.8 percent) of the allowances allocated to affected units are withheld for sale through an annual auction conducted by the EPA to encourage trading and to ensure the availability of allowances for new generating units. The revenues from this auction are returned on a *pro rata* basis to the owners from whose allocations the allowances were withheld.

The SO₂ cap-and-trade programme also contained several provisions that allowed generating units not subject to the cap until Phase II to opt-in to Phase I and to receive allowances for the year in which the unit participated. These units were then subject to the same compliance requirements as the 263 units that were mandated to be part of Phase I, namely, that they must surrender allowances equal to emissions in that year. Also, SO₂-emitting industrial sources not otherwise affected by Title IV could establish baselines and be allocated allowances and participate like any other unit in Phases I and II.

II. The Political and Regulatory Context of Title IV

Three features of the political and regulatory context are important in evaluating the SO₂ cap-and-trade programme. The first is that the cap-and-trade system is not the only means, nor the first means, of controlling SO₂ emissions from electric utility power plants in the United States. The cap-and-trade system supplements an extensive set of command-and-control regulations that has been in effect since the early 1970s. These regulations take two principal forms according to whether power plants were in existence when the regulations implementing the 1970 Clean Air Act Amendments became effective. Plants already in existence or under construction in 1971 must meet emission rate limits imposed by State Implementation Plans (SIPs), which the individual states are required to develop in order to bring all areas of the country into compliance with National Ambient Air Quality Standards (NAAQS) for six "criteria" pollutants (including SO₂). New units constructed after the effective date of the 1970 Amendments are required to meet the New Source Performance Standard (NSPS), which is a technology-based, uniform national requirement that, in the case of SO₂, effectively requires new coal-fired generating plants to install flue gas desulfurisation equipment (or a scrubber).³ New sources have additional requirements if they are to be located in areas not in attainment with the NAAQS (non-attainment areas). Sources locating in areas that are in attainment may also face prevention of significant deterioration (PSD) requirements, which are intended to ensure that areas in attainment do not slip into non-attainment status. Finally, any source located near a national park or other pristine (Class I) area may be required to meet additional limits, such as those aimed at preserving visibility. Typically, all of these pre-existing regulatory requirements impose either emission rate limits or technology mandates on individual units. This complex and comprehensive, underlying command-and-control structure means that Title IV is not burdened with meeting all environmental objectives. Other regulatory mechanisms are available to ensure that adverse local health effects are avoided and that other environmental values, such as visibility, are preserved. Another consequence of this regulatory context is that the ability of individual power plants to participate in emissions trading can be, and often is, limited by these other requirements.

The second notable feature of the political and regulatory context is that the motives lying behind enactment of Title IV are mixed, as is the case for most legislation. The ostensible purpose and most

3. The scrubber mandate for new units was added by the 1977 Amendments to the Clean Air Act. The original NSPS provisions of the 1970 Clean Air Act required only that emissions from new coal-fired power plants be limited to 1.2 #SO₂/mmBtu. This standard was achievable either by installing a scrubber or switching to a limited sub-set of coals (thereafter known as compliance coals) that emitted less than 1.2 #SO₂/mmBtu without scrubbing. Ackerman and Hassler (1981) provide the now classic account of the interest group politics and other considerations leading to the redefinition of the NSPS in 1977.

commonly cited motive is to reduce the effects of acid deposition, a cumulative environmental problem, the effects of which are experienced mainly in the Northeast in large part as a result of SO₂ emissions originating from the heavy concentration of coal-fired power plants in the Mid-West. Yet, SO₂ emissions from power plants located in other parts of the country, such as Florida, that have little effect on the Northeast or other areas suffering from acidic deposition are included in the Acid Rain Program; and emissions from these sources are considered, for the purposes of emissions trading, as completely equivalent to emissions from power plants located in areas that are far more likely to have an effect on sensitive receptor areas. Two other motives operated at the time of enactment. The first concerned fine particulates, which research on health effects was beginning to implicate as a threat to public health. Although considerable controversy surrounded the origin of fine particulates—and such questions would need to be resolved in order to revise the appropriate NAAQS—SO₂ emissions from coal-fired electric power plants were considered a likely contributor. A second, and probably more important, motive was a desire to narrow the disparity between the emission limits imposed on new sources by the NSPS and the limits imposed on existing sources by State Implementation Plans. If SO₂ emissions were to be reduced for any of these reasons, something more than the existing regulatory structure would be needed since nearly all areas of the United States were in compliance with the SO₂ NAAQS by the 1980s. Moreover, the use of tall stacks to loft SO₂ emissions high above ground to avoid violating the local ambient standard exacerbated the acidic deposition in more distant down-wind regions. A fifty percent reduction in the aggregate level of SO₂ emissions came to be viewed as a measure that would at once significantly reduce the amount of SO₂-originated deposition in the Northeast, contribute to some reduction of fine particulates, and largely close the disparity between the emission requirements imposed on new and existing sources. It is telling with respect to this last motive that the emission rate standard used to decide the cap and to allocate allowances in Phase II is identical to the original New Source Performance Standard enacted in the 1970 Amendments to the Clean Air Act.

The third and final feature of the political and regulatory context surrounding enactment of the SO₂ cap-and-trade programme is that it ended a decade of debate concerning additional controls on existing coal-fired power plants. Earlier proposals would have achieved a similar 50% reduction of total SO₂ emissions by mandating scrubbers on the largest power plants and mandating switching to lower sulfur coal with limited trading. These earlier proposals were viewed as very costly, they faced the adamant opposition of the Reagan Administration, and they failed to gain a legislative majority in several sessions of Congress. The willingness of the new Bush (père) Administration to back significant SO₂ emission reductions, so long as they were achieved by market-based mechanisms, and of some environmental lobbying groups, notably the Environmental Defense Fund, to experiment with new and potentially more effective means for achieving environmental goals broke the stalemate and allowed a legislative majority to coalesce around a proposal that would reduce aggregate SO₂ emissions significantly and achieve the disparate goals that motivated various actors in the political process.

III. Institutional Location and Methodology

Unless otherwise noted, this paper is based on the continuing ex post evaluation of the U.S. SO₂ cap-and-trade programme that faculty and students associated with the Center for Energy and Environmental Policy Research (CEEPR) at the Massachusetts Institute of Technology (MIT) have conducted since 1995. This evaluation has been a major focus of CEEPR's research programme, which aims to inform the public policy process by providing the results of objective, theoretically sound, and empirically rigorous research through publications and less formal presentations to interested audiences.

This evaluation effort was initially funded by the National Acid Precipitation Assessment Programme (NAPAP) to support the 1996 Quadrennial Report to the U.S. Congress and the research has received continued funding through grants from the U. S. Environmental Protection Agency and from the underlying financial support provided to CEEPR by a number of corporate sponsors. An equally, if not more, important contribution to the success of this multi-year evaluation has been the abundance and ready accessibility of emissions and allowance data that has been made available to analysts and the general public by the U.S. Environmental Protection Agency through its website.

The results of the first years of the evaluation conducted by MIT's CEEPR are presented comprehensively in *Markets for Clean Air: The U.S. Acid Rain Program* (Ellerman et al., 2000), which is cited by OECD (2001) as an example for conducting ex post evaluations. This paper updates *Markets for Clean Air*, and it incorporates more of the work of other researchers who have since published on various aspects of the programme.

In specifying the requirements of an ex post evaluation, OECD (2001) reinforced the admonition of Frondel and Schmidt (2001) that “the essential task of any evaluation analysis is the construction of a credible counterfactual situation—a precise statement of what economic agents would have done in the absence of the policy intervention.” With this in mind, the rest of this section describes the counterfactuals used in evaluating the SO₂ emissions trading programme.

Two counterfactuals are involved in assessing any emissions trading programme: one to assess the amount and cost of the emission reduction and the other to assess the cost savings and other effects of trading. The counterfactual for assessing the emission reduction requires assumptions about basic economic drivers, such as the demand for electricity and the relative price of fuels, and about other environmental regulations that may limit emissions, all of which can be observed ex post. In the case of the SO₂ programme, the observed utilization of individual units provides a reasonably close estimate of the effect of the basic economic drivers in any given year. The effect of the pre-existing regulatory regime can be captured in the emission rate observed shortly before the start of the cap-and-trade programme. Accordingly, the counterfactual used in this paper, as in previous work by the author and colleagues, is based on the heat input observed at affected units in each year and an unchanging pre-Title IV emission rate at those units.

This counterfactual assumption has the effect of making the estimated emission reduction equal to the heat-input-weighted changes in observed emission rates at affected units and to assume that no emission reduction can be attributed to changes in demand, either at individual units or in the aggregate. Since the demand for electricity is price inelastic, the cost of SO₂ controls is relatively small on a kilowatt-hour basis, and the major element determining the dispatch, or utilization, of individual generating plants is the cost of fuel, the error arising from assuming no effect on demand is probably small. Nevertheless, to the extent that the added costs from the programme reduce the demand for electricity or change the order of dispatch of generating units in meeting that demand, the effect of the programme is under-estimated.⁴ A more likely source of error arises from the assumption about the counterfactual emission rate. To the extent that other environmental regulations, or changes in relative fuel prices, cause the emission rate at affected units to fall during the period of evaluation, the effect of the SO₂ programme is over-estimated. Increases in the true counterfactual emission rates would have

4. The appendix to *Markets for Clean Air* contains an econometric estimation of the extent to which Title IV requirements changed the dispatch of generating units during Phase I. In brief, the demand placed on unscrubbed units subject to Title IV was shifted to affected, scrubbed units and to non-affected, Phase II units. Both effects are relatively small and the latter did not increase emissions perceptibly since the emission rates for affected unscrubbed units were generally higher than the emissions rates for non-affected units, all of which were exempt from Phase I because of a lower emission rate.

the opposite effect, but the scope for these is limited since all units face emission rate limits under the pre-existing command and control regulation and those limits are rarely, if ever, increased.

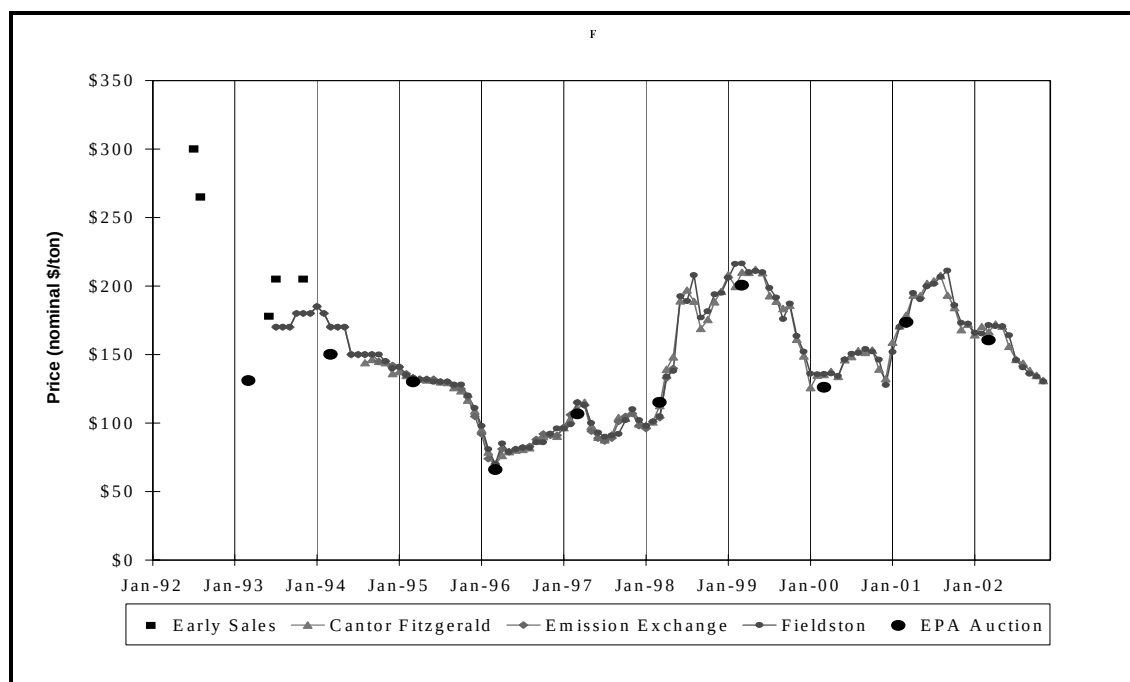
The other counterfactual, that used to assess trading, is much harder to specify. This other counterfactual requires a hypothetical, equally effective, alternative programme without emissions trading. Estimates of cost savings are necessarily more subjective since they depend directly on the degree of inefficiency assumed in the imagined alternative regime. In this paper, a source-specific, quantity limit equal to the allowance allocation to specific units is used. This assumption conforms with the well-established propensity to source-specific limits (although rarely on total emissions from an individual plant), but it is relatively benign in not having a technology mandate similar to that characterising much of the existing regulatory structure and to that contained in earlier, failed legislative proposals.

IV. Economic Efficiency

Two aspects of economic efficiency need to be distinguished in evaluating cap-and-trade programmes. The first concerns trading among firms subject to the cap and the extent to which they realise the full cost savings attainable through emissions trading. The second aspect of economic efficiency concerns the broader welfare effects from the tax and regulatory interactions resulting from the treatment of abatement costs and the scarcity rents generated by the environmental constraint. From the standpoint of this second aspect, it has been argued that Title IV did not achieve full economic efficiency because first, allowances were not auctioned *and* the proceeds used to reduce distortionary taxes on labour and capital, and second, the average cost rules applying to units remaining under public utility cost-of-service regulation prevent the full marginal cost of abatement from being passed on to customers in the price of electricity (Goulder *et al.*, 1997). A full discussion of this aspect of the economic efficiency of Title IV would involve consideration of the practical likelihood of economically efficient recycling, of equity concerns, and how public utility regulation is applied in practice: all topics that are beyond the scope of this paper. Henceforth, all references to economic efficiency in this paper refer to the conventional use in emissions trading, that is, to the cost savings resulting from the flexibility provided by emissions trading without regard to the larger welfare issues reflecting allocative inefficiencies that may result from the interaction of the trading programme with the existing regulatory and tax system.

The primary evidence for the economic efficiency of the SO₂ cap-and-trade system lies in the early emergence of an allowance market and the significant amount of trading that has occurred since before the programme started. Figure 3.1 depicts the movement of allowance prices from the earliest observations through late 2002 as reported monthly by various brokers and in the annual EPA auction.

**Figure 3.1: SO₂ Allowance Prices, 1992-2002
(Spot, Current Vintage Prices)**

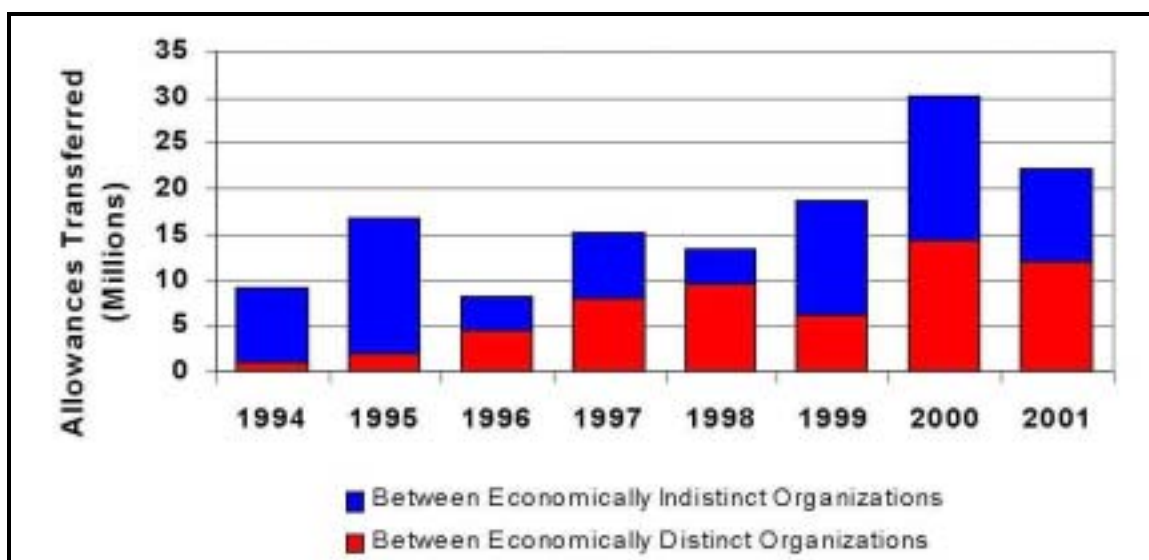


Source: Compiled by the author from monthly broker reports and the annual EPA auction reports

Prices have varied substantially over time—from an all-time low of \$65 in early 1996 to highs slightly above \$200 in 1999 and again in 2001—but at any one moment in time a single price prevails. The earliest reported trades took place at widely disparate prices, which were higher than the clearing price in the first EPA auction, held in March 1993. At this time, it would be hard to say that a market existed; however, by mid-1994, approximately six months before Phase I entered into effect, a market seems to have formed and the law of one price has prevailed since then.

Since allowances are readily substitutable for abatement, this single price provides a common point of reference and a coordinating mechanism for all owners of affected sources in deciding whether to abate more or less at any one time and thereby to equalise the marginal cost of abatement. Moreover, the significant and increasing volume of trading between economically distinct organizations, as illustrated in Figure 3.2, suggests that utilities are taking advantage of the cost-saving opportunities provided by emissions trading.

Figure 3. 2: Annual Allowance Trading Activity

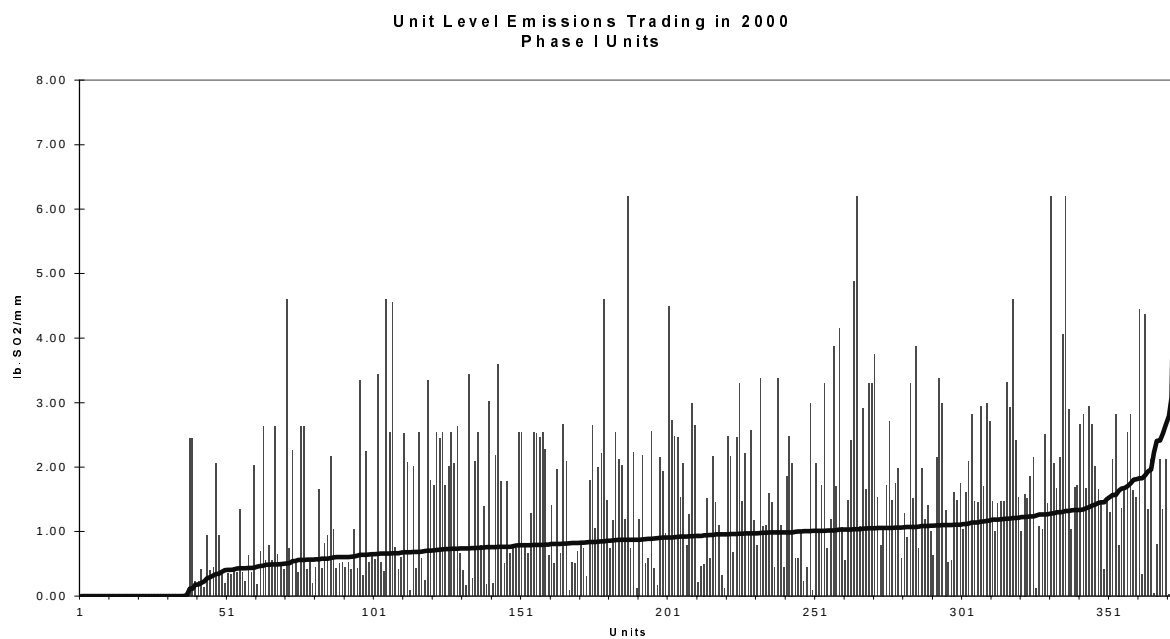
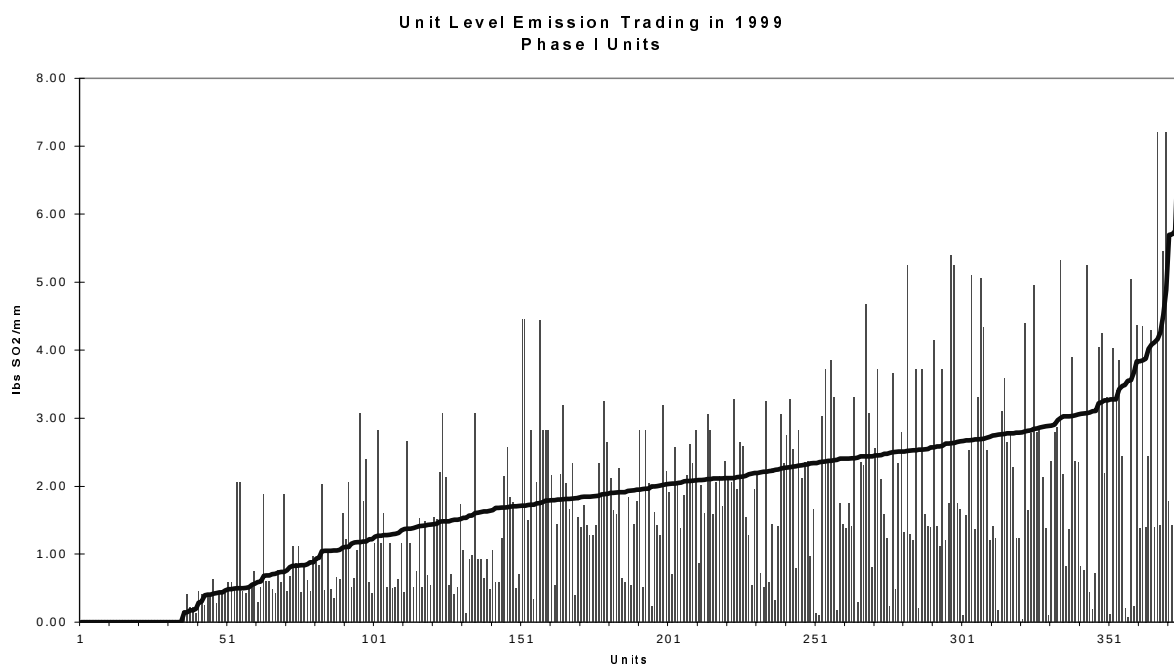


Source: US EPA

Since the equalization of marginal costs presumes a common price and trading among sources facing different costs, the preconditions for cost-effective abatement are being observed. An argument that the efficiency goals of the programme are not being achieved would require an alternative hypothesis to explain the existence of a market and the observed volume of trade. In fact, no observer argues that observed trades are motivated by other than expected cost savings. As will be discussed later in this paper, the only disagreement among analysts concerning the economic efficiency of the SO₂ cap-and-trade programme concerns the extent to which the full potential cost savings have been achieved.

Further evidence to support the argument for economic efficiency can be observed in the unit-level differences between allowances and emissions. The two panels of Figure 3. 3 show for Phase I affected units in 1999 and 2000 the emission rate that would be observed with no trading (the solid line) and the actual rate (the columns), given the heat input at each unit in these years.

Figure 3.3



Few units are along the solid line, where they would have to be in the absence of trading, either when the allowance allocation was relatively generous in 1999 or when the significantly reduced Phase II allocation went into effect. The average difference between observed emission rates and the no-trading rate is about 50% of the mean emission rate: 0.81 #SO₂/mmBtu over 1.64 #SO₂/mmBtu in 1999 and 0.86 #SO₂/mmBtu over 1.48 #SO₂/mmBtu in 2000.

A further indication of economic efficiency is given by the relatively small change in average observed emission rates (-10%) when the number of allowances made available to units for use in 2000 declined by 53% from the level in 1999, when Phase II began. This smaller change in emission rates could occur only with banking; and in fact these 375 units went from banking 1.8 million allowances in 1999 to drawing the accumulated bank down by 1.5 million tons in 2000. This pattern of aggregate abatement over time is characteristic of an optimal banking programme with certainty, in which firms take future required abatement and prices into account in formulating current abatement plans. In turn, this behaviour implies that allowance prices rise at the interest rate and abatement increases gradually over the entire banking period. Such a pattern is observed in the transition from Phase I to Phase II among the units affected in both years. Moreover, despite all the stochastic variation in allowance prices since early 1994, as shown in Figure 3.1, a definite upward trend can be observed.

Recent research by Ellerman and Montero (2002) confirms that in the aggregate banking has been surprisingly optimal. The surprise resides in the general consensus, voiced in *Markets for Clean Air* as well as elsewhere, that too much banking had occurred in Phase I. The explanation of the surprise lies in the discount rate applicable to SO₂ allowances. The prices shown on Figure 3.1 allow a discount rate to be derived for SO₂ allowances by application of the capital asset pricing model to determine the amount of *undiversifiable* risk associated with holding SO₂ allowances. This risk is expressed by the correlation of returns from holding allowances (i.e., the monthly change in allowance prices) with returns from a well-diversified portfolio of equities over the same period of time. This correlation is zero, which makes SO₂ allowances zero-beta assets that should be discounted at the risk-free rate for comparable holding periods.

Compliance Costs and Savings from Emissions Trading

While the emergence of an SO₂ allowance market and the concomitant growth in the volume of SO₂ allowance trading suggests strongly that cost savings are being realised, these data alone provide no estimates of the magnitude of the cost savings, nor of the relation of these savings to actual or avoided, command-and-control compliance costs. In the case of the Acid Rain Program, many assertions have been made about the cost savings, but only two rigorous ex-post evaluations of compliance cost have been made [Carlson et al., 2000; Ellerman et al, 2000; hereafter, respectively, CBCP (for the initials of the authors) and MCA (for *Markets for Clean Air*)]. These two studies agree in finding the more extreme claims of cost savings unfounded, and their estimates of actual compliance costs are approximately the same, but they differ concerning the extent of the cost savings in the early years, as well as in methodology.

Ex Post Estimates of Compliance Cost

In reviewing the debates about the cost savings from Title IV, two distinctly different definitions must be kept in mind: one, loosely defined but more repeated; the other, more rigorous but less frequently cited. The former defines the cost savings as the difference of observed costs from predicted costs. The difference is loosely attributed to emissions trading even though other factors can and did intervene to cause actual costs to be lower than predicted. The second definition, used by the two studies cited above, uses a more rigorously defined no-trading alternative that incorporates identifiable cost-reducing exogenous factors. Accordingly, the following discussion will discuss first

the findings of the two studies on actual compliance cost, then compare them with earlier estimates, and finally address the differences between the two studies concerning the magnitude of the cost savings.

CBCP and MCA agree roughly on the cost of compliance in the early years of the SO₂ Cap-and-Trade Programme. The latter estimates the cost of compliance at \$726 million in 1995 and about \$750 million in 1996, while the former places the cost at \$832 million in 1995 and \$910 million in 1996, all stated in 1995 dollars. These estimates are not as far apart as they would seem. Complete comparability is not possible because of differences in methodology; however, both treat scrubber expense in the same manner.⁵ Although they largely agree on the fixed cost of scrubbers (\$375 million in MCA and \$382 million in CBCP), they differ significantly on the variable costs associated with scrubbers (\$89 million and \$274 million, respectively).⁶ CBCP uses scrubber data that reflect pre-1995 estimates of the variable cost of scrubbing, but the actual performance of the Phase I scrubbers has been much better than predicted, as will be discussed more fully in the section of this paper concerning dynamic aspects. Correction of this item alone largely removes the disparity in cost estimates between these two ex post evaluations. As an approximate figure, \$750 million is probably a good estimate of the annual cost of abatement in the first years of Phase I.

Comparison with Ex Ante Estimates of Cost

The important difference, however, is not the minor one between CBCP and MCA concerning actual costs in 1995-96, but the larger one between these two careful ex post estimates and ex ante estimates of the same Phase I cost, as well as of predicted costs in Phase II. Most of the disparity between ex ante and ex post estimates reflects very different assumptions about the nature of proposed acid rain controls, the demand for electricity, and the relative availability and cost of low sulfur coal. For instance, the total annual costs associated with some of the early proposals to control acid rain precursor emissions were estimated at amounts ranging from \$3.5 to \$7.5 billion. Although the details of these earlier proposals varied, they generally mandated scrubbers at a significant number of units and allowed very limited emissions trading. Once the proposal that ultimately became Title IV was proposed (in 1989) and enacted (in 1990), the ex ante cost estimates for the fully phased-in programme with trading fell to a range from \$2.3 billion to \$6.0 billion, with most of this variation reflecting varying assumptions about the extent to which emissions trading would be used.⁷

5. MCA provides a bottom-up, plant-by-plant analysis based on reported capital costs and observed sulfur premia. CBCP conducts an econometric estimation of a translog cost function and share equations of unit-level data for 734 *non-scrubbed* units over the 1985-94 period and then takes the resulting parameter values to form marginal abatement cost functions for individual units, which are then used to estimate actual costs based on observed 1995-96 emission levels. Scrubbed units are handled separately on a cost accounting basis using identical cost of capital and depreciation assumptions as in Ellerman *et al.* (2000). It should be noted that the estimation of 1995-96 cost in CBCP is almost an aside to the main purpose of the article which is to explain the reduction in abatement cost from pre-1995 estimates and to provide updated estimates of the cost of compliance in 2010.
6. The numbers cited from CBCP are from their break-out of the costs of 2010 compliance. This estimate will be approximately the same as the scrubber costs in 1995-96 since the fixed costs are annualised over 20 years, fuel costs are assumed not to change after 1995, the number of scrubbers remains unchanged, and costs are stated in 1995 dollars.
7. MCA includes (pp. 231-235) a discussion of the few ex ante estimates of Phase I costs (in contrast to Phase II costs when the cap is fully phased in) and compares them with the MCA estimate of actual cost in Phase I. Most of the variation in these estimates, made only a few years before Phase I began, reflects differing assumptions about the extent to which utilities made full use of the flexibility afforded by emissions trading. When compared on an average cost basis to account for differences in assumptions about the quantity of abatement, the MCA estimate of actual cost in 1995 was slightly above (3-15%) ex ante estimates assuming full use of emissions trading and 20-35% below estimates that assumed relatively little use of emissions trading.

The now current estimates for compliance costs in 2010, as provided by CBCP and MCA, are significantly lower still, \$1.0 billion and \$1.4 billion, respectively, for what is the same programme but updated to reflect more current market conditions.

CBCP provides a very helpful quantification of the causes of the change between the early estimates of Title IV and the current estimates. In examining the changes over the period of their panel regression, 1985-94, they find that the marginal cost of abatement for a representative unit reduction has been approximately halved and that 80% of the reduction in cost is attributable to falling price of low-sulfur coal relative to the price of high sulfur coal and that the remaining 20% is attributable to technological change. The change in the relative price of low sulfur coal is discussed in more detail in Ellerman and Montero (1998), who attribute the change to reduced rail rates, made possible by rail deregulation, for transporting distant, but cheap western coal to mid-western markets where local, high-sulfur coal had predominated. They estimate that the switching of mid-western high sulfur coal units, most of whom were mandated to be subject to Title IV in Phase I, to lower cost and lower sulfur western coal reduced the amount of abatement required to meet the Phase I cap by about 1.7 million tons, or by about half of that predicted by early estimates of required abatement.

Table 3.1 provides CBCP's quantification of the effects of these exogenous changes on estimates of compliance costs for a fully phased-in Title IV programme.

Table 3.1: Total Cost of Compliance with Title IV in 2010 (billion 1995 dollars)		
Cost Assumptions	Command-and-Control	Efficient Trading
1989 Coal Prices and Technology	\$2.67	\$1.90
1995 Coal Prices and Technology	\$2.23	\$1.51
1995 Coal Prices and 2010 Technology	\$1.82	\$1.04

Source : Carlson *et al.* (2000). Table 2, p. 1313

The changes in relative fuel prices and technology between 1989 and 1995 lowered costs by about 20% and CBCP's preferred estimate for 2010, which maintains 1995 relative fuel prices but extrapolates the 1985-94 rate of technological progress to 2010, reduces predicted costs by another third. The assumption of continued technological change also explains the difference between the CBCP and MCA estimates of Phase II annual cost, since the latter does not make any allowance for this factor.

To summarise, most of the explanation for the lower than expected cost of Title IV is attributable to changes in the nature of the proposed controls, from prescribing technology to the flexibility of a cap-and-trade system, and to changes in related sectors of the economy that were causing SO₂ emissions to be reduced anyway. As can be seen by comparing cells in Table 3.1, the difference in total cost between a relatively benign command-and-control alternative and fully efficient trading accounts for a relatively small part of the difference from the earliest cost estimates, which remained for better or worse stuck in many observers' minds.

The impression of dramatically lower costs was reinforced by the price of SO₂ allowances, which has been the most visible manifestation of cost to most observers. No one predicted the allowance prices of \$100 and even less that occurred in late 1995 and for most of 1996. Most predictions of early Phase I allowance prices ranged between \$250 and \$400, prices that have yet to be realised. Furthermore, many casual observers remembered only the predictions of Phase II prices, usually after the bank had been drawn down, which ranged from \$500 to as much as \$1000. The very low, early 1996 allowance prices may have reflected an over-reaction to the correction in early expectations of

market conditions; but, with eight years of experience with SO₂ allowance trading, there seems little doubt now that changes in technology and the availability of low sulfur coal fundamentally changed the quantity and cost of abatement that would be required to comply with Title IV and shifted allowance prices commensurately lower.

The Extent of Cost Savings from Trading

The principal area of disagreement among analysts about the economic efficiency of the programme concerns whether the full cost savings potential of emissions trading is being achieved. The point in dispute concerns the effect of cost-of-service regulation on the incentives of electric utilities to engage in trading with each other. The argument takes two forms: first, that conventional cost-of-service regulation provides no incentives to trade in the external market, since the gains would be passed on to rate-payers and losses might not be recoverable; and second, that public utility commissions have adopted policies that encourage sub-optimal choices by individual utilities, such as to scrub local high-sulfur coal in order to protect in-state jobs (Bohi and Burtraw, 1997; Rose, 1995; Rose, 2000). Research that simulates the effect of asymmetric incentives and other potential regulatory distortions indicates that compliance costs could be increased several-fold (Fullerton *et al.*, 1997; Winebrake *et al.*, 1995).

Empirical research tending to confirm this effect has been published. The most striking result was that in CBCP which found that the actual cost of compliance with Title IV in 1995 and 1996 was slightly higher than the cost of compliance under a benign command-and-control alternative (quantity caps equal to allowances at each affected unit). Moreover, their estimate of total cost with fully efficient trading was some \$200-\$250 million lower. This finding indicated that the unrealized cost savings were substantial and implied that emissions trading had not resulted in any cost savings in the first two years of the programme. The authors were quick to note that the volume of emissions trading was increasing and to state that they did not expect the apparent forsaking of the gains from emissions trading to last. More recently, Arimura (2002) has published research supporting the view that public utility commission regulation influenced abatement choices and contributed to low allowance prices.

The contrasting point of view is associated with researchers at MIT and is stated most completely in MCA, although also published in earlier articles and working papers (Joskow *et al.*, 1998; Schmalensee *et al.*, 1998; Ellerman and Montero, 1998; and Bailey, 1996). Here, the findings are that a reasonably efficient allowance market emerged as early as mid-1994; trading volumes have increased significantly, even in the early years; the effect of state PUC rulings on trading activity is insignificant; and that cost savings have been realised.

Much of the contrast between these two interpretations is a matter of tone, although substantive differences do exist concerning the effect of PUC regulation on emissions trading. It is beyond the scope of this paper to explore these differences in any detail, but a reader not already familiar with this debate should keep several points in mind.

First, the argument on cost savings is as much one of whether the glass is half full or half empty. The MIT group makes no estimate of what the full cost savings might be and allows that some cost savings are undoubtedly unrealised, but they emphasise that cost savings have been realised and that no market is perfect. The MCA estimate of the cost savings in the early years of Phase I (\$350 million, about half the observed cost of compliance) is derived from observed data assuming that the data reflect nearly efficient choices by abaters. In other words, this particular estimate assumes away the problem insisted on by the other school. This particular estimate was developed to discourage the then current views that the cost savings from emissions trading under Title IV were much greater. With the exception of the CBCP finding, the other camp does not dispute the existence of cost savings from Title IV. For instance, Bohi and Burtraw (1997) refer to the “puzzle” of cost savings with limited

trading and Rose (2000) concludes that Title IV shows that “trading mechanisms appear to be robust enough to allow substantial savings...to occur even when faced with less than ideal conditions.” The problem with the accuracy of the scrubber costs in the CBCP finding has already been mentioned, but even setting this aside, the focus in CBCP is more on quantifying the extent of unrealised cost savings as it is insisting that their less costly CAC alternative is realistic.⁸ Thus, one camp tends to emphasise the short-fall, while the other stresses the achievement. Still, a difference remains concerning magnitude. The difference is perhaps more aptly whether the glass is nearly full or only half full.

A second point to be kept in mind is that the debate about regulatory influence is at bottom one about how public utility regulation works in practice. Although not so far publicly stated, the MIT group would not dispute the theoretical effect of the alleged influences; their contention would be that the theory of regulation applied is over-simplified and not representative of the performance-based, rate-making as practiced in the 1990s. The only direct empirical test of the hypothesis of significant regulatory influence on emissions trading is Arimura (2002), which is unsatisfactory in interpreting a difference between the abatement decisions at Phase I units owned by the Tennessee Valley Authority, a non-profit, federally chartered utility, and Phase I units owned by PUC-regulated utilities as confirming a hypothesis concerning a predicted difference in abatement behaviour between profit-maximizing firms and regulated electric utilities.

V. Environmental Effectiveness

The arguments in favour of emissions trading programmes always assume that trading will not jeopardise environmental effectiveness, and this is invariably the main concern of environmental groups and those who tend to be sceptical of emissions trading. The experience with Title IV has provided no grounds for concern about environmental effectiveness; in fact, the experience suggests that environmental performance may be better than that experienced with command-and-control analogues. This section of the paper addresses this point, adduces the evidence indicating greater environmental effectiveness, and provides some tentative explanations for this result.

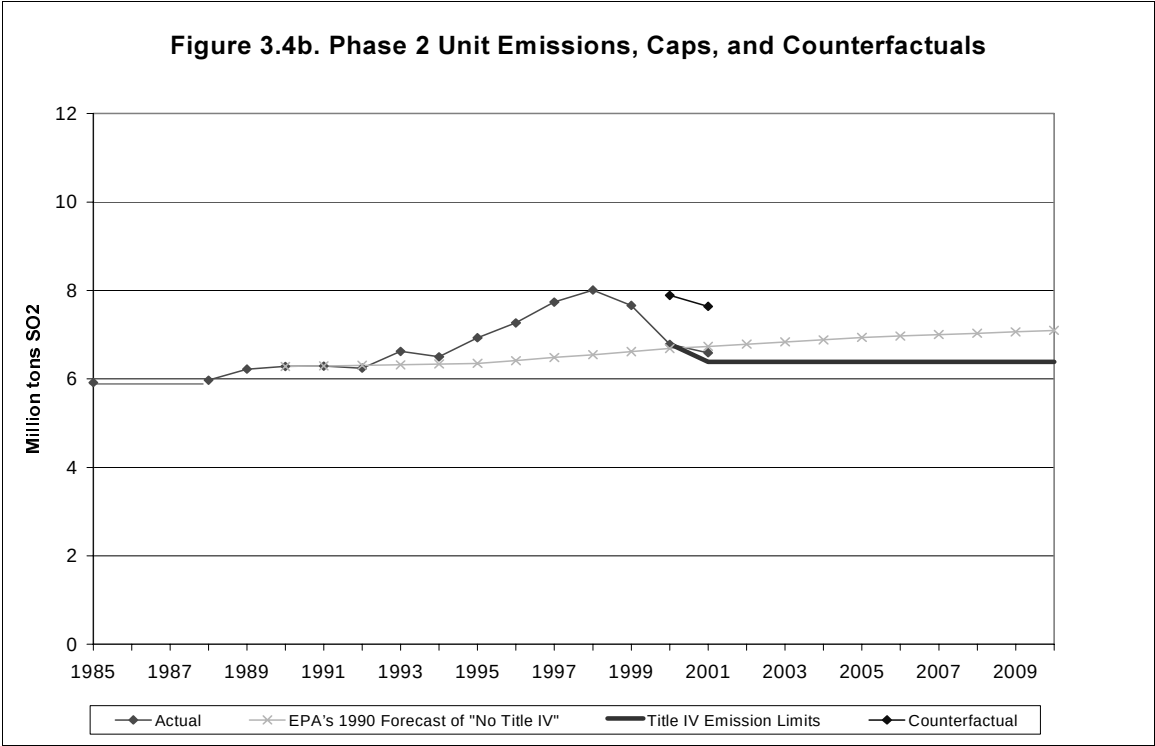
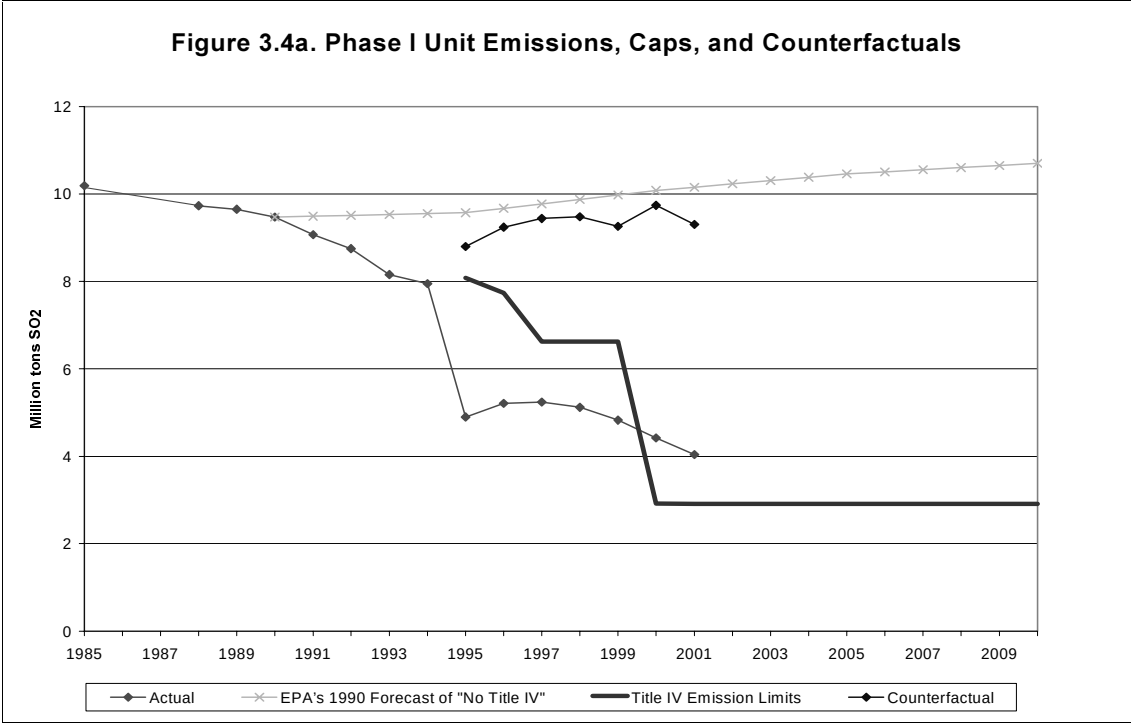
An important first issue in evaluating environmental effectiveness is identifying the appropriate metric. The acid rain motivation of this programme would suggest that an appropriate one would be the amount of wet deposition, or even the acidity of lakes and forests in sensitive regions; however, the most obvious and easily measured metric, total emissions, is the one typically used.

No doubt surrounds the issue of whether SO₂ emissions have been reduced.⁹ The two panels of Figure 3.4a shows actual emissions, the caps, and an estimate of counterfactual emissions for the 375 units first subject to Title IV in 1995 and for the much larger cohort of units that have been subject to Title IV since 2000. For both the Phase I and Phase II cohorts of units, the largest annual emission reduction is made in the first year, when the affected units first incur a cost for every ton of emissions. Given the phased-in nature of the requirement facing the Phase I units and the ability to bank, the annual reduction by these units was much greater than required. The annual reduction of emissions in 1995 was 3.9 million tons and that quantity of abatement has increased steadily and now stands at 6.3 million tons in 2001. Banking implies that emissions in the first years of Phase II will be greater than the allowances issued for these years, but the appropriate metric is the cumulative reduction since

8. Still, their CAC counterfactual is identical to the one assumed in MCA, which is found to cost about 50% more than the observed cost of compliance. Also, the methodology adopted by CBCP would attribute the same change in scrubber cost to the CAC alternative so that the finding of no cost savings would still hold.

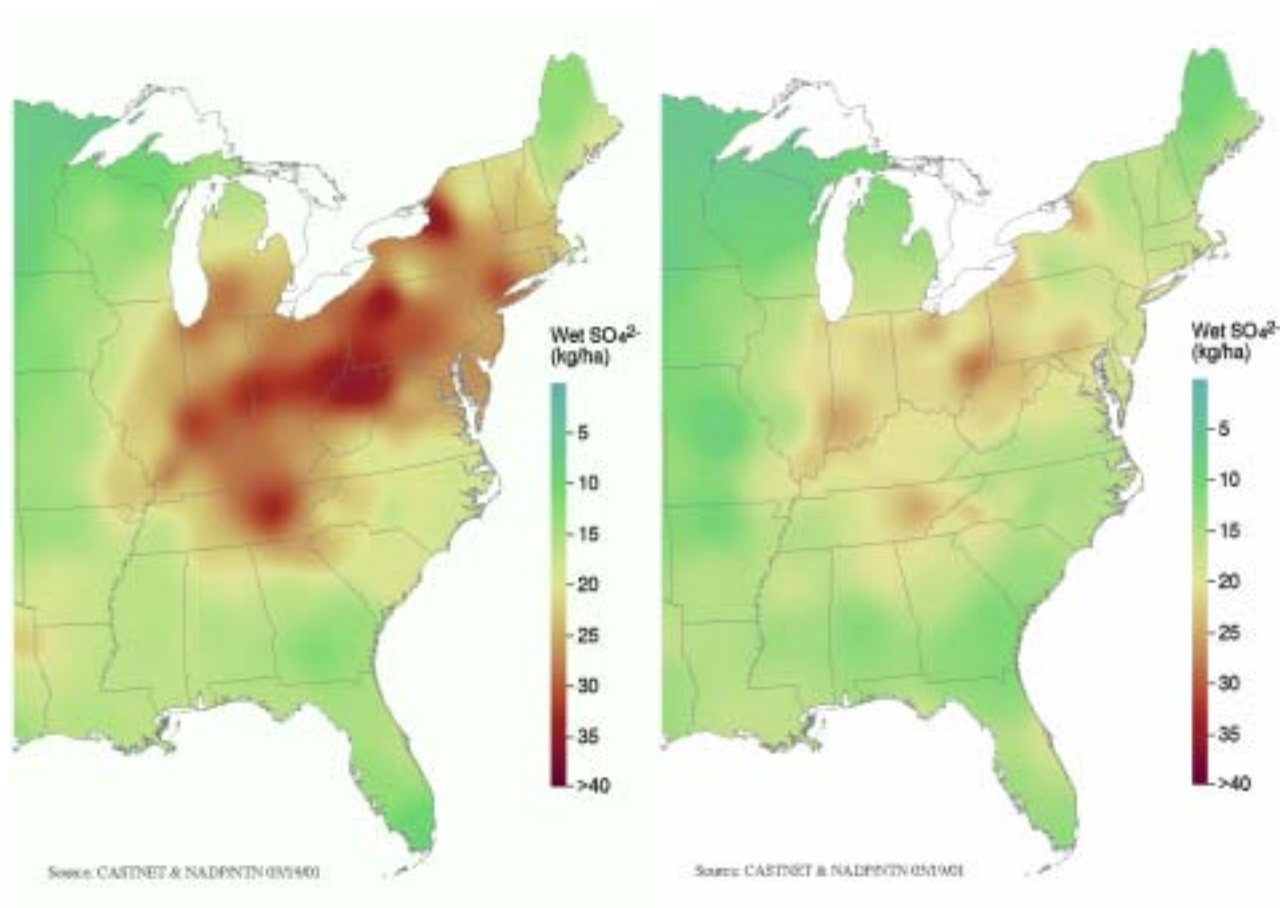
9. Suggestions to the contrary, such as those contained in *Darkening Skies* (Clear the Air, 2002), a publication of the New York Public Interest Research Group, are misleading in citing specific plants and comparing 1999 emissions with 1995 emissions, that is, after the large first-year reduction of emissions caused by the onset of Title IV.

1995, which has been 33.7 million tons, about 29% more than the 26.1 million tons that would have been required as of 2001 without banking. By the end of Phase I, the actual cumulative reduction was twice what was required, and that ratio will now decline steadily to 1.0 when the accumulated Phase I bank will be exhausted, probably in the second half of this decade.



The significant and accelerated reduction of emissions implies that the deposition of acidic particles has also fallen. A recent progress report from the U.S. Environmental Protection Agency (USEPA, 2002) reports that all of the conventional indicators relative to SO₂ have declined markedly because of the SO₂ Cap-and-Trade Program. Figure 3.5 provides a graphic illustration of the change in wet sulfate deposition in the eastern U.S. between the late 1980s and the late 1990s.

Figure 3.5: Monitored Reduction in Wet Sulfate Deposition



Similar diagrams could be shown for ambient concentrations of SO₂ and sulfate concentrations in the atmosphere, both of which have fallen generally across the Northeast and mid-Atlantic regions and in some places by as much as 50%. Sulfate concentrations in lakes and streams have declined significantly in all monitored regions of the Eastern United States, except Virginia, and in some areas, notably Pennsylvania and the Adirondacks, the acid neutralizing capacity of the soil has begun to increase, which is an indication of the beginning of recovery in ecosystems suffering from acidification.

Another aspect of the environmental effectiveness of the SO₂ cap-and-trade Program is the extent of compliance. With the exception of a few very small, new gas units in 2000, all generating units have been in compliance with Title IV requirements in all years. This record of virtually 100% compliance is not encountered with command-and-control regulation under which sources not infrequently receive various forms of dispensation that have the effect of delaying and sometimes permanently relaxing the applicability of the standard. The reason is that a single standard imposes greater costs on some than on others because of differing site-specific considerations and these firms pleading unique

hardship petition for administration relief that is often granted. Although such relief may be justified in the interest of equity, compensating tighter standards are not imposed on firms facing relatively less onerous costs and these latter never step forward to assume a greater cost burden in the interest of equity, nor are regulators able to identify who they are and thus to impose compensating, more stringent standards on them. The information asymmetries between regulator and regulated in CAC systems effectively lead to a form of adverse selection that makes the standard less effective than it otherwise would be.

This problem is avoided in a cap-and-trade system for two reasons. First, the market removes the rationale of unique hardship since the greatest burden borne by any is the price of an allowance; and, in a market with many buyers, no single one can claim to be uniquely disadvantaged. Second, the market provides at once a cheaper means of relief and the offset that preserves environmental integrity. Nothing prevents a firm from petitioning for relief from the requirement to surrender allowances, even if the grounds for doing so are weak; however, doing so can be costly and a market makes it cheaper simply to pay another to make the compensating reduction. In a sense, the ability to trade, and the market that it implies, renders special pleading uneconomic.

A frequently voiced worry about the environmental effectiveness of emissions trading programmes concerns “hot spots.” This phrase refers to the potential in a trading system for emission reductions to be transferred away from areas where emissions cause greater damage to those where the emissions cause less damage. Well-designed programmes would not have this problem since emissions would not be traded unless they had equal environmental effect; however, real programmes contain unavoidable compromises and the SO₂ programme is no exception. The enabling myth of the SO₂ programme is that location does not count, when in fact from the standpoint of acid rain effects, location obviously does. The fear in the acid rain programme is that emissions in the Midwest would not be reduced if utilities in this region could pay others located in parts of the country with little impact on the Northeast to reduce on their behalf.

This fear has proved to be unfounded (Swift, 2000). Sources in the Midwest have provided about 80% of the emissions reduction achieved in Title IV while accounting for about 55% of emissions in 2000. It may be argued that emissions from the Midwest are still too high, but it can hardly be argued that emissions trading has allowed sources in the Midwest to avoid abating. A tendency to autarkic compliance in initial planning and a programme incentive to scrub early also encouraged reductions in this region, but the more important reason appears to be that the cheapest abatement is to be found where the largest sources are located.

This happy result is not accidental. Most deep abatement technology, like scrubbing, is capital intensive and the per-ton cost depends upon how many tons are removed per MW^e of capacity. Higher utilization and higher sulfur content of the coal being burned means more tons of abatement over which the fixed capital cost can be spread and lower total cost per ton. Thus, where capital-intensive, deep-abatement technology is an option, market systems will direct abatement to relatively larger and more heavily utilised sources with relatively high sulfur coal. And, if these sources are the most damaging from an environmental standpoint, the experience with Title IV suggests they will be cleaned up first and that hot spots will not appear.

***Voluntary Aspects of Title IV*¹⁰**

Title IV had several provisions that allowed sources of SO₂ emissions outside of the cap to opt-in to the programme. Such features are attractive as a further means of lowering costs without

10. The discussion of this section is based largely on the work of Juan-Pablo Montero (Montero, 1999, and Montero, 2000), which is summarised in chapter seven of *Markets for Clean Air*.

jeopardizing programme objectives if sources that are not initially included are able to provide cheaper abatement to sources that are. In the case of Title IV, certain utility sources that were not required to be under the cap until Phase II could opt-in to Phase I, and non-utility sources that were otherwise not a part of the programme could do so in either phase.¹¹ The response of these two groups was very different: many eligible utility sources opted-in, while few industrial sources did so. The response of the utility sources also revealed an unavoidable trade-off between the economic and environmental objectives of the basic programme.

Over 200 electric utility units opted-in for one or more years of Phase I, and 110 of them participated in all five years. In contrast, only a few industrial sources chose to opt-in to the programme. The different response is largely explained by the differences in transaction costs for each category of participant (Atkeson, 1997). Industrial sources that considered participation but decided not to do so cited the costs of monitoring as the largest consideration. Moreover, the few that did participate already had monitoring equipment in place as a result of other environmental requirements or otherwise did not need to install monitors.¹² This obstacle was not faced by eligible electric utility units because all sources subject to the SO₂ cap-and-trade Program were required to install a continuous emission monitoring system by 1995 regardless of whether the unit was required to participate in Phase I, which began in 1995 or in Phase II, which began five years later. Another factor contributing to the different response is that utilities owning the units eligible for becoming substitution and compensation units in Phase I were already incurring the overhead costs of managing emissions and accounting for allowances, while industrial sources would have to undertake these new responsibilities. A final factor was that electric utility units did not have to undertake the expense of establishing a baseline. A final factor contributing to the differing response is that the number of allowances that would be granted to eligible electric utility units was pre-determined by a set of mathematical formulae that were similar to those used for units required to participate in Phase I. As a result of all these factors, the additional costs of participation were very low for eligible electric utility units and a significant number of them volunteered.¹³ In contrast, industrial sources would have had to incur the costs of monitoring emissions, establishing a baseline, and keeping track of allowances and emissions. These transaction costs were greater than the potential gains from trading that would have been possible through voluntary participation.

While the high rate of voluntary participation in the SO₂ cap-and-trade Program by electric utilities was heartening, an analysis of which eligible units opted in and which did not reveals a strong element of adverse selection, which resulted from the impossibility of specifying a true contemporaneous baseline (Montero, 1999). In theory, sources opting-in would receive allowances equal to what emissions would have been without participation. The pre-specified baseline, which

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11. The legislative and regulatory provisions for industrial units are known as the Industrial Opt-in Programme and utility units fell under the substitution or compensation provisions; however, all are referred to here as voluntary or opt-in participants. Electric utility units eligible for opting-in to Phase I were those owned by utilities with other units mandated to be part of Phase I. Provisions were included for opting-in units owned by a utility without Phase I units through contract with a utility having Phase I units, but these contract provisions were little utilised.
 12. For instance, in one case, an electric utility subject to the programme undertook to provide steam and power to an industrial facility thereby allowing that facility to shut down the boilers it had previously used to generate electricity and steam. Allowances equal to what the closed down facilities would have produced in supplying the ongoing needs of the industrial facility were then awarded to the electric utility providing the facility's power and steam needs.
 13. A further consideration was motivating electric utility participation was the NO_x grandfathering provision. Units with certain types of boilers could be grandfathered from Title IV's Phase II NO_x emission limits if they participated in the SO₂ programme in 1995. While many did, these units generally did not receive excess allowances and were not part of the adverse selection problem that characterised most electric utility opt-in units.

greatly reduced transaction costs, relied mostly on 1989-90 data; however, changes in coal markets and in the utilisation of electric generating units in the intervening years caused the true counterfactual emissions for eligible units in 1995-99 to be different. Thus, units that had already switched to lower sulfur coal for purely economic reasons because of changes in coal markets tended to opt-in and to receive some allowances in excess of their true baseline. And those who might have had low-cost abatement to offer but whose emissions had risen above the pre-specified baseline tended not to opt-in since they would incur the costs of reducing emissions to the baseline before they would receive any benefits from emissions trading. The end result was that the units opting in were not so much low-cost abaters, although some may have been, as they were units that were abating anyway.

This problem of adverse selection was exacerbated by allowing the owners of eligible units to wait until November 30 of each year to decide whether to opt-in for that year and to take the unit out of the programme in the following year if opting-in would be disadvantageous. While many eligible units remained in the programme for the entire five years of Phase I, a number of units can be observed opting in and out according to whether emissions in the particular year were higher or lower than the allowances they would receive as result of opting in.

While the evidence of this selection bias is very strong, the environmental effects from the loosening of the Title IV cap must be kept in perspective. The number of allowances that could be considered excess amounted to only 3% of the total issued during 1995-99 and the inflation of the cap during Phase II, when these allowances will be used is only about 2%.¹⁴ These magnitudes are not great and they cannot be said to have threatened the overall integrity of the SO₂ cap. In addition, many of the units opting in also abated emissions in response to allowance prices and thereby contributed some cost savings to the programme. Whether these cost savings were greater than the reduced environmental benefit depends greatly on the assumption about the true but unobservable baseline. In summary, it is hard to avoid the conclusion that the environmental damage was not great, but neither was the economic benefit, and that on balance, the voluntary features of Title IV were not worth the extra administrative effort.¹⁵

VI. Dynamic Effects

Theoretical work has long predicted that market-based instruments, such as a cap-and-trade programme, would provide greater impetus to innovation than command-and-control regulation, and thus add another cost-reducing attribute to these instruments (Magat, 1978; Milliman and Prince, 1989). Title IV has provided the occasion for testing this theoretical prediction and there is plenty of anecdotal evidence of what could be interpreted as innovation. Nevertheless, there is only one study that has attempted to address this issue rigorously and its results provide some confirmation, but not much (Popp, 2001). It may be still too early to be able to test the hypothesis confidently; and, under the best of circumstances, the difficulty of disentangling the effects of the regulatory instrument from exogenous technological change is great. Accordingly, in this section, the term, dynamic effects, is interpreted broadly to encompass factors other than the direct trading of emission rights that contribute to lower compliance cost.

14. Since opt-in units were required to meet the same compliance requirements as units initially included under the cap, most of the allowances issued to opt-in units in Phase I were needed to cover their own emissions during Phase I.

15. See Ellerman *et al.* (forthcoming) for an argument that this conclusion, which results from a balancing of costs and benefits, ought not to be carried over to potential applications of emissions trading for the control of greenhouse gases.

In considering dynamic effects, it is natural to focus of flue gas desulfurisation, or scrubbers, since they are capable of removing 95% or more of SO₂ emissions from the stack, they are commercially available and widely used, and they are costly. Moreover, the total costs of scrubbing for the Title IV scrubbers installed at the beginning of Phase I has been less than predicted and a second cohort of Title IV scrubbers that have come on line at the start of Phase II have shown even lower cost. The key components of this change in cost are given in Table 3.2.

Table 3.2: Evolution of Scrubber Costs

	Ex Ante Phase I	Ex Post Phase I	Phase II
Initial Capital Cost (\$/KW ^e)	\$240	\$249	\$150
Tons SO ₂ Removed per MW ^e	99	137	137
Per ton Fixed Cost (\$/ton)	\$273	\$206	\$124
Fixed O & M Cost (\$/ton)	\$75	\$15	\$15
Variable O & M Cost (\$/ton)	\$116	\$65	\$65
Total Cost per ton (\$/ton)	\$464	\$286	\$204

Source: MCA, Table 9.3 at p. 236 and discussion on p. 240.

The costs of scrubbing can be broken down into three components: 1) the initial capital cost, conventionally expressed as dollars per kilowatt of capacity, 2) the tons of SO₂ removed per unit of capacity in a year, which depends on the sulfur content of the coal and the utilization of the scrubber during the year, and 3) the O&M costs, which are often expressed as dollars per kilowatt or cents per kilowatt-hour, but which can be and are more properly stated for this purpose as dollars per ton removed. The third row, per ton fixed costs is simply the annual amortisation of the initial capital cost (first row) divided by tons removed (second row). The sum of fixed cost and fixed and variable O&M costs (fourth and fifth rows) determines the last row, total cost per ton.

Ex ante estimates for the cost of scrubbing a retrofitted Phase I unit typically fell between \$400/ton and \$500/ton, but ex post average cost has been below \$300 a ton. This average masks huge variation, from a few units with apparent costs higher than \$500/ton to several with costs around \$200/ton. As shown in Table 3.2, the calculated 33% reduction in average cost was due not to lower initial capital costs, which were as expected, but to 1) 25% higher utilisation of the retrofitted units and 2) a halving of operating and maintenance costs from what had been predicted. Fixed and variable operating costs were lower mostly because of improved instrumentation and control, which reduced the parasitic loss of power and manpower requirements, and it is probable that this improvement was a reflection of broader changes in information technology that were occurring throughout the economy.

The more interesting change from the standpoint of the effects of Title IV is the increase in tons removed per megawatt of capacity, which was due entirely to an increase in utilization from the predicted 65% of total hours to 85%. This shift in dispatch reflected the effects of the sulfur premium in coal markets that appeared with the start of Title IV across the entire sulfur gradient, instead of being restricted to “compliance” coal, that required in generating units meeting the pre-1978 NSPS by burning coal with less than 1.2 #SO₂/mmBtu. This premium appeared because the ability to substitute allowances for switching to lower sulfur content meant that allowance prices and the sulfur premium for less sulfur content would tend to equality across the entire range of sulfur content.

This differentiation in the prices of coals having more than 1.2#SO₂/mmBtu had two effects that increased the utilization of units with retrofitted scrubbers and thereby lowered the per ton fixed cost of scrubbing. Since allowance prices have been higher than the variable cost of scrubbing, a scrubbed unit has lower marginal cost for generating electricity than an unscrubbed unit, if all else is equal. The second effect reflected the change in fuel cost due to the new sulfur premium. This is the more

important effect because fuel costs constitute a larger proportion of the variable cost of generating electricity. Units that had not retrofitted scrubbers, typically burning mid- to low-sulfur coals, found themselves facing not only higher marginal abatement costs, but also higher fuel costs relative to scrubbed units, which would typically burn the higher sulfur coals that were now cheaper relative to coals with lower sulfur content.¹⁶ Both of these effects caused units with retrofitted scrubbers to move up in the merit order of dispatch and to increase their utilization relative to what it would have been without scrubbing.

After the first cohort of Phase I scrubbers were built, vendors touted a reduction in capital cost for follow-on scrubbers, and these claims became real in 1998 when allowance prices rose to \$200 and scrubber retrofits were announced for eight additional units, which are now online. Several of these units had initial capital costs around \$100/KW^e (which implied total costs per ton of SO₂ removed below \$200); however, these units were achieved because of economies that were made possible by previously installed scrubbers at other units at the same generating plant. The total cost indicated in Table 3.2 for Phase II scrubbers provides a good estimate of the initial capital cost without these economies and of long-run marginal cost of SO₂ removal by scrubbing, but that cost will rise as the scrubbers are retrofitted to less utilised units and units that do not have the option of switching to a cheaper, higher sulfur coal (Ellerman and Joskow, forthcoming). Even so, it is clear that there has been a large reduction in the cost of scrubbing, and the question is whether this can be attributed to Title IV.

The only research so far to address this question explicitly is Popp (2001) who compared patents relating to scrubbers from the early 1970s through 1997 with scrubber performance as reported in annual submissions to the Energy Information Administration. He finds that the passage of the 1990 Clean Air Act Amendments did not increase the level of innovative activity, and that in fact it fell somewhat, but that the nature of innovation did change in a more environmentally beneficial way. Throughout the period, the continuing level of innovative activity led to lower operating cost, but the patents granted after 1990 are associated with an improvement in removal efficiency that had remained constant previously. Popp's finding conflicts in part with those of two other studies of changes in scrubber technology (Bellás, 1998; and Taylor *et al.*, 2001). Bellás examined the same cost data as Popp but only through 1992 and found "no significant progress...in abatement technology," which he associated with "the small incentives for innovation [associated with] the form of regulation typically used in the U.S." Taylor *et al.* (2001) examine a slightly different question in seeking to determine the relative efficacy of R&D spending and regulatory constraint in inducing innovative activity related to scrubbers, and in doing so they find the same decline in patent activity as Popp but a continual increase in removal efficiency as well as a steady decline in capital cost, both of which are attributed to "learning by doing."¹⁷ These interesting but not entirely consistent results concerning the trend in scrubber costs do not provide very solid ground for assuming that emissions trading has a greater effect on innovation than other instruments.

While scrubbing can be considered the backstop technology for SO₂ abatement, it is not the only way, and it accounts for relatively less (40%) of the total reduction in SO₂ emissions in Title IV than switching to lower sulfur coal. Cost reductions in switching are not as easy to document, since switching does not attract the same attention as installing a scrubber, but cost-reducing changes can be inferred, most of all in the ability of boilers built to fire bituminous Mid-western coals to accommodate lower sulfur, sub-bituminous coal from the West. It was always recognised that these units could be

16. This effect applies only to scrubbed units using high sulfur coal. Unscrubbed units burning higher sulfur coals would pay less for fuel but require more allowances and on balance enjoy no advantage over unscrubbed units burning lower sulfur coals.

17. Popp (2002) and Taylor *et al.* (2001) use the patent data in different ways. Popp constructs a "stock of knowledge" using various diffusion and decay assumptions as the independent variable while Taylor *et al.* rely on the annual count of patent grants.

converted to the use of sub-bituminous coals, but the higher water and ash content of the latter would lead to a significant derating, or reduction, in the generating capacity of the unit. As a result, it was expected that the predominantly high-sulfur burning units in the Midwest would either install scrubbers or switch to low-sulfur bituminous coal produced in the Appalachian region. As the effects of rail deregulation increasingly reduced the significant transportation component in the cost of *western* low sulfur, sub-bituminous coals delivered to the Midwest, power plant engineers began to experiment with blending these coals with locally produced high-sulfur bituminous coals. While a 100% conversion to a sub-bituminous would result in a derating, it was equally evident that a 1% blend would have little effect and the operational question became at what mixture the unit would experience a reduction in operating efficiency. In what must be seen as a triumph of continuous thinking, the answer emerged that, depending on the unit and the coals being blended, mixtures of up to 60% of low-sulfur, sub-bituminous coal (and sometimes higher) could be used without significant derating in the generating capacity of the unit. The effects of this development extended well beyond the Midwest. Lower costs of abatement in the Midwest implied lower allowance prices, lower sulfur premiums for Appalachian low sulfur coal, and consequent lower costs for switching in regions to the East beyond the economic frontier for western low-sulfur coals.

Other cost-reducing changes that might be termed innovations can be observed upstream of the power plant in response to the sulfur premium. Mid-sulfur coal mines were developed in the Midwest where none existed before. These could supply a local coal at a price competitive with western blends, but when the only sulfur premium paid was for compliance coal ($<1.2 \text{ #SO}_2/\text{mmBtu}$) these mines could not compete with the lower cost but higher sulfur mines in the Midwest and were therefore not developed. A similar shifting downward of the average sulfur content of coal being supplied was observed in Northern Appalachia, the other high-sulfur coal-producing region, as mines exploited lower sulfur coal seams and a larger proportion of the sulfur was removed at preparation plants prior to shipping. These changes in coal supply to somewhat lower sulfur coals, which would still be considered mid- or high-sulfur coals, account for about 36% of the total reduction attributable to switching, or somewhat more than one fifth of the total.

A relevant question is whether these developments constitute innovation or simply diffusion of known technology. In general, these changes in the operations of power plants, mines, and preparation plants were not observed before and not expected. Previous regulatory instruments, which tended either to mandate scrubbers or to create artificial distinctions along a continuous sulfur gradient, provided no incentive for exploring or making incremental changes in operations that would reduce sulfur content. Title IV provided the incentive in the form of a premium extending across the entire sulfur gradient for any sulfur reducing change no matter how small or large. Whether these opportunities were known before to power plant operators, mining engineers, and prep plant operators and needed only the incentive to bring them forth awaits further research, but the answer will determine whether these innovative changes can be considered a change in the menu of technological options induced by Title IV or simply the diffusion of known techniques once the incentive was in place. Another possibility, which seems especially applicable to the experience in blending sub-bituminous and bituminous coals, is that what was initially diffusion led to further cost reductions through learning-by-doing.

One further contribution of Title IV to lower cost that does not involve innovation is noted in Burtraw (1996) and labelled cost savings without emissions trading. Burtraw noted that giving plants the ability to choose between scrubbing, switching, and purchasing allowances created a competition among suppliers of abatement that was not present before. The threat to purchase allowances implies some trading to be credible, but it would not require a fully developed market and even without this threat, the ability to choose between switching and scrubbing increased competition and contributed to lower costs.

What emerges from the experience with Title IV is that costs are lower for reasons beyond the ability to trade emission reductions among sources. Improvements in productivity were occurring throughout the American economy during this period and Carlson et al. (2000) find that unspecified, exogenous productivity improvement applied to SO₂ abatement as well and accounted for as much as 20% of the reduction in the cost between 1985 and 1994. Quite aside from this background trend, a variety of industry sources indicate that the ability to trade emissions, and actual trading, have had effects in upstream markets and on the choice of abatement technique that can be directly attributable to the flexibility that is inherent in market-based approaches to air emission regulation. Whether these changes, which often look like innovation, are true changes of the technical choices facing firms or simply the diffusion of known technology in response to the right incentive awaits further research. It is clear that costs are lower than expected for reasons beyond the extent of actual trading and that these changes were not expected.

VII. Other Costs and Effects

All air emission control programmes involve costs and effects beyond the directly observable abatement costs and the concomitant reduction in emissions. In the Acid Rain Program, administrative costs for both the regulator and the regulated are believed to have been less than in conventional regulatory programmes, but no comprehensive study has been conducted on this subject. The more important aspect of the programme's administration concerns the revolutionary change in the nature of the tasks that are now required of the regulator and the regulated (Kruger, McLean, and Chen, 2000).

The shift of regulatory instrument from site-specific mandates to cap-and-trade has been accompanied by a corresponding shift in enforcement from relatively labour-intensive but intermittent inspection to data-intensive but continuous measurement and accounting. When compliance is defined by adherence to some standard, it matters to the regulator what each source is doing to abate and a corps of inspectors is needed to check periodically on the performance of the regulated. In a cap-and-trade system, the requirement that allowances be surrendered for all emissions permits the regulator to be indifferent about each source's abatement, and therefore to do without the corps of inspectors (except to verify the accuracy of the emission monitors); however, the *quid pro quo* is continuous measurement and reporting of emissions. In turn, this requires the handling of more data and a greater focus on accounting than was true of more conventional regulation.

The hallmark of the new system of regulation is continuous emissions monitoring and these monitors impose a non-negligible cost on operators that is estimated at 7% of direct compliance cost (MCA, pp. 248-50).¹⁸ As shown by Atkeson (1997) in her study of Title IV opt-in candidates, this cost can be a significant deterrent to voluntary participation. In the case of electric utility units subject to Title IV, continuous emissions monitoring and reporting was mandated for SO₂, NO_x and CO₂. To the extent that the information from these systems is used for the implementation of other air emission control programmes, such as the Title IV NO_x averaging programme or the Northeastern NO_x Budget Programme, or that the data provide benefits aside from compliance uses, this cost should be shared with those other uses. Nevertheless, the experience with Title IV makes clear that the cost of this prerequisite for emissions trading is not negligible.

The administrative costs incurred by EPA are significantly less and of a different nature than would be incurred with a conventional regulatory programme. McLean (1997) states that EPA

18. McLean (1997) provides a considerably higher estimate of monitoring cost, \$200 to \$300 million per year for monitoring at all sources. The MCA estimate of 7% of direct compliance cost for the Phase I units in 1995 implies annual cost of around \$50 million. Since direct compliance costs are generally less for Phase II units, the percentage for the entire programme is probably higher than 7%.

expenditure to implement and operate the Acid Rain Program in the first five fiscal years following enactment (FY91-95) amounted to \$44 million out of a total \$1.09 billion required to implement all the programmes in the 1990 Clean Air Act Amendments. Although there is no estimate of the EPA expenditure that would have been required to implement and operate an equivalent command-and-control programme, the importance of the Acid Rain Program are certainly greater than 4% of the total achievements of the programmes associated with the 1990 Clean Air Act Amendments. Kruger, McLean and Chen (2000) describe the significant data handling requirements that are now faced and they suggest that this would not have been much more costly before recent advances in computing and data management. Also, the high degree of automation and the reduced inspection requirements have greatly reduced personnel requirements. McLean (1997) notes that administration of the programme requires about 150 work-years at the federal, state, and local levels, which is about 1% of the 150,000 people working on air pollution control at the same agencies. As is the case with expenditures, this is a very small percentage for a programme that has certainly contributed more than 1% of the reduction in environmental damages from air emissions.

Although no researcher has attempted to address the issue, the administrative costs of the cap-and-trade programme for the regulated are not as clearly less than they are under conventional regulation. The cost of continuous emissions monitoring is the main item in this accounting. As is the case for the regulator, corporate administrative resources are shifted to emissions reporting and allowance management, but a good comparison of how these costs compare with what is required for dealing with inspectors and reporting under conventional command-and-control systems has not been made. It may not be any greater, but it is not clearly less. Whatever the case, regulated firms seem to be unanimous in expressing their preference for this new type of regulation, presumably because the gains in reduced, direct compliance costs more than offset whatever additional costs are involved in monitoring and allowance management.

Another notable achievement in the realm of other costs is the reduction in the transaction costs involved in trading. The creation of a standard unit of account in allowances and the lack of any review requirement for trading has avoided the very large transactions costs that limited EPA's earlier experiments with emissions trading (Ellerman *et al.*, forthcoming; Kruger, McLean, and Chen, 2000). The right to emit has been made into a readily tradable commodity and broker commissions are correspondingly low. This feature has, of course, greatly facilitated the development of a market and the concomitant cost savings.

Two effects of the Acid Rain Program that are not related to ancillary costs are also important. The first has been the creation of institutions with a continuing interest in emissions trading. The emergence of intermediaries, such as brokers, banks, and others who can offer trading and risk-management services, has already been mentioned. And, as is perhaps inevitable for any economic activity of note, an association has been formed, the Emissions Marketing Association, to promote emissions trading through a variety of educational, lobbying, informational, and other out-reach programmes. Finally, there seems to be no end to the conferences, meetings, and workshops that bring participants from the private and public sectors and academia together to discuss one aspect or another of emissions trading.

While this institutionalisation of emissions trading has occurred, somewhat of a backlash has also emerged recently as represented by Clear the Air (2002) and Moore (2002). The latter succinctly states the position of these groups: "trading ought to be rejected when proposed and repealed where it now exists" (p. 2). Both of these purported studies are lobbying documents occasioned by the Bush Administration's Clear Skies Proposal, which in addition to lowering the SO₂ cap by two-thirds and instituting national NO_x and mercury caps would effectively exempt units subject to these proposed caps from the best available control technology requirements of the existing Clean Air Act. Based on the experience with Title IV, one might conclude that this is a good trade-off, as has been advanced by

some academics (Ellerman and Joskow, 2000) and as suggested by the publications of some environmental organizations (Goffman and Dudek, 1995; Environmental Defense, 2000) and by researchers at some environmentally-oriented research organizations (Swift, 2000; Swift, 2001); but this is far from a universally shared view among the environmental community. The reasons for rejecting emissions trading are beyond the scope of this paper, but disdain for pollutant trading as morally reprehensible and concern for the loss of administrative discretion (and its many uses for non-environmental purposes) are always present. Although these attitudes may be viewed as a rear-guard reaction to an increasingly dominant consensus, they do find an echo on the editorial page of the *New York Times* and they have been translated into a law in New York that would restrict emissions trading. In what is perhaps an example of the new institutions, this state law has been struck down in the federal court as a violation of the interstate commerce clause of the U.S. Constitution in a motion for summary judgment brought by members of the Emissions Trading Association (CAMG vs. Pataki, 2002).

VIII. Conclusions and Implications

The experience with Title IV and, to a lesser extent, other cap-and-trade programmes marks a turning point in the regulation of air emissions in the U.S. This experience has shown that market-based incentive systems can reduce emissions as effectively, and even more so, and at considerably less cost than conventional command-and-control mandates. As a result, it has become virtually obligatory that any legislative proposal to limit air emissions in the U.S. include emissions trading. While the agreement of left and right in the political spectrum is not as complete as it may appear on the surface, there seems little doubt that emissions trading will play an increasing role in the regulation of air emissions in the U.S. and probably elsewhere.

The conventional wisdom holds that emissions trading will be necessary for new emission control initiatives and that the existing structure of command-and-control regulation is sacrosanct. Hence, all legislative proposals granting new authority to regulate air emissions include emissions trading; yet, their passage has been no faster for this reason. The same issues of cost and benefit and the same imperatives of building a viable political consensus remain. While legislative proposals that include emissions trading do not appear to be going anywhere fast, a less noticed and potentially more important change is occurring. Cap-and-trade systems are being adopted as a preferred means for achieving environmental goals for which ample legislative and regulatory authority already exists to implement conventional regulatory measures. Examples are the RECLAIM and Northeastern NO_x Budget Programmes, as well as the NO_x SIP call. In effect, these new cap-and-trade programmes have quietly supplanted the existing conventional regulatory system. This trend is in keeping with the increasing reliance on market forces that has become manifest in one regulatory domain after another over the past several decades and it indicates that the increased use of cap-and-trade programmes may occur as much through such incremental changes in the existing command-and-control structure as through bold new advances in the legislative domain.

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Chapter 4

THE DUTCH NUTRIENT QUOTA SYSTEM: PAST EXPERIENCE AND LESSONS FOR THE FUTURE

by

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I. Introduction

Trading in emission allowances based on a cap-and-trade system is broadly recognised in the theoretical literature as a cost efficient and environmentally effective instrument for emission reductions. However, there is considerable debate about the efficiency and effectiveness of these schemes in practice. Issues of concern are the potential impacts of market imperfections, the administration costs of allowance allocation and market establishment, and the competitiveness impacts on so-called 'exposed' sectors.

Tradeable pollution permits (TPP) do not play a significant role in agri-environmental policy. They are rarely discussed, in contrast to environmental taxes on, for example, fertiliser, pesticides or nutrient surpluses.² To our knowledge there is only one example of a TPP specifically for agriculture, namely the Dutch system of phosphate quota in animal production. The experiences with the Dutch quota system example could be very useful in view of the recent interest of including non-point source pollution from agriculture in trading programmes that traditionally only would cover point source pollution.³

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1. The views expressed in this chapter are those of the author and do not necessarily reflect those of the institutes with which the author is affiliated or the OECD.
 2. In Europe, both Sweden and Denmark have a fertiliser tax and the Netherlands has a tax on unallowable nitrogen and phosphate surpluses. Finland and Austria had fertiliser taxes before they entered the European Union. Outside the EU, there is a fertiliser tax in Norway.
 3. Woodward and Kaiser (2002) give an overview of water quality trading programmes in the US that include both point source and non-point source pollution. Fifteen cases are discussed in which trading is either in place or under development. Of the 11 existing programmes only 2 existed prior to 1989 and 7 have been started since 1996.

We proceed as follows. Section II presents a summary of the environmental problems in Dutch animal agriculture. In Section III an overview of the phosphate quota regime is given. Section 4 discusses the institutional background and methodology used in the available ex-post studies. Sections V-VII give an overview of the results, a comparison with ex-ante studies and the policy implications, respectively.

II. Environmental Problems in Animal Agriculture and Initial Policy Regulation

In Western Europe the expansion and intensification process in the pig, poultry and dairy sectors gathered pace during the 1960s and 1970s. In the Netherlands the expansion of animal numbers, specifically pigs and poultry, was particularly evident on the sandy soils in the eastern and southern part of the country. Specialised pork and broiler production was an ideal option to improve the labour productivity and farm incomes on the small farms in these regions. The increase in pork and broiler production was made possible by a growing international demand for animal products and a EU agricultural policy that favoured the import of feed.⁴ As a result, Dutch livestock producers, particularly in the pig (some 10,000 farms) and the poultry sectors (some 2,000 farms), developed intensive livestock operations on farms with small acreages. Between the early 1960s and the mid-1980s, the number of pigs and poultry increased by 10 million (+ 450 percent) and 50 million (+ 125 percent), respectively (Wossink and Benson, 1999).

Consequently, a manure surplus developed, particularly in the eastern and southern parts of the country. The national surplus of manure was about 16 million metric tons or 19 percent of all manure in 1987 (Tamminga and Wijnands, 1991).⁵ In terms of phosphate the surplus equalled 75 million kg (CPB, 2000). Excess manure leads to over application of manure on crops and consequent discharges into groundwater.⁶ It was estimated in the mid-1980s that the maximum EU standard of 50 mg nitrate per litre of groundwater was exceeded on 60% of agricultural land in The Netherlands (Becker, 1992).⁷

The first warnings about the polluting effects of manure surpluses on the environment were already being voiced in the 1970s.⁸ Growing political concerns in response to the alarming environmental developments in the 1980s, as outlined above, meant that some actions had to be taken

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4. Due to price support for cereals in the EU and the absence of import levies on imported feed stuffs, import of, for example tapioca, soy, citrus pulp and maize gluten became attractive for Dutch farmers, who exploited their proximity to the port of Rotterdam.
 5. Notice that the surplus calculation refers to the farm level. When manure surpluses at the aggregated level are discussed three aspects are to be considered: farms with surpluses, farms with deficits and different crops, and the willingness of farmers with deficits to accept manure. The value of manure will depend on the nutrient content, organic matter, cost of transportation, tolerance of crops, risks of spreading of diseases, etc.
 6. Over-application was most evident for silage corn, which was mainly grown as a waste disposal crop. In 1985, for example, the application rate for phosphorus exceeded the requirement of this crop, which is 200 kg P₂O₅ per ha, by a factor of four, on average. Furthermore, it was estimated that 300,000 hectares or 50% of the cultivated land on sandy soils was saturated with phosphorus. Phosphorus saturation of the soil receives little attention in the EU, except in The Netherlands. The same applies for ammonia emission.
 7. See de Walle and Sevenster (1998) for details on the extent of the problem, on regional differences and on how the Dutch situation compares with the rest of the EU.
 8. The early signals came from agronomists who pointed at the risks of eutrophication due to nitrate leaching, in particular (*e.g.*, Henkens, 1975). Their findings were rebuffed, however, because of the economic importance and political clout of the livestock sector. In particular, the Ministry of Agriculture and the farmers' lobby resisted and succeeded in postponing measures to address these issues (Frouws, 1997). Up until 1984 animal waste problems were mentioned in political discussions but no measures were taken (Tamminga and Wijnands, 1991).

(Frouws, 1997). The first policy action was the imposition of a moratorium, the Interim Law for Restriction of Pig and Poultry Farms (November 1984). This Law prohibited the creation of new livestock farms in the eastern and southeastern regions of the Netherlands and placed restrictions on the expansion of existing intensive livestock farms. However, farmers took full advantage of the expansion opportunities under this Law and further action was needed.⁹ Further action was also needed as the eutrophication of surface water and groundwater pollution became issues of international concern.¹⁰

The moratorium was followed by a more comprehensive approach with the objective to achieve a balance between production and utilization of the nutrients in manure by the year 2000. This objective was set out in the first National Environmental Policy Plan of 1989, to be implemented stepwise under a Three-Phase Plan covering the periods 1987-1990; 1991-1994 and 1995-2000. The figures in Table 4.1a identify the *original* targets of the Three-Phase Plan, the figures in Table 4.1b present the targets as actually implemented. At the outset, the phosphate content in manure was the target nutrient of the regulations.¹¹ In the first phase (1987-1990), the Manure Law and the Soil Protection Act¹² (both of 1987) replaced the Interim law. The Manure Law introduced the manure quota system and the manure book-keeping system. The main aim of the latter system was to ensure that actual phosphate production did not exceed the quota amount and to monitor manure application in view of the admissible rates in Table 4.1b (see Breembroek *et al.*, 1996).

In the second phase (1991-1994) ammonia was addressed through the Guideline on Ammonia and Cattle Farming under the Nuisance Act (now Environmental Management Act). This Act only allows expansion of livestock farms if acidification is below a threshold of 30 mole acid per hectare annually. Markets for ammonia quota were established in parts of the country through the development of county ammonia reduction plans.

III. Overview of the Quota System¹³

Initial Allocation

Since the enactment of the Manure Law on January 1, 1987 (Amvb, 1986a), Dutch legislation allows a total manure production from all animal sources of up to 125 kg of phosphate (P_2O_5) per

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9. A total growth of stock of 10% was justified on existing farms. Despite this restriction, the number of pigs actually increased *annually* by 7.5% between 1983 and 1987 (Tamminga and Wijnands, 1991). This increase was mainly possible because of the large number of building requests filed just before the moratorium (*cf.* CPN, 2002, p. 25).
 10. Important in this respect were (de Walle and Sevenster, 1998, p. 11-13, 30): (a) the statement of the second International North Sea Conference (1987) to reduce the nutrient supply to the North Sea by 50 %, and (b) the Paris Convention of 1988 which requested the 50% reduction by 1995 in comparison with 1985 levels.
 11. The reasons for focusing on phosphate related mainly to soil physics. The groundwater table in the Netherlands is rather high, being on average less than 1 meter below the soil surface. Saturation of soils with phosphate, with leaching into groundwater and eutrophication of surface waters was thought to be likely. Furthermore, it was assumed that a decrease in nitrogen problems would follow naturally from imposing limits on phosphate application. N application was calculated indirectly by assuming a fixed N: P ratio of 2:1 (Frederiksen, 1997). However, experience showed that this assumption was invalid and, starting in 1998, explicit standards for nitrogen application were established as well.
 12. The Soil Protection Act sets restrictions on: (a) on the application of manure with the aim to gradually reduce these admissible application rates, see Table 4.2, and (b) the period of the year the waste could be land applied (for details see Tamminga and Wijnands, 1991; de Walle and Sevenster, 1998).
 13. The larger part of this section is from Vukina and Wossink (2000).

hectare of land. Farmers producing more manure in terms of phosphate need additional registered animal-based manure production rights. The system of manure production rights was introduced in two steps: in 1987 for the production of manure from cattle, swine and poultry, and in 1992 for the production of manure from sheep, goats, ducks, foxes, nutria and rabbits. Each farm was grandfathered a so-called “reference amount” based on the actual manure production. Actual manure production for each individual farm was estimated by an inventory of animals and standards for the manure production for each specific animal category measured in kilograms of P_2O_5 per year¹⁴. These animal specific standards were calculated as the difference between phosphate supply (in feed, animals, fertilizer etc.) and phosphate removal (in meat, milk, eggs, animals, etc.). The residual is assumed to represent the phosphate content in manure of the specific animal category. For example, the phosphate standard for finishing pigs is 7.4, meaning that one finishing pig will produce 7.4 kg of P_2O_5 per year.

Assessments were made of all land either owned or leased long-term (minimum 6 years and officially registered) used for agricultural purposes in December of 1986 and again in December of 1991. The difference between the reference amount and the assessed acreage-based phosphate rights was used to establish a distinction between manure-surplus farms (with manure production in excess of 125 kg of P_2O_5 per hectare) and manure-deficit farms (with phosphate production below 125 kg / ha). A deficit farm could still increase animal production on the basis of unused land-based manure production rights. For a manure surplus farm such an increase in production capacity was possible only with an increase in the reference quantity of manure production rights through land acquisition.

**Table 4.1a. The *original* targets of the Three-Phase Plan:
maximum admissible application rates of animal manure in kg P_2O_5 / hectare/year.**

	<i>Phase 1</i> 1987-1991	<i>Phase 2</i> 1991-1995	<i>Phase 3</i> 1995-2000	<i>From 2000</i>
Grassland	250	200	175	125 ¹
Silage corn	350	250	175	125 ¹
Arable land	125	125	125	125 ¹

1. These figures later to be adjusted for crop nutrient uptake.

**Table 4.1b. Maximum admissible application rates of animal manure
in kg P_2O_5 / hectare/year as *actually* implemented**

	<i>Phase 1</i> 1987-1991		<i>Phase 2</i> 1991-1992	1993	1994	<i>Phase 3</i> 1995	1996-1997
Grassland	250		200	200	200	150	135
Silage corn	350		250	200	150	110	110
Arable land	125		125	125	125	110	110

Source: Jongbloed and Lenis (1998).

1987-1993: Trading in Land Related Manure Production Rights Only

From 1987 until 1993 the transfer of the manure production rights was severely restricted to prohibit a further exacerbation of the manure problem. The conditions are specified in the “Relocation Decision” document enacted on May 1, 1987 (Amvb, 1987). It states that the reference quantity is only

14. For the assessment of the number of animals different dates were used: December 31, 1986 for pigs, poultry and cattle, and December 1991 for sheep, goats, rabbits, ducks, foxes and nutria. (Amvb, 1986b).

transferable: (a) as part of the transfer of a whole farm,¹⁵ (b) with marriage and heritage, (c) with annulment of a lease contract for a farm, in which case the lessor was entitled to transfer the reference amount to another farm in his possession.

The limited possibilities for increasing the manure production rights of the farm and hence its production capacity, meant that expansion could have been only realised by means of land acquisitions. However, buying an additional hectare of land would result in a net increase in aggregate (animals plus land) manure production rights only for a deficit farm. For a surplus farm, buying additional land could cause a proportional amount of the existing reference (animals-based) quota to “sink” into the land-based quota with no net increase in aggregate manure production rights.¹⁶ So in fact, more animal production was possible only by starting a new animal farm or by starting an animal enterprise on completely non-used land quota, that is by relocating animal production to crop farming regions. Expansion of the existing livestock farms in the Netherlands came to a standstill in the regions where animal production has been traditionally concentrated, *i.e.*, in the South and East of the country. The regulation indirectly caused a freeze of the agricultural structure in those regions, hampering the adaptation and investment processes required for solving the national manure problem. To counteract these limitations, a new law regulating transfer of manure production rights was enacted on January 1, 1994 (Haerkens and Walda, 1994).

1994-1997: Relaxed Trading In Manure Production Rights

The main element of the new law regulating transfer of manure production rights was that the production rights became tradeable. For each farm the reference amount was converted into “manure production rights” (manure quota) to indicate the change in policy. In contrast to the homogeneous reference amount, manure production rights became highly differentiated with the goal to restrict the trading. This was done in three steps.

First, a farm’s total manure quota was officially divided into two parts: a land-based part and a non-land-based part. The first part amounts to 125 kg of P_2O_5 times the number of hectares of land on the farm, whereas the non- land-based part is calculated as the difference between a farm’s reference amount and the land-based quota.

Second, a farm’s non- land-based quota was allocated to specific animal categories reflecting the situation on an individual farm. Using the inventory figures on animals from earlier assessments of the reference amount, each farm’s total manure quota was partitioned into animal categories. This was accomplished by using a ranking scheme reflecting the extent to which keeping various categories of animals is truly land related. Three classes were established: (1) cattle and turkeys, (2) sheep, goats, foxes, nutria and ducks, and (3) pigs and chickens or broilers. In the Netherlands, hog, broiler and other small animal farms are confined animal husbandry operations, whereas a cattle farming is more directly land related. The ranking scheme was used to determine which animal category is associated with the non-land (tradeable) quota.

15. Under two important restrictions: the amount could only be bought and sold in its totality and the farm had to be continued as an autonomous enterprise at the same location as before.

16. As an example, consider a 3-hectare farm with 1,000 fattening pigs. As mentioned before, the animal specific transfer coefficient for pigs is 7.4 kg of P_2O_5 per animal per year. Therefore, this farm’s total reference amount is 7,400 kg of phosphate per year. Given that the land-based phosphate allowance is 125 kg of P_2O_5 per hectare, the total reference amount is composed of $3 \times 125 = 375$ kg of land-based quota and $7,400 - 375 = 7,025$ kg of animals based quota. Buying 1 additional hectare of land would increase the land-based quota to 500 kg but would at the same time decrease the animal based quota to 6,900 kg without changing the total available quota. Consequently, this farm would have to buy an additional 55.2 acres ($55.2 \times 125 = 6,900$) of land before its total quota available for production would go up.

As seen in the upper portion of Table 4.2, the available land quota (125 kg of P_2O_5 per hectare) will first cover the manure production from cattle and turkeys and then, if there is some land quota left, it will be assigned to the other two animal categories until the entire land quota is exhausted. The remaining difference between the farm's reference amount and the land-based quota becomes non-land-based and hence tradeable. As the result of this regulation, most of the tradeable quota ended up being allocated to the third animal category (pigs and chickens). The denomination of the tradeable quota is important because trading was restricted across animal species. The quota was made upward compatible in the sense that the third animal category quota can be used for the production of the second and the first category of animals. On the other hand, pigs and chickens can be produced only with the third category quota since using phosphate quota from any other animal category for pigs and chickens is not permissible. The rationale behind this rule was to prevent farmers from using other animal category quota to further increase swine production, which was perceived to be the source of the most serious environmental problems.

Third, the non land-based quota allocated to cattle, turkeys, pigs and chickens (broilers) was differentiated further to account for the improvements in feed conversion in these sectors. For each of these animal categories the "dormant quota", i.e., the difference between the original reference amount and the actual phosphate production was assessed.¹⁷ This was done for each farm individually. As seen in the lower portion of Table 4.2, the dormant part of the quota became non-tradeable. There were other restrictions that inhibited the trading of the production rights. For example, with each transaction 25% of the quota was retired. This reduction applied to all animal categories. In addition, the farmer who acquired additional quota had to certify that he had either sufficient land on his own farm to dispose of the total manure for the next two years or had a manure disposal contract with another farm.

Manure quota could still change hands through land transactions (lease or sale) with no reduction in the available quota. However, just as in the earlier period, if additional land was acquired by a surplus farm, the land-based quota that goes with it automatically 'sunk' into the animals based quota (see note 15 above). So, for a surplus farm, the acquisition of more land reduced the tradeable part of the quota by increasing the land-related share and reducing the non land-related share.

17. Information on the actual manure production for each surplus farm is available from the annual manure bookkeeping accounts. To assess dormant manure production rights for cattle, poultry and pigs, the highest manure production figures are selected from the three-year period: 1988-1990.

Table 4.2. Example of the assessment of the tradeable animal based quota

<i>Farm description:</i>				
• 9 hectares of land • Reference amount 2800 kg P₂O₅, of which 800 due to cattle/turkeys and 2000 due to pigs/chickens/broilers • Highest actual manure production in 1988-1990: 2200 kg P₂O₅, of which 500 due to cattle/turkeys and 1700 due to pigs/chickens/broilers				
<i>Calculation</i>	Total		cattle/turkeys	pigs/chickens/broilers
Reference amount	2800		800	2000
Land related quota (9×125)	<u>1125</u>	Allocation	<u>800</u>	- <u>325</u> (1125-800)
Non land related	1675		0	1675
Actual manure production	2200		500	1700
Land related 9 × 125	<u>1125</u>	Allocation	<u>500</u>	<u>625</u> (1125 -500)
Non-land related (tradeable)	1075		0	1075
Dormant (non tradeable)	600			600 (1675-1075)

Regional Differences and Geographical Restrictions on Trading

Based on the concentration of animals per unit of agricultural land, the Netherlands is divided into two regions. The manure-surplus region has an average manure production of more than 125 kg of P₂O₅/ha and encompasses parts of four provinces: Gelderland and Overijssel in the East and Noord-Brabant and Limburg in the South (see Figure 4.1). In the past, farms in the surplus region were typically small with a mixture of enterprises. Given the farm size, the lack of better alternatives forced farmers in the region to specialise predominantly in confined livestock production with off-farm purchases of virtually all feedstuffs. The manure deficit region covers the rest of the country (Groningen, Flevoland, Friesland, Drenthe, Utrecht, Noord-Holland, Zuid-Holland and Zeeland), for which the average manure production is below the 125 kg of P₂O₅/ha threshold. Until recently, this region had a very limited experience with confined livestock production. Historically, it was characterised by larger farms specialised in dairy, crop farming and horticulture.

Since the government's main objective was to keep the animal population from increasing in the surplus region, an important element of the system of manure production rights is a set of geographical restrictions on trading. The transfer of quota was allowed within regions and from a surplus region into a deficit region, but prohibited from a deficit region into a surplus region. In addition to phosphate quota, farmers willing to expand animal production in the surplus region had to acquire ammonia rights (MANMF, 1995). Trade in ammonia rights was only allowed within a county and hence was even more spatially restricted than trading in phosphate quota. There was no such ammonia regulation in the manure-deficit region.¹⁸ Despite the fact that the new regulations

18. The combined costs of phosphate and ammonia production rights increased the start-up costs of a new animal production unit in the surplus region by about 20% relative to the deficit region. The investment costs for a new swine finishing facility in 1995 required about NLFL 1,155 per pig place (Gibo, 1996). In addition, starting a new operation in the surplus region would require purchasing 7.4 kg of phosphate quota and 2.5 kg of ammonia quota per pig place. Using 1995 quota prices of 25 guilders per kg of phosphate quota and 25 guilders per kg of ammonia quota translates into an additional expense of about NLFL 250 per pig space. 1 NLFL $\approx$ 0.5 US\$.

significantly eroded the positive agglomeration effect that the hog industry in the surplus region had historically experienced, hog farmers did not leave the surplus region en masse.¹⁹

Figure 4.1. Livestock concentration areas in the Netherlands



Policy Uncertainty

The future of the quota system was uncertain from the beginning because of expected changes in policy. In 1993 the central agreement between the Ministries of Agriculture and the Environment and the farmers' union (Landbouwschap) was that by 1998 the quota system would become obsolete²⁰ with the introduction of a nutrient accounting scheme at the farm level. The short time period during which the quota system would be in effect raised the question whether this system had to be introduced at all (Haerkens and Walda, 1994).

The nutrient accounting scheme mentioned above included strict nutrient application standards not only for phosphate but also nitrogen per hectare plus a prohibitive tax on any surpluses. With such a scheme a farm's legal production capacity would not be determined by the amount of quota,

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- 19. Social and cultural factors (family ties, differences in religion and dialects) are known to limit the mobility of the farming population in the Netherlands considerably, despite the country's small size. Research on this topic has particularly focused on glasshouse horticulture, see for example Voskuilen and Van Elk (1990).
 - 20. Art. 22-3 of the "Law regulating transfer of manure production rights" (Amvb, 1987) states that the quota system would be terminated on January 1, 1997.

but by its capacity for manure disposal, either by land application on the farm itself or by hauling it to a crop farm in the deficit region. The scenario was met by massive protest from farmers, forcing union leaders to distance themselves publicly from the plan (Frouws, 1997). Besides, the plan was met with serious doubts on the part of the environmental organisations and drinking water suppliers.²¹ In addition each of the stakeholder groups had concerns about the objectives for ammonia reduction.²²

In 1995 the definitive governmental proposal regarding nutrient accounting was launched (MANMF, 1995). Nutrient accounting would become obligatory for both phosphate and for nitrogen. Nutrient surpluses above the waste standard would be subject to a high tax to ensure manure disposal to the deficit regions. In addition, additional requirements for new buildings were announced in order to reduce ammonia emissions. Also in 1995, the existing animal-based quota for pig and poultry farmers was cut by 30% in response to the development of low-nutrient feed.²³ Added to the political agenda by mid July 1997 was an additional 25 percent reduction in quota specifically for the swine sector.²⁴ Competing farmer action groups (including a radical union of pig farmers, Nederlandse Vakbond Varkenshouders or NVV) voiced their protest and attacked with renewed vigour, eager to gain time. Eventually, on January 1, 1998 the nutrient accounting scheme and building requirements were enacted but without abolishing the quota system. The 30% reduction of 1995 was revoked and the quota system was further detailed by separating the swine quota from those for chickens and broilers and by further detailing the swine quota. On September 1, 1998, quota for finishing and farrowing pigs were allocated on the basis of 90% of the actual number of these animals on each farm in 1996 or 1995. New housing requirements aimed at the improvement of animal welfare became imperative with the purchase of extra quota for pork production. A quota buy-out programme was initiated specifically for the swine sector.

The 10% reduction of September 1998 was part of the Pig Farming Restructuring Act (Wet Herstructureren Varkenshouderij), aimed at reducing the generated manure and the pig herd by 25% by the year 2000 (with 1995/1996 as base year). The Act also included stricter health and veterinary requirements for the pig production industry. Originally the Act passed the legislative hurdle. However, it became subject to litigation between the NVV and the Dutch government and was, therefore, halted until January 2000. On January 20, 2000, the Court in The Hague ruled in favour of the government and declared the 10% reduction in the size of the pig herd as enacted on September 1, 1998 to be legitimate. However, the court decision exempted the second generic reduction by 15% that was announced for 2000.

A survey in 1997 showed that Dutch livestock farmers in general perceive policy uncertainty as very relevant to their decision-makers and of the same importance as the uncertainty from production

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21. This was due to a debate whether the nutrient accounting/prohibitive tax system indeed would be capable of reducing the national manure surplus. Also additional research on the auditability of the scheme was requested as it was expected to be susceptible to fraud.
 22. The objective for 2000 was a reduction of 50% compared with 1980 and 70% compared with 1970. A recent evaluation showed that these targets were not met (CPB, 2000, p. 28).
 23. This cut was in response to the development of modified feed for pigs, which resulted in a reduction in the phosphate content of the manure of about 20-25 percent. There has not been a comparable innovation in poultry. However, only 80 to 90 per cent of the quota was actually being used on most farms in this sector, which meant that the cut did not have a big impact.
 24. The serious outbreak of swine fever in spring 1997 in the south of the Netherlands is seen as the incentive for this reduction in quota. Due to this outbreak pork production received even more media coverage. The large concentration of pigs was now associated with a whole complex of problem (animal health, animal welfare and surplus manure) and the public perception was that a reduction in animal numbers was the only solution.

and markets (Meuwissen et al., 2001). The respondents in this survey associated regulatory risks with environmental policy, animal welfare policy and the value of production rights, in particular.

In summary, in the context of the Dutch phosphate quota system, regulatory uncertainties arise for two main reasons: (1) the uncertainty of the continuance of the quota system, and (2) the uncertainty of the introduction of future constraints on quota use. The impact of constraints on quota use or of a quota phase-out is obvious: quota values would always be affected negatively. The impact of a cut in quota is less clear. The Dutch phosphate quota system does not allow leasing, so users of quota are all owners of quota. In that situation users of quota would all suffer a loss in asset values with a cut in quota. On the other hand, a generic cut would lead to an increase in the marginal value per unit quota, which would lead to higher quota prices. Hassett and Metcalf (1999) argue that the net directional impact of the overall effect of potential policy changes that follow a discrete jump process, such as those that might affect the quota programme, is uncertain. Barichello (1996) concludes that specifically for quota the perception is that the policy risks are highly asymmetric. Some options may exist for policy changes that would increase quota rents, but these are largely outweighed by possible changes that would reduce rents, consistent with the results of the above-mentioned survey study (Meuwissen et al., 2001).

IV. Evaluation

There is only a limited number of ex-post studies that evaluate the Dutch efforts to control the manure problem or parts of these efforts. In the overview presented here specific attention is given to reports written in Dutch in order to make them accessible to an international audience. The first sub-section highlights the institutional background, motivations for evaluation and data use of the ex-post studies and the next sub-section discusses the methods used in these studies. The results of the evaluation studies are presented in the next section.

Institutional Framework, Motivations for Evaluation and Data Use

The most specific evaluation study of the quota system is a 15-page report by the Dutch Ministry of Agriculture (MANMF, 1996) which covers the period January 1994-June 1996. It discusses the number of trades and trade volume, whether total farms where traded or parts thereof, price development, the impact of the quota system on structural change and on the development of natural resource and environmental policy, problems encountered with monitoring and control and other bottlenecks. Data used for this study are from *Bureau Heffingen* and from *IKC* [Information and Knowledge Center, renamed Expertise Center-LNV], both of which are agencies of the Dutch Ministry of Agriculture. *Bureau Heffingen* is responsible for the implementation of the regulations on manure and nutrients and records and monitors information on number of animals, quota and quota trade and manure application by farm.²⁵ The agency does not record quota prices. The price information reported in the study was collected by IKC by means of expert elicitation.

In 2000, the Netherlands Bureau for Economic Policy Analysis (CPB, 2000) published a substantial ex-post evaluation study of the total mix of manure regulations covering the period 1983-

25. The detailed records of Bureau Heffingen are not readily accessible but summary information is available (T.P.M. den Teuling, personal communication, Bureau Heffingen Assen, 2000). This summary information covers the period January 1994-Sept 1998 and includes the volume of trade and the total transferable (animal based) amount of quota in kg phosphate. The data is organised by 6 months periods and irregular shorter periods of mostly 2 months (e.g., Aug 17-1996; Nov 4-1996; Feb 1-1997). Trade is disaggregated in three senses: trade in the concentration areas, in the rest of the country and from the concentration area to the rest of the country. No distinction is made in the three classes of quota (cattle & turkeys; pigs, chickens & broilers; sheep, goats, foxes, nutria and ducks). The number of trades is not reported.

1999. This study was part of a wider project covering four prominent and persistent environmental problems in Dutch society and was commissioned to support the development of the Fourth National Environmental Policy Plan.²⁶ The study presents a detailed description of the policy changes since 1983 and of the development of the viewpoints of the various stakeholders. This evaluation however is for the total mix of regulations of which the quota system is only one element. The study uses commonly available sources and there seems not to have been any preferential access to data, or this was not used. An important source of data is the series of the Agricultural Economics Research Institute in the Hague (LEI-DLO) on the environmental performance of Dutch agriculture (LEI-DLO, various years). The series provides data on the development of animal numbers, waste volume and the cost of the regulation at the farm level. This information is gathered through annual farm surveys (CBS, various years) and the records of a panel of 1,500 farms maintained by LEI-DLO.

De Walle and Sevenster (1998) present an overview of agricultural policies on manure and nutrients in six EU member states, the U.S. and Canada. This study was commissioned by the Technical Soil Protection Committee, which advises the Netherlands Ministries of the Environment and Agriculture. The analysis of the Dutch legislation includes a very short discussion of the environmental effects for the period 1985-1992 (De Walle and Sevenster, 1998, p. 41-43), but no evaluation of any other criteria. The evaluation of the environmental effects is based solely on OSPAR data (OSPAR, 1995).²⁷

Finally, there are two academic studies that address single aspects of the phosphate quota programme. Vukina and Wossink (2000) analyse the regional differences in the impact of the policy regime specifically for land prices. Two hypotheses are tested at the level of provinces for the 1987-1996 period. First, the existence of the quota with regional restriction on trading should have caused a disproportional increase in price of agricultural land in the surplus region where the quota is binding relative to the deficit region where the quota is not binding. Second, the increase in cost of environmental compliance from other policy restrictions²⁸ that became considerably more stringent in the 1990s should have generated an eroding effect on the existing gap in land rents and consequently land prices between regions. Data used in this analysis all came from LEI-DLO and Statistics Netherlands.

The other academic study is Wossink (2000) and addresses the effect of policy uncertainty on the performance of the quota programme. It is hypothesised that policy uncertainty leads to a wait-and-see attitude that impedes the efficient functioning of the quota market. Uncertainty arises for two main reasons in the setting of the Dutch quota system: (1) the uncertainty of future constraints on quota use, and (2) the uncertainty of the continuance of the quota system. Stricter rules on waste disposal or the introduction of a pollution tax would lower returns to quota and thereby reduce the volume of trade. The phasing out of the programme would make the quota valueless and would lead to complete inaction on the quota market. The only data used in the analysis are quota prices, earnings

26. These four problems include: manure; CO₂ and the greenhouse effect; SO_x and NO_x from traffic and transportation, and aircraft nuisance of Amsterdam airport. The research of these four persistent problems was commissioned as background research for the preparation of the Fourth National Environmental Policy Plan (NMP-4) that sets the national environmental objectives for the period 2010-2030. The Plan was issued in 2001.

27. OSPAR is the combination of the Paris and Oslo Conventions, PARCOM and OSCOM respectively. PARCOM aims at the nutrient pollution of the (North) sea from sources on land and in the air. More or less parallel to PARCOM, OSCOM has focused on nutrient pollution from sources at sea. OSPAR has two objectives: monitoring and assessment and taking measures for environmental protection. Members of OSPAR are: Belgium, Denmark, Germany, France, Netherlands, Norway, Sweden, UK, Ireland, Spain, Portugal, Iceland, Finland and the EU (de Walle and Sevenster, 1998, p. 13-14).

28. The additional costs were mainly those of waste disposal because of the land application standards and costs of extra animal waste storage capacity.

from quota use and the interest rate. Data for quota prices²⁹ comprised a series of 166 and 103 individual transactions in quota for swine in the region south and east period in April 1996-May 1998, respectively. The Dutch Association of Realtors provided the latter data.³⁰

Methodology for Evaluation

The evaluation by the Dutch Ministry of Agriculture (MANMF, 1996) focuses on the main objectives of the law that introduced trade of phosphate quota: (a) to encourage structural development in animal agriculture by way of reallocation of non-land base phosphate quota, (b) to solve the bottlenecks encountered in the period of limited trading (1987-1993), and (c) to relocate farms away from nature reserves and forests. After the enactment of the “Guideline on Ammonia and Cattle Farming” a fourth objective was added: (d) to facilitate trade in ammonia quota. The MANMF report is organised according to eight evaluation questions: (1) the number of trades, (2) trade volume by region, (3) trades of whole farms as part of the total number of trades³¹, (4) price development, (5) whether quota trade enhanced structural adjustment, (6) prevention of increase in total manure production and prevention of further concentration of farms close to nature reserves, (7) monitoring and reporting, and (8) bottlenecks in the trade rules. It was not the intention of the study to perform an economic evaluation of the efficiency and effectiveness of the quota system in comparison with other policy instruments. The focus is on the dynamic effects of quota trade on the structure of the agricultural sector. The baseline used in the study is the situation before the quota became tradeable.

The evaluation by the Bureau for Economic Policy Analysis (CPB, 2000) addresses each of the seven key criteria listed by the OECD for the evaluation of environmental policy instruments (OECD, 1997). Environmental effectiveness and cost efficiency are evaluated as well as administrative costs, dynamic effects, wider economic effects and the role of attitude and awareness. The study includes an extensive discussion of the counterfactual, i.e. what would have happened in Dutch animal agriculture in 1983-1999 in the absence of the manure policy. A separate baseline is distinguished for the dairy sector, for pork production and for poultry. The remaining animal categories are not discussed. A comparison with ex-ante studies is also included.

De Walle and Sevenster (1998) only discuss the environmental effectiveness. These authors compare the average application rates of manure and fertiliser (in terms of N and P in kg per ha) in 1985 and 1993. They also discuss nitrogen surpluses in 1994 and 1995. The counterfactual is the situation without the policy efforts to reduce the manure problem.

Vukina and Wossink (2000) address the impact of the quota regime on land prices. The econometric analysis for 9 regions and 10 years enables detailed attention to be given to the counterfactual. The pooled cross-section time-series data include variables that represent the changes in land acreage, farm income, total manure output, cost of environmental compliance (manure storage and disposal) and the cost of capital. Remaining systematic differences in the change in land prices between the two concentration areas and the rest of the country are attributed to the quota system.

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29. The data on individual transactions used in the analysis covers 12.5% of the total volume of trade in the regions East and South over April 1996-May 1998 (T.P.M. den Teuling, personal communication, Bureau Heffingen Assen, 2000). For a revision of the paper (Wossink and Gardebroek, in preparation) panel data are used from LEI-DLO on quota prices and farm level earnings. The panel data covers 78 transactions in 1994-1997. The results of the revised paper are very similar to that in Wossink (2000). The panel data are not publicly available.
 30. For a revision of the paper (Wossink and Gardebroek, in preparation) panel data are used from LEI-DLO on quota prices and farm level earnings. These panel data are not publicly available.
 31. This distinction is important for the total quota volume after trade. In the case of the sale of a whole farm the farm's quota is not cut by 25% as with other quota trades.

Wossink (2000) argues that the value of quota as policy-created assets reflects market participants' assessment of the level, variability and duration of future returns to policy. The possibility of future change in policy regime leads to risks for the market participants. The stream of future incomes from quota use could be reduced or stopped by a change in policy. The paper asks to what extent policy risk affects quota prices and quota market efficiency. It is generally acknowledged that the price of production quota in agriculture (e.g., dairy, eggs, tobacco, and peanuts) is well below the expected capitalised value, indicating high risk-premiums for policy uncertainty.³² However, a simple comparison of quota prices and discounted returns does not include transaction costs which may also affect quota values and trade and does not address the impact of policy changes over time. Wossink (2000) uses the option value theory to assess the link between political uncertainty and quota price volatility. Based on the option value theory, an econometric model is developed for investing and disinvesting in quota. The econometric estimation enables the impact of policy risk on transaction costs to be distinguished from that of other, conventional, sources of transaction costs (e.g., cost for services by an estate agent and a notary). The evaluation baseline is the situation without policy uncertainty.

V. Overview of Main Results

This section follows the guidelines for ex-post evaluation of tradeable permit programmes as set out by the OECD Working Party on National Environmental Policy (OECD, 2001). The following criteria are discussed: economic efficiency, environmental effectiveness, administrative costs, dynamic effects and innovation, and "soft" effects.

Before we do so, we need to point out a distinct characteristic of the phosphate quota system that has general implications for the assessment of results and their interpretation. The Dutch permit system to control animal waste is in fact a quota on livestock numbers that is used as a proxy for the environmental impacts from animal waste. Such a system obviously only caps livestock and can only be environmentally effective and efficient if combined with other policy measures that directly aim at (the emissions from) waste handling and disposal and animal housing. In the Netherlands a series of such additional measures were introduced after 1987 and they became gradually more stringent with time. As a consequence, an evaluation of the quota system is complicated as it is difficult to disentangle the effects of the nutrient quota system from that of the other measures.

Economic Efficiency

Relevant indicators of the functioning of the permit market are price dispersion between trades, the volume of trade, the number of buyers and sellers, geographical patterns of trade and transaction costs (OECD, 2001). For each of these aspects the main findings of the evaluation studies are given.

Price of Quota

Considerable variation in quota prices was observed, both between regions and over time. The MANMF report discusses average quota prices for the two surplus regions and the rest of the country in 1994 and 1995. Prices for pig/poultry quota ranged from 25 to 60 guilders³³ per kg in the south, between 20 to 50 guilders per kg in the east and between 12 to 30 guilders in the rest of the country. Prices for cattle/turkey quota were 15 to 40 guilders per kg in both the south and east. Later on quota prices were higher. For the period April 1996-May 1998, prices for pig and poultry quota ranged

32. The rate of discount usually is 20 to 30% in real terms and reveals that buyers recognise that they are taking a sizeable risk and price the asset (quota) accordingly (Barichello, 1996). This assessment does not correct for transaction costs.

33. One Dutch guilder $\cong$ 0.5 ECU $\cong$ 0.5 US\$.

between 29 and 100 guilders in the south and 20 and 80 guilders per kg in the east, respectively (Wossink, 2000).

Volume of Trade

Quota became tradeable on January 1, 1994. By December of 1994, about 2,200 transactions had taken place with a total trade volume of 1.8 million kg phosphate (MANMF, 1996). This was however only 1.5% of the total volume of quota available for trade (T.P.M. den Teuling, personal communication, Bureau Heffingen Assen, 2000). Further data from Bureau Heffingen shows that by December 1997 in the concentration areas 8.1% of all non- land-based quota had been traded, and in the non-concentration areas 9.5%.

Several causes for the slow development of trade can be identified. Firstly, administrative approval of the trade was required -- farmers acquiring additional quota had to certify that they had a manure disposal plan in place for the next two years. In 1994, no less than 37% of the trade applications had to be resubmitted because of shortages in the manure disposal plan (MANMF, 1996). Secondly, farmers might have been allocated the optimal amount of quota and hence did not need to buy or sell to satisfy their production requirements. The initial allocation was based on information on animal numbers provided by the farmers themselves. The December 1986 survey organised for this purpose had been announced and likely led farmers to mention the maximum stable capacity instead of the average occupation. It is estimated that initial quota was over-allocated by 10 to 25% (CPB, 2000) and this likely affected trade volume. Thirdly, the restriction of quota transfer within region and by animal category limited the tradability and thereby reduced the market efficiency of the system. The even more limiting spatial restrictions (within counties) on the trade of ammonia quota further reduced the efficiency of the phosphate quota system.

Geographic Pattern of Trade

Vukina and Wossink (2000) conclude that in the earlier stages of implementation, the programme served as a barrier to entry into the swine industry in the manure surplus region. Later, an upward shift in environmental costs acted as a stimulus for producers in the surplus region to move their production to the deficit region. Due to the higher quota prices in the surplus regions, it was more profitable to leave (sell) the quota in the surplus regions and to buy land in the rest of the country. The manure production rights needed to expand production in the rest of the country were almost entirely land-based, as is confirmed by data from *Bureau Heffingen* (T.P.M. den Teuling, personal communication, Bureau Heffingen Assen, 2000). The rents formerly created by the phosphate quota programme gradually dissipated as new environmental policy constraints on manure storage and land application took effect. Searching for less constraining environmental policies, a number of pork producers left the surplus regions and relocated to the rest of the country (or migrated).

Transaction Costs

Wossink (2000) estimates the conventional transaction costs incurred by the markets participants for swine and finds significant estimates as high as 17% of the average quota price. Many other studies indicate the prevalence of significant transaction costs in tradeable permit markets (Stavins, 1995). Recall that in the specific case addressed here, formal approval of the trade was required for which buyers had to certify that they had a manure disposal plan in place. The cost of arranging such a plan and the administrative cost associated with obtaining approval can be assumed to be important particularly because as these costs are invariant to the size of the transaction. Besides, the trading system as such was very complicated, the brochure explaining the system to farmers contains no less than 66 pages (MANF, 1993). This complexity likely led to pecuniary and also non-pecuniary transactions costs for farmers.

Wossink (2000) also estimates the impact of uncertainty in terms of transaction costs and market efficiency. Uncertainty drastically increased the transaction costs and market equilibrium was virtually absent. The probability of a wait-and-see attitude due to uncertainty was found to be considerably higher for the eastern region where the additional transaction cost was also much higher. The latter increase was estimated to be 71.31 guilder per kg of quota in the southern surplus region and 152.57 guilders in the eastern surplus region, respectively. In comparison, the average quota prices paid were 66.11 guilder and 48.69 guilder per kg in the South and East, respectively. The announcement in July 1997 of a future cut in quota led to significant further increases in transaction costs and less efficient markets. The policy announcement particularly affected the quota market in the south. The differences between the estimated option values in the South and East regions, and the estimated increase in option values after July 1997 correspond well with anecdotal evidence for the quota market.

Environmental Effectiveness

As described above, the Dutch permit system to control animal waste is a quota on livestock numbers which is used as a proxy (by fixed emission coefficients by animal type and age category) for damages arising from phosphate and indirectly also nitrate.³⁴ Optimal control would require continuous monitoring of nonpoint-source emissions but is prohibited by the intermittent nature of the discharges and the fact that pollutants are widespread. In such a situation a useful alternative for regulatory bodies is to grant political legitimacy to environmental indicators such as a fixed phosphorus coefficient per animal. Such an indirect approach of input-output oriented environmental indicators is acceptable for policy design whenever there are no indications that the suspected polluters possess better information about the ultimate pollutant release than the regulatory agency (Dosi and Moretto, 1993). This reasoning was definitely applicable at the time the phosphate quota system was first introduced.³⁵ Furthermore it is argued that generally, the indirect approach of environmental indicators should be preferred whenever it is believed that the cost for regulatory agencies of acquiring information about ultimate environmental implications of productive decisions in terms of ambient pollutants levels are prohibitively high (Dosi and Moretto, 1993). This point of view is particularly applicable to agriculture, which involves many producers, many pollutants and many production situations.

The Dutch phosphate quota system imposed a ceiling on the maximum phosphate output and a cut by 25% in the case of trade. This 25% retirement rule was imposed for environmental reasons. Phosphate quota is differentiated by animal category and the phosphate standard per animal category directly links quota volume to animal numbers. With these trade rules, animal numbers and total phosphate output will at least be stabilised and could be reduced by as much as 25%. Notice however that the retirement rule is a disincentive to trade. It reduces gains from trade by more than 25%, since buyers will discount the resale value as well. Secondly, it makes the cap completely uncertain since it is dependent on trade volumes. None of the evaluation studies analysed pays attention to this issue.

34. It was assumed that a decrease in nitrogen problems would follow naturally from imposing limits on phosphate application and a fixed ratio was assumed for N: P of 2:1 (Frederiksen, 1997).

35. After the introduction of the detailed nutrient bookkeeping system in 1992 farmers quickly gained an understanding of the environmental impacts of animal production and how to reduce emissions through changes in management. Nutrient bookkeeping was developed by CLM (Centre for Agriculture and Environment) and different groups of farmers and was originally designed to provide management support. The environmental data provided by this management information system were particularly insightful in combination with farm economic data, i.e. to provide empirical evidence of significant variations in environmental inefficiencies. From 1998 onwards the management information system was used as a policy instrument to tax farm level surpluses of both N and P (see Breembroek et al., 1996 for details).

MANMF (1996) reports that in 1994, 1995 and the first six months of 1996 a total of 6.7 million kg phosphate was traded of which 1.6 million was cut. The trade covered 6.6% of the total quota volume in the concentration areas. Outside the concentration areas this was 7.4 %.³⁶ Trade in quota for pig and chicken and broilers made up 85% of the trade volume and trade in cattle and turkey quota was responsible for 11%. The report highlights that additionally there had been 2,200 trades of in total 6.6 million kg animal-based quota that were not cut because the quota trade was part of the sale of a complete farm. The MANMF report discusses that the change in the volume of animal-based quota is not directly indicative for the change in environmental pressure from animal production. The start of new farms on land-based quota in the non-concentration areas and additional production based on previous "dormant" quota could actually have increased total manure production. The report refers to statistics based on a sample of the manure records of farms in animal production that showed that the actual production of phosphate in Dutch agriculture decreased from 163 million in 1993 to 134 million in 1995. The MANMF reports implicitly attributes this reduction of 17.8% of the total phosphate production in 2.5 years to the quota programme.³⁷

The Bureau for Economic Policy Analysis (CPB, 2000) emphasises that it is incorrect to ascribe to the policy the observed reduction in quota volume and in actual manure production. In cattle and dairy farming there were only minimal impacts from the mix of manure regulations and the quota programme. The reduction in animal numbers and in actual waste production in the cattle and dairy sector (Table 4.3) can be completely ascribed to the EU's Common Agricultural Policy. The introduction of milk quota in the early 1980's and the reduction in price support for beef were the main determinants of the changes in this sector. In contrast, the regulation on manure did have an impact in pork production. The quota system prohibited a further increase in animal numbers and the compliance cost of the regulations on manure application and storage increased the cost of production considerably. Without the policy on manure the number of pigs would have been 10 to 20% higher based on the counterfactual for the swine sector in Folmer et al. (1995). For the poultry sector the same conclusion is reached, analogous to the analysis for the swine sector. The CPB study concludes that without the policy, total manure production from livestock production in the Netherlands would have been 5 to 10% higher.³⁸ The specific contribution of the quota programme is not addressed.

36. Data from *Bureau Heffingen* (see note 24) shows that by December 1997 in the concentration areas 8.1% and in the non-concentration areas 9.5% of all animal-based quota had been traded. The total tradeable quota volume was 97.5 million kg phosphate by the end of 1994 and 96.1 million by the end of 1997.

37. The only caveat mentioned in the report is that 95% of those farms that sold their entire quota amount quit animal production completely and that these farms likely would have stopped anyway, even without the quota programme. This insight was based on survey information from IKC and was not statistically confirmed in terms of quota volume and actual phosphate production.

38. The CPB study emphasises as additional positive effects the improved distribution of the manure over total agricultural land and the improvements in (swine) animal nutrition that reduced the phosphate content of manure. These two effects are entirely attributable to the regulations on manure application and cannot be ascribed to the quota system.

Table 4.3. Trends in animal numbers and total animal waste production, The Netherlands, 1980

	<i>Animal Numbers</i>			<i>Actual Animal Waste Production</i>							
	Pigs	Cattle 1000 Head	Poultry	Pigs Manure MMT	P ₂ O ₅ Mil.kg	Cattle Manure MMT	P ₂ O ₅ Mil.kg	Poultry Manure MMT	P ₂ O ₅ Mil.kg	All livestock Manure MMT	P ₂ O ₅ Mil.kg
1980	10,138	5,226	82,035	14.6	..	68.5	..	1.7	..	85.7	230
1990	13,915	4,926	92,765	16.4	69.6	65.4	133.3	2.3	32.7	86.0	226
1993	14,964	4,797	98,086	17.0	72.0	65.5	126.1	2.5	34.8	86.0	235
1994	14,565	4,716	93,953	16.4	65.3	63.6	119.6	2.3	33.0	81.2	219
1995	14,565	4,716	93,953	16.2	60.1	63.7	118.4	2.1	30.7	80.9	209
1996	14,419	4,551	93,552	16.9	56.9	61.7	103.6	2.4	30.5	80.4	207
1997*	15,189	4,411	95,295	15.5	57.2	59.7	101.0	2.4	32.1	77.3	186

* Affected by outbreak of classical swine fever. Source: CBS/LEI-DLO, various years, and www.mininv.nl/infomart/feiten/1997/feitlandb6.htm

Administrative Costs

The only study to discuss the cost of enforcing activities is the report by the Bureau of Policy Analysis (CPB, 2000). The total annual costs are estimated to be about 44 million Euro (110 million guilders).³⁹ This figure covers the administrative cost associated with manure policy. The enforcement and control cost of the manure bookkeeping system can be assumed to be particularly considerable. On the other hand, this figure does not include the research expenditures of governmental and semi-governmental and private organisations or the cost of the additional staff requirement at the Ministry of Agriculture and its agencies.

Dynamic Effects and Innovation

The 1996 MANMF report mentions that 95% of those farms that sold their entire quota amount quit animal production completely. The report concludes that the quota system enhanced structural changes, i.e. it encouraged exit. This insight was based on survey information and was not statistically confirmed in terms of quota volume and actual phosphate production.

There is significant evidence that the policy mix on manure provided a continuing incentive for animal producers to search for innovative approaches to reduce emissions (CPB, 2000; Wossink and Wefering, 2002). However, these reductions have to be ascribed to the measures on manure storage and application and cannot be attributed to the quota system. The emission reductions were made possible by nutritional measures particularly in the swine industry, which was most seriously affected by the regulations on manure. Aside from improvements in nutrition efficiency using existing diets, significant improvements were achieved by the development of modified feeding regimes for pigs. The modified diets reduced the N and P intake and thereby the N and P surplus while maintaining or even enhancing daily weight gains. With storage of additional feed types accounted for, total costs per farm were assessed to be negative for finishing pig farms and very limited for farrowing pig farms (Leneman et al., 1993).

Soft Effects

A psychologically important point was that many farmers were not convinced of the environmental benefits of the policy (Katteler and van den Tillaart, 1989). To some extent these benefits are also still disputed in academic circles (CPB, 2000). The fierce response of the swine

39. This figure is based on the estimate given in MANMF (1995).

producers, in particular, has to be viewed in this context. Besides the swine sector was severely affected by the regulations on manure storage and land application that were later added to the policy. The compliance cost of these regulations had to be borne mainly by the swine producers and these predictably did not embrace the policy.

The CPB report emphasises that there were far too high expectations in policy circles of the possibilities of technical development and the export of processed manure (CPB, 2000). Consequently the position of the swine industry as an 'exposed sector' was not recognised in time and this became a big bottleneck. The Dutch swine sector dominated the policy debate but in fact is relatively small in macro-economic terms.⁴⁰ A quota buy-out in the early nineties for swine production could have prevented a lot of environmental damage and human grief (CPB, 2000).

VI. Comparison with Ex-Ante Studies

There are several ex-ante studies that address the expected economic impact of the manure regulations. None however looks specifically at the phosphate system. Tamminga and Wijnands (1991) present an ex-ante analysis of the costs to achieve a balance between production and utilisation of the nutrients in manure by the year 2000. Stolwijk (1989) assesses the direct economic effects of three policy scenarios for animal agriculture by 1986. Both studies address the cost of changes in environmental management at the individual farm and indicate that for the majority of the swine producers, compliance cost would amount to 40-50% of their average income. The poultry sector would also be severely affected.

VII. Conclusion: Implications for Policy Reform

The Dutch experience with the phosphate quota programme provides considerable experience and data to draw on. The implications for policy reform can be summarised as follows.

Baseline Data and Permit Allocation

There must be accurate data on the baseline for permit allocation together with a reliable and accurate system of monitoring and accounting. Theoretical studies commonly state that "regulators influence the success of tradeable permits systems by permit allocation mechanisms." The evaluation of the Dutch phosphate quota programme shows that initial quota was over-allocated 10 to 25%. Allocation was based on a farm survey in which many farmers mentioned the maximum stable capacity instead of the average occupation. Consequently there was less incentive for Dutch livestock farmers to buy quota.

Consistency

Ambiguous property rights complicate any environmental regulation, but can cripple a regulation involving around trading of the rights. In the context of permit systems regulatory uncertainties arise for two main reasons: (1) the uncertainty of the continuance of the system, and (2) the uncertainty of the introduction of future constraints on permit use. There must be confidence in the stability of the policy system for a reasonable period of time. Without this, participants will not trade. Consequently, policy uncertainty incurs aggregate welfare losses because of misallocation of resources to produce the permitted output. The process by which a decision on alternative policies is taken can generate uncertainty. Uncertainty can also be created through the prospects of policy changes and even by uncertainty about the implementation of programmes to enhance trade. Because these factors are likely to be present to one extent or another in all future trading programmes, it is

40. In 1997 there were 6,304 pig farmers (full-time).

important to understand their implications on market performance in order to judge and evaluate such programmes accordingly.

Permit Unit

Specifically for the control of non-point source pollution it is common to use a proxy for the environmental impacts. The Dutch permit system uses livestock numbers as an indicator of the environmental impacts from animal waste. For a permit system based on such a proxy to be environmentally effective it has to be combined with other policy measures that directly aim at the emissions. However, such measures may undermine the effectiveness of the permit system (Tietenberg, 1980). Moreover, the discussion about additional policy measures leads to policy uncertainty.

Autonomous Developments

Environmental pollution is not influenced by governmental policy only. Autonomous developments, such as technological, structural and market developments also affect pollution levels and their impact can be positive or negative. Policy makers in general appear to be far too optimistic with respect to the effects of both regulations and autonomous developments. It should be prerequisite to use ex-ante studies in policy design to prevent policies being developed based on pertinently unrealistic assumptions.

Timing

The Dutch example shows that a tradeable permit system as the single policy instrument to achieve policy targets might only work when actual pollution and policy targets are reasonable close. Therefore it is important to act in a timely fashion. If the quota system had been introduced in the late 1970s the manure problem would probably have been solved long before.

Commitment

A strong policy commitment to the system and its objectives is crucial. The constituency of political support should include the affected parties and the implementing agency. Specifically in agriculture where there are many relatively small enterprises, support should include agencies providing administrative assistance. Otherwise conventional transaction costs can be very high.

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Chapter 5

EVALUATING THE NEW ZEALAND INDIVIDUAL TRANSFERABLE QUOTA MARKET FOR FISHERIES MANAGEMENT

by

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I. Introduction

Inshore fisheries depletion, the development of the quota-based programme for offshore fisheries, and the general orientation of the New Zealand government in the 1980s toward deregulation, combined to create an atmosphere conducive to fundamental change in New Zealand fisheries management. After several years of consultation with industry, the Fisheries Amendment Act of 1986 was passed, creating New Zealand's individual transferable quota (ITQ) system. Modifying legislation has been passed several times since, but the basic structure of the system has remained intact.

The system has been evaluated many times since its inception. Most evaluations have been qualitative in nature with the emphasis on identifying problems and improving the design of the regulations. The state of the fish stocks is reviewed regularly, though significant uncertainty remains. This report presents results from the first systematic assessment of the economic efficiency of the system. We assess only the likely cost efficiency of the ITQ market. We do not assess the environmental effects of the system and hence cannot comment on the overall economic efficiency of the regulation. We begin by giving a brief description of the ITQ system. We then discuss our motivation and the institutional framework for the evaluation. We discuss our methodology and basic results and then compare these with other evaluations of ITQ fisheries programmes and some other economic research on the New Zealand system.

II. Overview of the Tradeable Permit Regime

The environmental objective of the NZ ITQ system is primarily to ensure that catches are limited to levels that can be sustained in the long term. The total allowable catches are intended to be

1. The views expressed in this chapter are those of the author and do not necessarily reflect those of the institute with which the author is affiliated or the OECD.

set to achieve Maximum Sustainable Yield (MSY). The ITQ system is part of a broader Quota Management System (QMS) that includes direct controls on fishing areas, techniques and seasons.

The basic motivation for the new regulations was to manage the fisheries as an economic resource in a sustainable and efficient way. The regulations were seen as a service to the fishing industry and there was a strong push to ensure that the costs and risks of management were borne by the fishers who gain the benefits from the system. Early on, the government planned to extract the rent from the industry on behalf of taxpayers in general. Resource rents continue to be collected, but are not a major revenue source. In contrast to many international fisheries, regulators were not concerned about protecting small fishers or fishing communities. This had to be revisited with regard to Maori fishing rights, but had important impacts on the design of the system.

Quotas are defined in terms of species and location. The ITQ system covered 33 species by 1998. The New Zealand EEZ is geographically delineated into quota management regions for each species based on the location of major fish populations. In 1998, the total number of fishing quota markets stood at 157, with the number of markets per species ranging from 1 market for hoki (*Macruronus novaezelandiae*) to 10 for paua (abalone). As of 1996, the species managed under the ITQ system accounted for more than 85% of the total commercial catch taken from New Zealand's EEZ.

Each year, the Minister of Fisheries sets an annual total allowable catch (TAC) for each fish stock based on a biological assessment as well as other relevant environmental, social, and economic factors. The process for setting these TACs involves scientists, the Ministry, industry, and environmental groups. The TACs are intended to be set with a goal of moving the fish population toward a level that will support the largest possible annual catch (i.e., maximum sustainable yield), after an allowance for recreational and other non-commercial fishing.

Individual quotas were initially allocated to fishermen as fixed annual tonnages in perpetuity based on their average catch level over two of the previous three years.² By denominating quotas as fixed tonnages, the government was counting on its ability to purchase quotas on the open market if it wanted to reduce the total catch from a fishery. After several years of high costs, and faced with the prospect of spending NZ\$100 million to reduce TACs (Sissenwine and Mace 1992) for Orange Roughy alone, after prolonged negotiations the government switched from quota rights based on fixed tonnages to denominating the quotas as a share of the TAC, beginning with the 1990 fishing year. In doing so, the burden of risk associated with uncertainty over future TAC levels was transferred from the government to the industry. Most TACs have not been altered since this change so for most fishers the effect has been potential, not actual.

Fishing quotas are generally tradeable only within the same fish stock, and not across regions or species or years, although there are some minor exceptions.³ They are traded one-for-one. The quota rights can be broken up and sold in smaller quantities and any amount may be leased and subleased. As of October 1, 2001, annual quota leases were supplanted by sales of "Annual Catch Entitlements" or ACEs, which are issued annually by the government equal to each quota owner's

2. Since 1996 if a new ITQ fishery can support a TACC greater than the fishers histories returned (plus 20% for Maori), the government can auction the excess quota by competitive or closed tender.

3. Given the uncertainty around quantity and composition of catch, additional flexibility was introduced into the system in four ways (Clement & Associates 1997). First, a by-catch trade-off exemption allows fishermen who incidentally take non-target fish to offset the catch by using quota from a predetermined list of target species. Second, quota owners can carry forward to, or borrow from, the next year up to 10% of their quota; this right does not apply to leases. A third option is to enter into a non-monetary agreement to fish against another's quota. Finally, a fisherman can surrender the catch to the government or pay a "deemed value," which is set based on the nominal port price to discourage discarding of catch at sea and targeting stocks without sufficient quota (Annala 1996).

annual quota allocation. Thus there are now two quota instruments: the ACE and the right to the perpetual stream of ACEs. There is no centralised trading platform. The government initially created one, but bilateral trading and brokers quickly replaced it.⁴ High levels of trade (see later) suggest that transaction costs are relatively low and the introduction of ACE aims to reduce the legal costs of trading even further.

There are legislative limits on aggregation for particular stocks and regions, and limitations on foreign quota holdings.⁵ No pre-trade approval is required. However, all trades must be reported to the Ministry before the buyer can use the quota. Harvested fish must be matched against quota when they are sold to a licensed fish receiver. Compliance and enforcement is undertaken through a detailed set of reporting procedures that track the flow of fish from a vessel to a licensed fish receiver (on land) to export records, along with an at-sea surveillance programme including military aircraft, on-board observers (Boyd and Dewees 1992).⁶ Misreporting is a criminal offense.

The New Zealand system involves a large number of participants, averaging over 1,500 over the history of the programme with more than 1,400 in 1998. The scope of the system ensured that the markets had the potential for liquidity. Although many small players were excluded when the system was initially created, a healthy number remained in every important market. Individual markets have had a median of 45 quota owners.

As of 1998, the number of participants in the offshore fishery had been barely affected by the ITQ system in part because it already had a quota system before 1986 and partly because these are large players. The number of participants in the inshore fishery has gradually declined. This could indicate consolidation of quota ownership. The number of owners of shellfish stocks rose in 1990 primarily because of the addition of new stocks to the system. The Treaty of Waitangi Fisheries Commission is currently the sole representative of Maori collective quota owners (they held more than one third of all quotas as of 2002). They manage the quotas on behalf of all Maori. When these quotas are allocated to individual iwi (tribes) in the near future, the number of participants will rise further.

The New Zealand ITQ system is a dynamic institution that has had many refinements since its inception more than 15 years ago. Nonetheless, the basic tenets of the system — setting a total allowable catch and leaving the market to determine the most profitable allocation of fishing effort — have remained intact.

III. Context of Evaluation

Motivations

Our motivations for the study discussed here and our ongoing research are three-fold. First, we are assessing the New Zealand system to identify areas of success and/or possible improvement or expansion within the New Zealand system. Such evidence can help protect the positive features of the system against pressure for change, and offer evidence in support of beneficial changes.

4. The electronic system was closed in 1988 due to insufficient volume (Dewees 1996).

5. Initially, the aggregation limits were on *holding* quota. Substantial changes were written into the 1996 Fisheries Act, one of which was changing the limits on holdings to *ownership* levels. The 1996 legislation also relaxed the aggregation limits for particular species and region combinations.

6. In a survey of fishermen operating under the New Zealand ITQ system in 1987, Dewees (1998) found that 40% thought enforcement and 66% thought highgrading were potential problems with an ITQ management system. Highgrading is the practice of maximising the quality of the catch to be counted against one's quota by dumping less valuable fish over board. These numbers dropped to 21% and 25% respectively in 1995 (Dewees 1998).

Second, and maybe more importantly we are seeking to provide evidence that is useful for other countries considering using individual tradeable quota systems. We were particularly motivated by the intense debate in the United States where some systems have been created (and even evaluated; Grafton et al 2000), but where there is also fierce opposition to the more widespread implementation of ITQ systems.

Finally we are motivated by the ability to provide evidence understanding tradeable permit markets more broadly. The New Zealand ITQ system has some features that are relevant to other potential tradeable permit markets, including in the area of pollution and not just the resources used.

ITQ Markets and Other Tradeable Permit Markets

The spatial specificity of fisheries markets is analogous to non-uniformly-distributed pollutants (e.g., ground level ozone). The existence of some very thin markets and high degrees of vertical integration are relevant for situations where there are concerns about market power. The joint harvesting process, where several species are caught simultaneously, is similar to the multi-pollutant problem where one source produces several pollutants simultaneously. In both cases, the need to balance a portfolio of quota across species (pollutants) and locations creates complexity that potentially raises transaction costs. These interrelationships also create difficulties, and additional opportunities, for evaluation, which we have not yet addressed. Quota prices of interrelated species should also be interrelated. Both production inter-relationships and ecological relationships (e.g., predator-prey) also make evaluation of the environmental implications of the programme more complex.

The ITQ system is only a partial solution to the fisheries management problem. Localised 'hot spots' with local over-exploitation can occur for species like shellfish or lobster, as the timing of harvesting is critical with respect to breeding cycles. This is similar to problems with situations where a pollutant can be highly toxic if it is emitted in large concentrations or in proximity to vulnerable populations (e.g., near a school). This is partially addressed through separate regulations on fishing seasons, closed areas and marine reserves that directly avoid 'hot spots'. Using separate controls, rather than trying to achieve this aim through limitations on trade, keeps the ITQ system kept simple and 'clean'. Some also argue that greater concentration of quota ownership would provide a greater incentive for fishers to internalise local environmental effects because each owner would feel they had a direct stake in the future value of the resource.

In response to fiscal risk to the government, the NZ ITQ system defines quotas as shares of a shifting cap. This approach is relevant to quota systems where the government bears some fiscal risk (e.g., where some allowances are auctioned or where the government needs to buy back allowances to cut the cap). The different risk-sharing characteristics of this approach might be particularly relevant for design of a greenhouse gas market where the scale of the market might have significant fiscal and even macroeconomic effects.⁷ The NZ fisheries case provides an extremely rich source of information from which to explore these issues. We have only begun to tap it. Assessments of specific tradeable permit systems can be more valuable if they can be designed to yield results of general interest beyond the immediate system and even beyond the specific issue. This is especially true given the limited opportunities for empirical evaluation.

7. For more discussion of this issue see Kerr (2003).

Institutional Framework

Our analysis so far has been done in two stages. Resources for the Future and Motu funded the first stage out of internal funds. During this phase we negotiated a confidentiality agreement with the Ministry of Fisheries to provide us with access to some of their confidential data. We paid relatively low costs for them to extract the data we needed. The Ministry provided us with considerable indirect support through access to staff. It was relatively easy for us to get information even as independent researchers partly because of the constructive attitude of the Ministry, partly because of the excellent reputation of Resources for the Future and the individual researchers, and partly because New Zealand is very small.

The fact that the Ministry of Fisheries did not initially fund us made it easier for us to gain the confidence of both industry actors and the environmental community. Again, the RFF reputation for objectivity and independence was helpful. It also helped that the issues we chose to study were the least contentious, at least in New Zealand. They were not directly associated with on-going policy reform. Finally, the policy is widely perceived to be a success and most of the players involved are quite proud of it. This made them very supportive of research that could rigorously show the value of the programme.

We are now partly funded by the Ministry of Fisheries. This is a process with relatively loose oversight. We have the right to publish our findings regardless of the outcome. We maintain our independence. Our closer relationship has slightly facilitated data access and we no longer pay even nominal sums for data. Our research methods and findings are open to public and academic review. All our research is available through our website as it becomes working papers. We have presented our research at a large number of conferences and seminars throughout the United States and in New Zealand. We are committed to making some of our data available for other researchers after we have published our results. This will still be limited by confidentiality requirements so that most data requests will need to go through the Ministry of Fisheries.

IV. Description of Assessment Methodology

Our evaluation framework is based in economic theory. A large part of our work to date has involved documenting the history of the programme (Straker et al 2002) and in developing and cleaning the databases that can be used for empirical evaluation. We are now beginning to analyse that data. Newell et al (2002) combines descriptive statistics motivated by economic hypotheses, with econometric work on the determinants of quota prices. It also includes a data appendix that describes our current data in some detail.

Our empirical work is focused on assessing the economic efficiency of the quota market. We do not consider the overall economic efficiency of the programme, including environmental gains. However, we assess economic efficiency indirectly by testing whether we observe patterns we would expect to see in an efficient market. This avoids the need for a baseline but clearly has the limitation that we cannot give absolute numbers for the economic gains relative to the baseline.

The design of the New Zealand system facilitated our evaluation in some ways. The first key was the relative simplicity of the system. The rules are simple, consistent across species and areas, and have changed little over the history of the programme. This makes it relatively easy to model optimal fisher behavior when faced with these rules and reduces idiosyncratic features of the data. The stability of the rules allows us to look for time-series effects without worrying significantly about regime shifts.

The administrative requirements of the system create a comprehensive database of ownership and trades. Although these data had not been used previously for research (except at an

aggregate level) they were available in consistent computerized format. The reporting system also requires the price of each trade. Many people have claimed that prices that are voluntarily reported in tradeable permit markets are unreliable. To a certain extent this was true; there were many trades with a price of either \$1 or very high prices (e.g. including a boat in the trade). We were able to clean up the data however and produce useful, robust results.

Using information obtained from New Zealand government agencies and other sources, we assembled a comprehensive panel database of information on the New Zealand individual transferable quota (ITQ) system over the period 1986–1999. The data include information on the name of each fish stock, quota transactions (i.e., prices and quantities of quota leases and sales), the export prices of fish species covered by the ITQ system, quota ownership, the total allowable commercial catch (TAC) and actual catch for each fish stock, biological information on fish species, climatic variation, and interest rates.

V. Main results

Do the Market Conditions Support Efficiency?

All results discussed in this and the next section are presented in much more detail in Newell et al (2002). As discussed earlier, the number of participants in most markets suggests that the markets are potentially very liquid. Transaction costs appear to be low and the markets are very active. The quota sale markets were most active in years when quotas were allocated with a high of 3,250 trades in 1986. This likely reflects rationalisation of quota portfolios. The 1990 sales peak partly reflects adjustment to new allocations of shellfish quota, but also reflects allocation of quota to Maori and possibly some response to the redefinition of quota as shares. In recent years around 4% of quota are sold each year in the median market. The lease market has developed considerably from only 2,000 trades in 1986 up to 14,500 by 1998. In the median market, the percentage of quota leased has risen consistently to about 40% annually. Some markets are relatively inactive, but they tend to be economically unimportant. This high level of participation and activity suggests that the markets might be operating well.

The degree of price dispersion also suggests that these markets operate well. We measure price dispersion as the mean absolute percentage price difference between individual trade prices and the monthly mean price for that stock. We find that price dispersion is lower for sale prices than leases, which one would expect given that leases have quality differences for which we cannot control, such as agreements to provide the caught fish to the lease owners. We also find that dispersion falls over time, which is consistent with a period of market learning and development. The dispersion appears to be in a range compatible with many other products and tradeable permit systems that are typically thought to have well-functioning markets, although direct comparisons are difficult. Further work is being undertaken to explore this issue.

Do Quota Prices Reflect Fundamentals?

A one-year quota lease should be worth the expected annual profit from one tonne of fish, or the difference between the price of fish and the marginal cost of catching fish. Simply mapping either lease or sale prices against fish export prices we find a strong relationship between the two. This however does not control for the cost of catching fish, or for whether the TAC is actually binding (if it is never binding the quota price might be expected to be close to zero, as there is no scarcity).

We use a reduced form approach to relate export prices, and factors that affect costs, to lease prices.⁸ We find that we can explain around 95% of the variation in lease prices over time. Export prices

8. We use Feasible Generalised Least Squares, with and without fixed effects for each fish stock, to econometrically estimate the relationships across all time periods and species.

have a strong positive relationship with quota prices. Higher fishing costs are associated with lower quota lease prices. Higher quota demand (more binding TACs) pushes quota prices up. Greater ecological uncertainty, measured in terms of weather variability and natural mortality rate of the species, which implies more uncertainty catch and hence cost uncertainty, lowers the value of quota. Prices also rise when the economy is doing well in general.

Finally, after controlling for all other factors we find an upward trend in quota prices in general. One of the justifications for a quota system is that it will improve the efficiency of the fishing industry and hence its profitability. This should lead to higher quota prices. This effect will be most marked where stocks were being overfished but are now more sustainable because of the regulation. In the New Zealand system some stocks were initially overfished while others were at sustainable levels. All of these might have been overfished if the system had not been introduced but we expect the initially over fished stocks to benefit most. We observe a significantly higher quota price trend for stocks that had their catch levels cut when they were introduced to the ITQ system. This suggests growing fishing profitability as a result of rebuilding stocks.

The price to buy a quota in perpetuity should be the discounted value of future profits, i.e. the sale price should be the lease price divided by the discount or interest rate. If fishers expect lease prices to rise or fall, the current sale price would also incorporate these expectations. The sale price should then be roughly equal to the lease price divided by the discount rate minus the lease price growth rate.

Empirically, the sale prices respond to fundamentals in a similar way to lease prices. One notable difference is that the effects of transitory factors, fishing cost changes and short- term ecological uncertainty (from weather variability) are lower, as we would expect. We find that the implicit discount rate inferred by the relationship between sale and lease prices is broadly consistent with the level and trend in real interest rates in New Zealand in this period. Our results do not suggest that fishers, or at least owners of quota, have unusually high discount rates. We might also expect that sale prices would incorporate uncertainty about the expected length of the regulatory programme, and about the security of the property rights. After issues with Maori quota, and the issue of moving from tonnage to percentages of TACC were resolved, the value of quota might be expected to rise. These effects cannot be easily distinguished in the time series.

Overall the evidence suggests a reasonable level of economic sophistication, implying that market-based quota systems are potentially cost-effective instruments for fisheries management. Anecdotal evidence (and some studies, see below) suggests that the industry structure, technology used, behaviour and product quality have changed substantially since the system was initiated. Many of these changes might have occurred in the absence of the ITQ programme, particularly with the more general economic reforms in New Zealand. Because the ITQ system was introduced before overfishing was a problem for many stocks, the distortions created by command-and-control regulation in many countries are not as much of a feature of the New Zealand industry. These distortions may well have arisen if the fishery had been regulated by command-and-control during its rapid development phase but assessing this would require a complex counterfactual.

VI. Comparison of Results

Comparison with Other Assessments of NZ ITQs

Many evaluations of the NZ ITQ system have been qualitative in nature, supported by a few descriptive statistics. Pearse (1991) offers the first key report with his independent review of the state of New Zealand's fisheries management five years after the introduction of the QMS. However he and most other early reviews concentrate on direct analysis of the regulations and anecdotal opinion. Straker et al (2001) gives a more complete list of studies relating to the New Zealand system. Here we focus on studies relating to economic efficiency.

Several studies have addressed the efficiency effects of the move to the QMS with varying degrees of empirical analysis. Dewees (1989), who carried out extensive industry surveys, finds that fishers were changing behavior in response to the ITQ system. In particular they were changing on-board methods for handling individual fish that could then be sold to lucrative markets in Japan. Sissenwine and Mace (1992) also give some concrete indications of economic change including consolidation of quota ownership, and changes in fishing practices to maximise the market value of catch. Annala (1996) confirms both of these effects. For example Annala states that the catch for Hoki (*Macruronus novaezelandiae*) is now spread across the year, rather than concentrated in the spawning ground during the spawning season, with small catches per tow.

Batstone and Sharp (1999) provide some evidence on overall economic performance. They show that fishing employment has grown overall since the ITQ system was introduced. They show that profit levels in 1992 range from a 4.1 to 19.9 percent return on assets across fishing activities. They also show that the total value of the catch has risen considerably between 1986 and 1995. They are unable to compare to previous trends or an estimated counterfactual.

Four studies use more detailed empirical analysis. Using two years of data on quota sale and lease prices from the New Zealand ITQ system, Lindner et al. (1992) attempt to measure economic rents, but conclude that a more thorough analysis of the determinants of quota prices is needed to properly assess market performance and rents. Akroyd et al (1999) consider the time series relationships in lease and sale prices for two species, snapper and orange roughy.⁹ They suggest that quota price information could be used to inform TACC setting but their conclusions are mostly based on theory not derived from empirical evidence. Batstone and Sharp (2000) analyze the effects of market power on lease prices in the snapper fishery and find that some large players have the power to affect the prices they pay or receive when trading quota.

Connor (2000 and 2001b) presents valuable analysis of the changes in quota holdings concentration and fleet (vessel) capacity since the introduction of the QMS in New Zealand. He finds that the major changes in the fishing industry were in ownership of quota rather than ownership of the fishing fleet. He concludes that in New Zealand, the main gains in efficiency were in processing; export marketing; security of supply; and synergies between inshore and expanding offshore operations; rather than improvements in harvest efficiency.

Comparisons with Other International Evaluations

ITQ systems are now used in many countries. Wallis (1999) summarizes some of the systems and evaluations of those systems. Here we simply focus on a few papers that have done detailed empirical analyses similar to ours.

One significant previous empirical study of the efficiency of a similar market is Grafton et al (2000) who studied the Canadian Individual Vessel Quota (IVQ) system for Halibut in British Columbia. They use firm level data on costs, earnings, equipment (gear used and vessel size), and fish landings, for 107 vessels each observed at one of three points in time, 1988, 1991 or 1994. They compare actual efficiency to a counterfactual estimate based on 'best practice' efficiency. They conclude that benefits may take a few years to materialise and can be critically compromised by restrictions on the property right and its transferability. They emphasise the importance of pre-existing regulations and the bundling of property rights. They also find that benefits arise from changes in the quality and price received for fish as well as reductions in cost. Their approach is quite

9. As cited in Clarke et al (2001).

different from ours. They benefit from detailed vessel-specific data while we benefit from a much broader dataset with many species, stocks and years as well as a much simpler regulatory system.

Casey et al (1995) show that in the same IVQ system the timing of fishing changed from being highly concentrated in two months and with the product sold frozen, to being spread across the season and largely sold in the higher value fresh market. They find significant revenue gains relative to the counterfactual of US halibut fishers. On the whole their study is more qualitative and uses only one year (two separate surveys in 1993 and 1994) of survey data.

Connor and Alden (2001) explore some proxies for efficiency in the South East Trawl Fishery of Australia. They find high levels of activity in the leasing market, but low levels of sales. They also find that the number of participants rises over time. They find levels of quota concentration similar to those in New Zealand.¹⁰ They also explore how unused quotas are distributed across fishers. They have no price data.

Other Issues: Environmental and Social Effects

Environmental Effects

We analyse only the efficiency of the market, or the cost-effectiveness of the ITQ system. We do not judge its environmental efficacy. However well a market works, the regulation is not efficient unless it achieves its primary goal. In the NZ case it seems highly likely that the environmental outcome is better than it would have been without the ITQ system. Available evidence indicates that the biological health of the fish populations within the ITQ system is almost never worse off,¹¹ in some cases shows clear signs of recovery, and in other cases is likely to be improving given current TAC levels (Annala et al 2000).¹² Even if these estimates were based on excellent data, however, they are not sufficient to say that the fish stocks and TACs are at efficient levels. That would require data and modeling that is beyond the resources of the New Zealand researchers. Partly because of the scientific uncertainty about sustainable TAC levels, the process of setting TACs is a negotiated one. This raises the risk that industry may effectively resist TAC reductions even when they would be environmentally justified. It is an on-going challenge to maintain balance. Anecdotal evidence suggests that industry pressure to increase or at least maintain TACs has reduced somewhat as the system has matured and fishers are becoming more committed to the long-term sustainability of their stocks.

TAC levels are only environmentally effective if they are monitored and enforced. In New Zealand, the extensive formal enforcement system is complemented by a certain amount of self-monitoring within companies that want to avoid legal action and by other fishers using the same ports and fishing grounds. Despite this, some enforcement problems remain. These are more pronounced in species such as Paua (abalone), which has very high value, where fishing requires little equipment or skill, and can be done from many parts of the coastline.

10. We are exploring concentration in the New Zealand market in other research.

11. One exception is for some geographically specific stocks of Orange Roughy. There are 8 different QMA for orange roughy, stock ORH3B is divided into 6 sub regions, and both ORH2A and ORH1 are divided into 2 subregions (Clement, 2002). A zero catch limit was set for the Puysegur sub-region of ORH3B in 1997/98 (MFish 2001b). ORH10 now has effectively a zero catch limit, of 10 tonnes.

12. There are, however, some important caveats. First, for over half of the fish stocks, sufficient data to measure changes in fish populations over the course of the programme do not exist. Second, the duration of the programme is too short to assess whether measures taken to improve fish stocks with very long lifespans (e.g., orange roughy) are succeeding.

Setting the correct TAC and enforcing it is necessary for environmental sustainability but is not sufficient. Many other environmental issues are directly addressed through regulations on seasons, closed areas, and fishing technology. The ITQ system actually creates other concerns but might also offer a partial solution.

When quota limit total catch fishers have an incentive to increase the value of that catch by 'high grading' or discarding low value fish and low margin species even if the discards would not survive. This can be partly addressed through the use of on-board inspectors. Also the bycatch ratios on inspected versus uninspected boats appear to be somewhat different, so discards are probably not avoided altogether.¹³ Inspectors may also be 'captured' by the fishermen or simply deceived by them.

Quota systems can also encourage increased catch in non-quota species, thus simply moving the environmental problem from one species to another. This apparently did occur to a certain extent in New Zealand but the problem is limited by the very wide scope of the system. As new species become commercially significant, and therefore under threat of over-exploitation, they are added to the quota system.

It is possible that the changes in fleet composition and fishing techniques induced by the quota system could have negative (or positive) environmental impacts. The effects would depend on how behavior changes and to what extent the environment is protected by separate regulations. This would need more exploration.

Finally, it is likely that the quota system enhances incentives and opportunities for internalising the complex externalities involved in how exactly fish are harvested. By changing the fishery from an open access resource to having limited access rights (through the TAC and individual allocation) the group who need to cooperate on optimal management is limited and clearly defined. In at least one case, the Challenger Scallop fishery, this has led to local co-operation and voluntary controls that go beyond the government regulation. Greater concentration in the fisheries could make incentives to cooperate even stronger, though concentration may also have negative economic and social implications.

Social Implications

Protecting artisanal fishers and fishing communities was not an objective during the creation of the New Zealand system. In part this reflected the liberalisation in the New Zealand economy on the whole at that time. Artisanal fishers also did not have an effective lobby group. Small, part time fishers were excluded from the system from the beginning by being denied fishing permits. This made the system more manageable administratively.

Some smaller fishers who did receive allocations may still have been a little disadvantaged by the likely greater variability in their catch history (and, anecdotally, a tendency to under-report catch for tax reasons). There were years of litigation over the initial allocations. Smaller fishers might also have suffered more from the initial adjustment period as the rules were established and monitoring systems were made more user-friendly.

Our analysis thus far shows that there has been net exit of ITQ owners under the ITQ programme, but there are still very many small fishermen active in the industry and new owners do enter. While it appears that exit is oriented more toward smaller owners (as efficiency would suggest is likely to be the case), we have not yet investigated this thoroughly. Also, it is unclear to what degree

13. In the deep-water trawl fisheries, vessels carrying observers have reported larger quantities of non-target QMS species than vessels fishing in the same area that do not carry observers, indicating that discarding has occurred (Annala 1996).

any consolidation is due to the ITQ programme or to other forces. Yandle and Dewees (2000) find some evidence that, at least in the Auckland area, quota owners and companies are more positive and optimistic about the benefits of the quota system than small fishers and those who lease quota.

Many of the smaller, part time and artisanal fishers were Maori (indigenous New Zealanders). Their initial exclusion from the fisheries, combined with wider Treaty of Waitangi concerns about who actually 'owned' the fishery, led to a prolonged negotiation with government over Maori fishing rights. This issue has been addressed in two ways: allocation of quota to Maori, and establishment of customary fishing rights to provide *iwi* (tribes) with access to and control over their local fishing resources.

Maori now own more than one-third of all quota. The Treaty of Waitangi Fisheries Commission (Te Ohu Kai Moana) currently holds these quotas on behalf of all Maori. Negotiations among Maori about their ultimate allocation to individual *iwi* are on-going. The final decisions are not yet made, but it seems likely that sales of quota from Maori to non-Maori will be restricted to ensure that Maori ownership is maintained. Maori are able to use their quota in the same way as anyone else including leasing it to non-Maori. They are not required to fish using customary methods.

Customary catch refers to the traditional Maori right to harvest and gather seafood. Customary fishing regulations, introduced in the Fisheries Act 1996 govern non-commercial customary fishing only, and the harvest cannot be traded. Traditional customary fishing areas are those of special significance to *iwi*(tribe) or *hapu*(sub-tribe) as a source of food or for spiritual or cultural reasons. Provision for management of customary areas was first recognised by the Maori Fisheries Act (1989), and then re-affirmed by the 1996 Fisheries Act. *Taiapure* (local coastal or estuarine fisheries) can be formally declared by lodging a proposal to the Crown.

Iw ior trust groups can establish (through the Crown) *mahinga maitaitai* in larger areas important to customary food gathering. Commercial fishing is generally excluded from these areas.¹⁴ The *maitaitai* is managed by a Tangata Kiaki/Kaitiaki (in the South Island), a Tangata Kaitiaki/Tiaki (in the North Island); *iwi* representatives nominated by the *tangata whenua* (local Maori) and appointed by the Minister. They authorise customary and/or commercial fishing within their *maitaitai*, but are under no obligation to do so if the proposal is inconsistent with the *tikanga* (protocol and practices) of that area.

VII. Potential Implications for Policy Reform

Overall our results confirm those of earlier studies. We find evidence that supports the assertion that fishers behave in a reasonably rational fashion and that the markets are relatively efficient. We do not find major changes in participation in these fisheries as a result of the system. We find evidence that suggests that the ITQ system is improving the profitability of fisheries in New Zealand.

These factors suggest that New Zealand would want to have non-economic justifications for any significant changes to the system. It also suggests that countries considering adopting an ITQ system might not only be optimistic about the potential economic gains, but may also want to replicate some of the features of the New Zealand system that likely have led to market efficiency. These features include simple standardised rules for quota definition and trading across species and

14. To date only two *maitaitai* have been established: Koukourarata, December 2000 (for management of Banks Peninsula cockle beds); and Rapaki *Maitaitai*, December 1998 (Lyttelton Harbour). These *maitaitai* were established under the Fisheries (South Island Customary Fishing) Regulations (1999) (Ministry of Fisheries (2001e).

areas, very few restrictions on quota trading and holding, relative stability in the rules over time, and low levels of government involvement in the trading process.

To date, our study is very incomplete. We will be able to study many more detailed issues with our rich data: bycatch, changes in timing and quality of product, and causes and effects of concentration of quota, among other things. Our evidence thus far suggests that the market is operating in a reasonably efficient manner and is providing significant economic gains. We need to explore the finer details of the market operation more closely to assess economic efficiency more fully.

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Chapter 6

LEARNING FROM THE MARKET: EX-POST WATER ACCESS ENTITLEMENT AND ALLOCATION TRADING ASSESSMENT EXPERIENCE IN AUSTRALIA

by

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I. Background

Australia, arguably more than any other OECD country has made a commitment to develop tradeable markets for access to water resources. The prime driver for this has been the development of a National Competition Policy, involving some monetary transfers of taxation revenue collected by the Commonwealth Government to State and Territory governments being conditional upon stepwise attainment of micro-economic reform goals. The focus of these National Competition Policy reforms has been in areas traditionally managed by State and Territory governments rather than the Commonwealth Government. The general aim has been to impose market disciplines on government enterprises associated with water, electricity, telecommunication, gas and rail. A ten-year reform programme has been put in place with three formal reviews. To gain access to these funds (billions of dollars), State and Territory governments have had to meet performance targets relating to aspects such as pricing, separation of regulation from service delivery, and the development of competitive markets (see Council of Australian Governments, 1994).

In the area of water policy, States and Territory governments now are expected to allow water licences to be held separately from land title and traded via a market process. Progressive transition to integrated catchment management and full cost pricing that includes the cost of externalities is also expected.

As a result, all States have introduced new water legislation and begun a process of water policy reform. New water markets are emerging and new arrangements being trailed. This paper focuses only on one dimension of this policy reform process, namely, ex-post experience in assessing the development and expansion of markets for water access entitlements and periodic allocations.² Most

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1. The views expressed in this chapter are those of the author and do not necessarily reflect those of the institute with which the author is affiliated or the OECD.
 2. It is common for many people to describe water entitlement and water allocation arrangements as water rights. As property rights are defined differently by different academic disciplines and interest groups, it has become clear that ambiguities can be avoided by not using the terms water rights or property rights.

attention is given to water trading in Australia's Murray Darling Basin. A recent evaluation of the pilot interstate water trading trial is used as a case study.

II. Definitional Overview

Unfortunately, terminology used to define opportunities to access and use water varies among states and territories. In fact, as one moves from one state to another the meaning of the word "allocation" can reverse (Carmichael and Cummins, undated). In this paper,

- an **access entitlement** is a legal arrangement that defines the rules by which periodic allocations are made; and,
- a **periodic allocation** is a fixed quantity of water that can be applied to land or transferred to another entity.

An access entitlement is like a pollution licence. In effect, it gives the holder a proportional share of the access opportunity through time. A periodic allocation is equivalent to a permit to extract a fixed quantity of water from a defined water source within a stated period of time. Strictly, allocation is an option to extract. While the licence is owned, the water is not "owned."³

Across Australia, water access entitlements and periodic allocation arrangements were rolled out without any consideration to the need to manage water as a scarce resource where trading would be the norm. As a result, a plethora of licensing systems exist. Trading arrangements have been "bolted on" (Young and McColl, 2002). In the Murray Darling Basin system alone, over 26 different surface water allocation systems are used to define water access entitlements and water allocations. Complex, cumbersome and confusing systems are used to allow trading among systems. Young and McColl (2002) set out a generic framework for a system that is designed to be institutionally robust in the sense that the framework would not have to be completely respecified as circumstances change. The focus of this system is on the clear specification of opportunity and risk, separation of the component parts in a manner that minimises transaction costs, and the management of externalities using independent systems and processes. A well-functioning trading system – if it is to have low transaction costs, be able to respond in a dynamically changing environmental and market conditions and facilitate the efficient management of externalities – needs to be designed with these factors in mind.

Under current arrangements most access entitlements are issued as an irrigation licence – defined in terms of the reliability that a periodic allocation of a volume of water will be made available in any one year. Some systems give all licence holders the same priority, while others define two or more levels of priority. Where there are two types of priority, generally those that have "secure" preferential access to water are known as "high-security licences", and the remainder as general security licences. In the Murrumbidgee River System:

- a typical horticultural producer may hold a 2,500 ML high security licence that for at least 90% of the time would provide an annual allocation of 2,500 ML;
- while a typical rice producer may hold a 2,500 ML general security licence that in a very wet year may provide an allocation of say $0.9 \times 2,500 = 2,250$ ML, but only provide an allocation of $0.1 \times 2,500 = 250$ ML in a drought year; and,

3. It is a common pool resource.

- Licences stipulate among other things the rules by which periodic allocations are made. Once made, all periodic allocations are equally tradeable. Many geographic restrictions on trade apply (Hassall and Associates, 2002).

III. The Condition of the Resource

In both the Murray Darling Basin and the South East of South Australia, both dryland and irrigation salinity are major issues. As a result, water licences are used to both manage extraction and to manage externalities. Typically, an irrigation licence also stipulates how water may be applied to land. Many of the major environmental problems associated with these systems are characterised by significant time delay between water use and the emergence of an environmental externality. The time delay between irrigation and a salinity impact, for example, can be anything from one to sixty years. One consequence of long time lags is that ex-post assessments of policy reforms must be conducted before all the consequences of the reform are fully understood. Another consequence is the simple fact that it can be very difficult to assign cause among reform instruments.

Without policy and infrastructure change, the future health of the River Murray in both quality and flow terms is in question. While it is still too soon to predict an outcome if no changes are made, the general consensus appears to be that the likely extent of the decline in river health would be unacceptable. As a result, a major consultation process has been implemented. It is expected that this process will result in the reallocation of water from irrigation to the environment (Murray Darling Basin Commission Ministerial Council, 2002). The scenarios under consideration explore irrigation access entitlement reductions of between 10% and 20%. No matter how this is done, the market response and consequent socio-economic impact is likely to be considerable.

IV. The Trading Environment

Driven by the “Water Reform Process”, development of a market for water resources has been quite rapid. Three types of trades now occur:

- Access entitlement trades (known as *permanent trades* because ownership of part or all of the irrigation licence is changed permanently);
- Periodic allocation trades (known as *temporary trades* because there is no permanent change to the licence); and,
- Access entitlement *leases* where part or all of a licence is transferred to another entity for a period of time (usually less than 10 years).

Periodic allocations are equivalent, in a legal sense, to a permit to extract a fixed volume of water from a river, a channel or a groundwater source. An access entitlement is equivalent to a proportional share of any allocations that are made from time to time.

Water trading, particularly, temporary water trading has increased rapidly. In the Goulbourn-Murray regions, for example, in 1989/90 there were only 43 temporary trades representing some 22 gigalitres of water. Ten years later, policy changes had been such that the size of this market in 1999/2000 was 3600 temporary trades involving some 204 gigalitres of water (Bell, 2002).

A significant number of water trading exchanges now exist, water brokers are active, the market is becoming more competitive, traded volumes are increasing and participation rates are higher. In the southern parts of the Murray Darling Basin and as indicated in Table 6.1, most irrigators now participate in the water market.

Table 6.1. Proportion of irrigation farm businesses who have not participated in the water market by area in selected parts of the River Murray

Irrigation Area	% of Farm Businesses
Pyramid Hill/Boort (Victoria)	11%
Torrumbarry/Boort (Victoria)	35%
Murray Irrigation Limited (NSW)	12%
Private Diverters Murray Region (NSW)	27%
Private Diverters River Murray SA (Riverland)	61%
Private Diverters River Murray SA (Lower Murray)	45%
Central Irrigation Trust (SA)	85%

Based on inspection of water access entitlement registers and water trading registers up to 30 June 2001.

Source: Bjornlund, H. (2002) "What impedes water markets." Paper presented to the 4th Australasian Water Law & Policy Conference, Sydney, October, 2002.

Trading activity and participation is heavily dependent upon seasonal conditions, market prices and the quantity of water that is available in the system. The figures in Table 6.2 illustrate the extent of activity in one irrigation area. Participation in the market varies from year to year. A range of different market processes and settlement systems also apply (Bjornlund, 2002).

Table 6.2. Percentage of irrigation farms involved in temporary trading in the Murray Irrigation Limited region (NSW)

Year	% of farms in involved in <i>Buying water</i>	% of farms in involved in <i>Selling water</i>	% of farms <i>Not trading water</i>
1995/96	23	23	54
1996/97	21	15	64
1997/98	40	29	30
1998/99	39	32	30
1999/00	45	25	30
2000/01	37	30	33

Source: Bjornlund, H. (2002) "What impedes water markets." Paper presented to the 4th Australasian Water Law & Policy Conference, Sydney, October, 2002.

V. Market Assessments

Methodologies and Counterfactuals

The OECD Workshop organisers specifically asked participants to include descriptions of methodologies for assessment of trading arrangements and to consider difficulties associated with defining appropriate "counterfactuals" when seeking to evaluate the specific contributions of trading permit regimes.

With regard to the issue of counterfactual arrangements, Australia has invested considerable resources in developing biophysical modelling systems that provide a framework for the assessment of costs and benefits. Two initiatives stand out:

- The *first* is the Salinity Audit prepared by the Murray Darling Basin Commission that provides an assessment of the likely salinity loads in the River Murray in 2050 and 2100 (Murray Darling Basin Ministerial Council, 1999). Among other things, the authors of this audit observe that the average salinity of the lower River Murray (monitored at Morgan) will exceed the 800 EC threshold for desirable drinking water quality in the next 50-100 years. By 2020 the probability of exceeding 800 EC will be about 50 per cent.

- The *second* is the National Land and Water Resources that sought predictions on the likely condition of Australia's agricultural resources in 2000, 2020 and 2050. The models used do not include assessments of likely changes in technology, prices or institutional arrangements. Nevertheless, they have had a profound impact on policy development assessments. For example, the recent preliminary assessment of the costs and benefits of environmental flow enhancement for the River Murray was built upon a baseline scenario of increasing river salinity and, hence, declining regional income (see Young *et al*, 2002). This part was not considered to be controversial.

Nevertheless, development of social and economic baseline or counterfactual scenarios is still in its infancy in Australia. The National Land and Water Resources Audit, for example, observes that depending upon assumptions about future price and cost conditions, the most appropriate counterfactual for Australian agriculture over the next 20 years is a 30 to 50% decline in the number of people who describe themselves as farmers or farm managers (National Land and Water Resources Audit, 2002). Most of the social scientists involved in the work summarised in this paper are aware of this underlying consideration, but do not use this information in the counterfactual assessments that they develop. Nevertheless, as prediction and identification of processes due to other considerations tends to require analytical courage, most continue to prefer to assume away reality. Too often, the result is a highly misleading conclusion.

Hall *et al.* (1994) estimated that by making water licences tradeable, net income across the Murray Darling Basin would increase by around 5% per annum. Since then, however, there has been no attempt to measure the gains that have been achieved either in terms of net income or in terms on net social benefit (see Hall *et al.*, 1994). More recent studies have suggested similar gains.

Methodologies used for assessments of water trading arrangements are summarised in Table 6.3 below. Most methodologies tend to mix collection of data ex-post with the development of a benchmark standard. Five broad approaches can be identified:

Table 6.3. Methods used to conduct assessments of water trading arrangements
(Notes to this table are provided as endnotes)

Method	State of the art
Surveys of action, attitude to change and perception of the situation	Designs usually give little reference to the change of the resource even when that is expected. Examples include Syme <i>et al</i> (1999) and, also, Tisdell <i>et al.</i> (2001) analyses of fairness in process and reallocation of rights. These are often classified by experience with water trading and/or the nature of the licences held.
Surveys of actual transactions that are occurring	The counterfactual is covered largely via reference to either a theoretical benchmark, a government policy goal or the actions of non-participants in the market. Examples include Young <i>et al</i> (2000) review of interstate water trading and the work of Bjornlund and McKay (2000) on water market transactions.
Comparison with a hypothetical reference point such as a perfectly competitive market analysis	These reviews are usually conducted by focusing on the nature of the process rather than changes in the state of a resource. For example, "are water licences separated from land title and hence is it legally possible for a desired outcome to occur?" A recent example is Hassall and Associates and Musgrave (2002) recent review of impediments to water trading in the Murray Darling Basin
Simulation models referenced against a formally defined counterfactual	These models are increasingly characterised by the inclusion of biophysical process information with social and economic information. One of the most widely respected is the SALSA model developed by ABARE with input of biophysical data provided by CSIRO (2000). This was used in the preliminary assessment of environmental flow enhancement for the River Murray and has been used for several assessments of the likely economic benefits and costs of changes to water allocation and trading policy.
Comparison with a benchmark defined by government policy	As a result of the COAG process and intergovernmental agreements Australian governments have been prepared to state policy goals and, following implementation of the policy, commission independent review. For example, the recent review of the pilot interstate water trading trial was required to determine whether or not the salinity impact of each trade had a neutral impact on river salinity.
Comparison with industry and other benchmarks of "best" practice	Best practice is defined usually by reference to national and international studies. The above mentioned pilot interstate water trading trial, for example, observed that forms spent a total of 35 days per trade in the mail between the various government, brokers and licensees involved.

Within Australia, it is not uncommon for assessments to use an extensive set of assessment criteria. For example, Young and Hatton MacDonald (2000) in their assessment of water trading arrangements in the South East of Australia, use the following criteria:

- **Environmental protection/improvement** – the trading system should not result in any backsliding in terms of environmental standards;
- **Productive and allocative efficiency** – trading encourages water resources to be assigned to their most valuable use and to be applied efficiently;
- **Low information requirements** – information requirements should be kept to a minimum – often this means asking: "What do administrators really need to know?";
- **Low administrative costs** – the cost of setting up, enforcing and reviewing the system should be kept low. Simplicity is often the key;
- **Equity** – no group should be severely disadvantaged by the trading system and it should not allow individuals to hold others to ransom;
- **Adaptability** – the system should be able to adapt easily to changing technology, prices and climatic conditions;
- **Dependability** – the system should keep use within sustainable limits as they are redefined through time, even when information about the likely responses is uncertain;
- **Continuing incentives** – resource users should have an incentive to exceed targets including environmental standards and efficiency of resource use; and,
- **Permanence** – the trading and allocation system should not be vulnerable to erratic or sudden changes.

Virtually all assessments that have been undertaken give careful consideration to allocative efficiency with or without consideration of equity (fairness) considerations. A key feature of the work on fairness is a focus on fairness in processes used to assess the case for, negotiate, and implement the change (Syme *et. al*, 1999). Assessments prepared for governments typically tend to be broader in their focus than those under taken primarily from a research perspective.

As yet, no ex-post assessments of markets have made extensive use of non-market valuation techniques. Instead, the focus has been upon “switching” or “break-even” approaches to the evaluation of environmental outcomes. For example, the recent preliminary economic assessment of environmental flow enhancement in the River Murray took a break-even approach and found that for some scenarios benefits were greater than costs even if there was no environmental improvement. Taking a different approach, the assessment of pilot interstate water trading trial assessed performance against government policy targets and commitments.

An Ex-Post Assessment of a Pilot Inter State Water Trading Trial

Pushing the frontiers in water trading, States that use the River Murray and its tributaries agreed to conduct a trial of permanent water trading among States. Pragmatically, the trial was to be conducted over part but not all of the system – from Nyah to the River Murray Mouth (see Figure 6.1).

Figure 6.1. The trial pilot trading area



Legislation to facilitate the trial stated that at the end of the first two-years of the trial, trading would be reviewed. Following a competitive tender process, CSIRO an independent research organisation was commissioned to undertake the review. CSIRO's Policy and Economic Research Unit led the assessment. Other members of the team came from Adelaide University, the University of South Australia and a private consulting firm that specialises in hydrological assessments of water issues. Appendix 6.1 contains the executive summary of that review.

The pilot interstate water trading trial commenced in October 1998. The review period was from then up until September 2000. Applications for 53 trades were received and all but two trades approved. A total of 9.8 GL was traded.

A summary of interviews with water brokers and information obtained from Commission records indicates that:

- There has been a net trade of water into the trial area of approximately 30 GL from Victoria;
- Significant amounts of NSW water is trading from the Murrumbidgee Irrigation Area into the trial area;
- There has been a net transfer of water rights to South Australia (Table 6.2);
- Less than 1% of the water traded across State boundaries was being used at its origin (i.e. virtually all the water being traded permanently was sleeper water⁴);
- Trades occurred in every direction except from South Australia to Victoria (see Table 6.4 and 6.5);
- The 51 trades involved 35 legally separate entities buying water;

4. Sleeper water is water that has been allocated to an individual but never used by that individual. It needs to be appreciated that in many cases this 'unused' water remains in the system and is eventually reallocated to others.

- At least six trades involved inter-dependent entities (effectively the one inter-dependent parties moving water from one location to another);
- Brokers were involved in 100% of trades involving people at arms length from one another;
- Most trades involved a broker acting for the seller and a separate broker acting for the buyer;
- Victorian residents are buying water and "holding" it in South Australia where it can be held without having to attach it to a land title and without completing an environmental clearance; and,
- South Australia has received 90% of all trades, New South Wales 7% and Victoria 3% (see Figure 6.2).

More information describing the results of this review are summarised in the tables and figures below and in the appendix to this paper.

Figure 6.2. Net volumes traded among States
Assuming all transactions outstanding as of 30 September 2000 are completed.

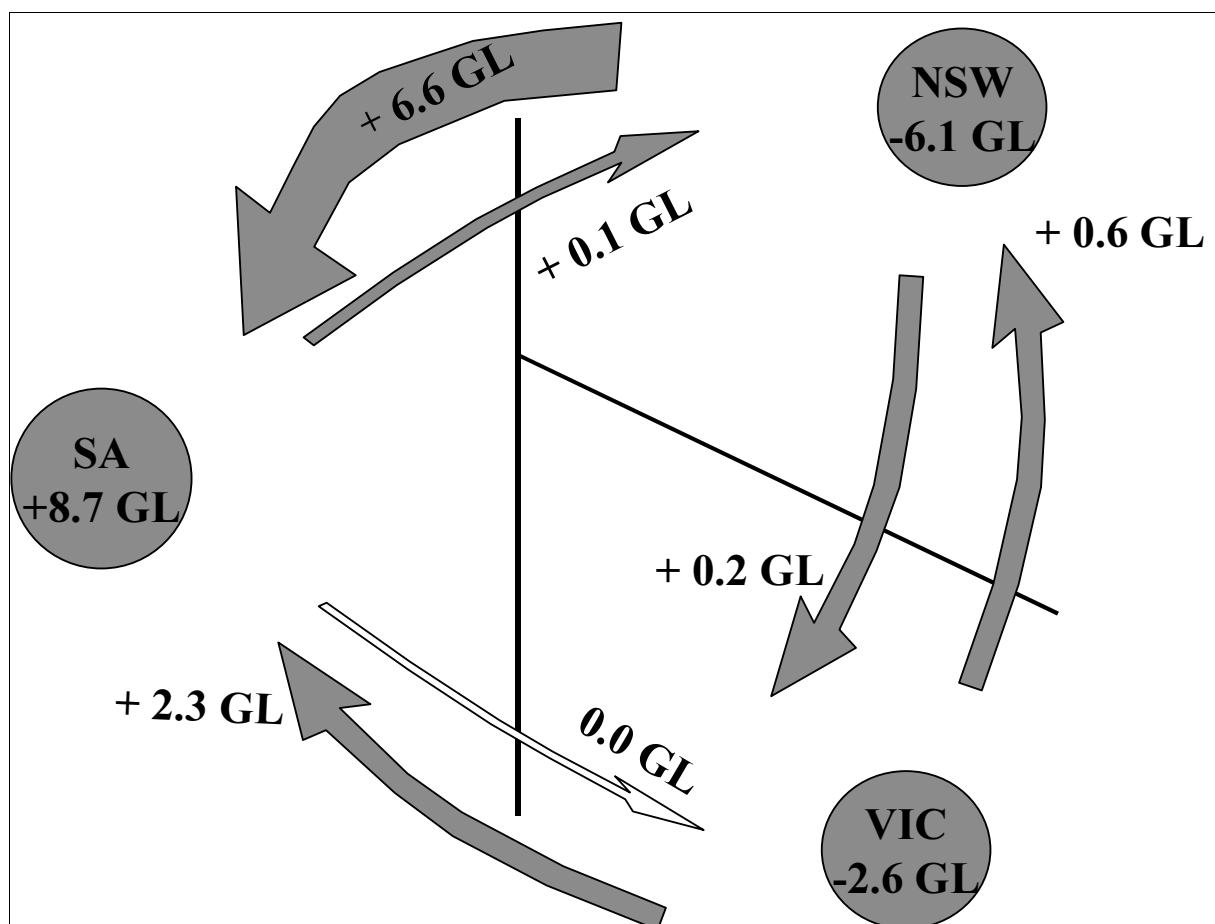


Figure 6.3. Cumulative volume of water traded by source and destination assuming all outstanding transactions are completed.

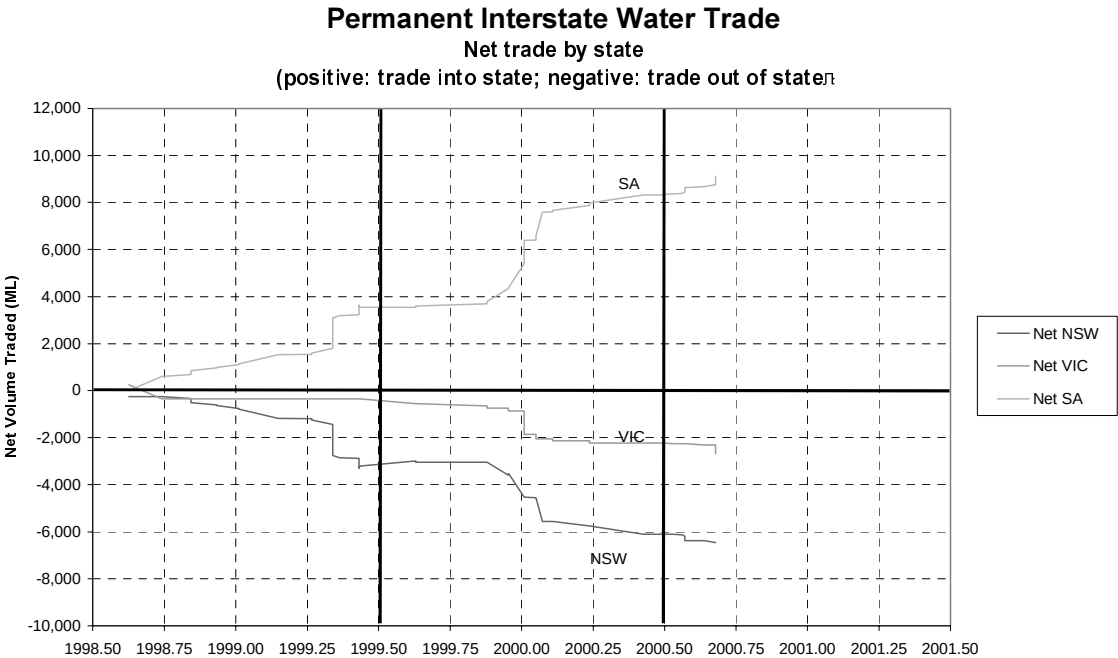
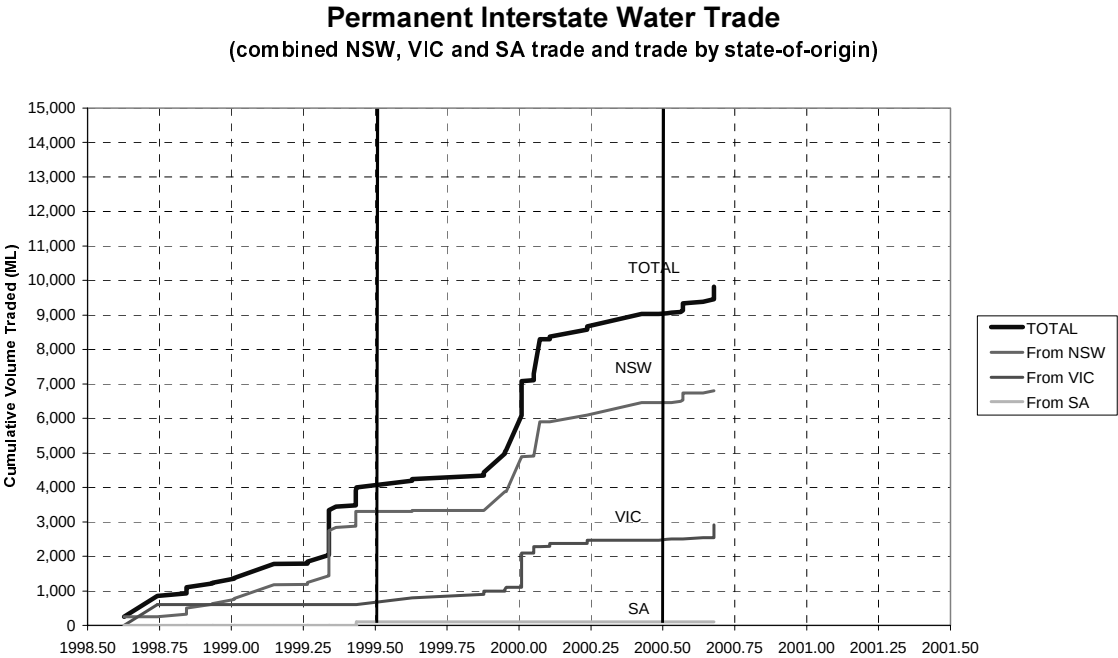


Table 6.4 Number of Trades by Origin and Destination*

Origin	Destination			Total
	NSW	Victoria	SA	
NSW		1	33	34
Victoria	2		14	16
SA	1	0		1
Total	3	1	47	51

* Excludes two trades on hold because purchaser in breach of licence conditions.

Table 6.5. Value of water traded during the trial period*

<i>Size of purchase</i>	<i>Number of purchases</i>	<i>Estimated Value</i>
0 ML - 99 ML	24	\$823,830
100 ML - 499 ML	21	\$3,834,600
500 ML - 999 ML	3	\$2,118,900
1,000 ML	3	\$3,150,000
Total	51	\$9,927,330

Table 6.6. Permanent Trade in the Trial Area, Nyah to the SA Barrages¹

<i>State from</i>	<i>ML</i>	<i>Trade out (ML)</i>	<i>Inter-State as % Allocation</i>	<i>Trade of</i>
NEW SOUTH WALES	216,504	6,805		3.14
VICTORIA	394,940	3,261		0.83
SOUTH AUSTRALIA	495,000	100		0.02
GRAND TOTAL	1,106,444	10,166		0.93

1. Trade as a percentage of the High Security Access entitlements and Pumped Districts within the Pilot Zone, September 1998 to October 2000

VI. Overview of the Evaluation of the Tradeable Permit Regime

Objectives of the Assessment

The formal objects of the trial were as follows⁵

- (a) to facilitate and promote the interstate transfer of water allocations co-ordinated by the Commission;
- (b) to provide for the Scheme to be implemented on a trial basis in that it will initially only apply to specific water diverters and districts, and to high security water allocations and to a restricted part of the Mallee region;
- (c) to improve the efficiency and effectiveness of consumptive water use in ways which facilitate environmental sustainability but which do not increase or accelerate environmental degradation;

5. As set out in Schedule E of the Murray Darling Basin Agreement

- (d) to establish a procedural framework and set of standards so that the Scheme is accountable and does not result in increased levels of salinity, reductions in environmental flows or degradation of the natural environment;
- (e) to provide for the suspension of the operation of this Schedule if there is an increase in, or acceleration of, environmental degradation from the use or management of water that has been diverted interstate in accordance with the Scheme;
- (f) to give effect to water trading arrangements in accordance with the terms of the Ministerial Council's decision regarding the long-term level for off-stream diversions in the Murray-Darling Basin; and
- (g) to enable the Commission to adjust the quantity of water to be delivered to the States as a result of the Scheme.

Institutional Framework and Motivations for Evaluation

The motivation for legislating for an assessment at the end of two years was a clear indication that interstate water trading was a new area for States to enter into. Each State defined water access entitlements in a different manner and all were concerned that the system could result in serious problems. The language used to define the trial also indicates this. The project was called a “trial” or “pilot.”

Use of this cautionary approach enabled the States and the Commission to support the process without having to make a long-term commitment. The national water reform process committed States to progress towards development of an open market and this mechanism enabled them to explore the implications of moving to a Basin-wide market.

As several States and the Commonwealth were involved, it was obvious that the review would need to be independent. Independence removes the opportunity for administrators to cover up mistakes.

Description of the Evaluation Methodology

The methodology chosen for this assessment was to assess trading arrangements against a series of benchmarks. Some of these arise from legislated arrangements, some from policy decisions and some from knowledge of best practice. Allocative efficiency and administrative efficiency were two key considerations.

The approach taken was to assess what happened with *each and every* interstate trade, compare this with knowledge of trading arrangements within as distinct from among States, and to collect data enabling effectiveness to be assessed. That is, the review took the shape of an audit designed so as to maximise learning opportunities. This approach is unusual but makes it possible to begin the review process at a much earlier stage. It also improves opportunities to use qualitative as well as quantitative information.

On a confidential basis, full access was given to all Murray Darling Basin Commission records so that an audit-like assessment could be undertaken. This made it possible for the review team to contact all license holders and all brokers involved.

Focus group meetings were also held with community representatives, irrigators, brokers and water supply companies to assess implications.

Overview of the Main Results

As summarised in the appendix, it is clear that interstate water trading was contributing to an increase in allocative efficiency as defined by the market. Virtually all the access entitlements traded were not being used at their original location and, once traded, were being put towards very high value irrigation. In practice, however, it needs to be appreciated that much of this water, while unused at the location where it traded from, was subsequently reallocated to another users. Thus, the efficiency gain from these trades is the difference between the value of use at the location where it was traded to and the average value of use in the region. Nevertheless, the review observed that introduction of interstate trading had resulted in trades whose aggregate value represented around \$10 million, and that this had resulted in over A\$100 million of investment in infrastructure.

A major issue for the assessment was whether or not interstate water trading would have the desired effect on environmental outcomes. The assessment of environmental outcomes was qualified and concluded that:

“From an environmental perspective, our findings are qualified. The environmental flow impact of inter-state water trading has probably been positive but, in all reality, the 10 GL is so small in flow terms that this gain is probably impossible to measure.

“States admit that monitoring and enforcement of plans and licence conditions is a problem. **From a salinity perspective and in the long-run, inter-state trading can be expected to have a negative impact on river salinity.** Most water is being transferred to South Australian land that has not been previously irrigated with the consequence that river salinity can be expected to increase. South Australia, however, is aware of this and is in the process of putting in place arrangements to ensure that salinity impacts remain within acceptable levels. The new salinity strategy is expected to form the backbone of this initiative. As part of this process, it will be critical that salinity prevention obligations, and their equivalent in other States, are implemented and maintained. “If adequate arrangements are put in the place, then the long-run net effect of recent trades could be neutral. For this to occur, end-of-valley targets will have to be set, and appropriate incentives and administrative arrangements put in place.”

This assessment of environmental impacts and the language chosen, highlight the difficulty in conducting an ex-post review of a trade before all its ramifications can be assessed. In every case, the trade had occurred and the water now being put to a different use but at the time of the “ex-post” assessment none of the salinity impacts and many of the other environmental impacts had yet to occur. States claimed that arrangements were going to be put in place to manage them but, at the time, this still had not happened. There was a commitment but, as of the time of the assessment, unconvincing action. This highlights the importance of establishing a temporally-relevant counterfactual. Most of the water access entitlements traded had not been used but this did not mean that the water was unused. In most cases, it meant that the water was used elsewhere with the consequence that it did have salinity impacts but that they were probably less than the would be in the more saline South Australia. Consistent with the Tinbergen Principle, most economic policy analysts, however, would argue that for maximum gains separate instruments should be used to control trade in volumes of water and control salinity and other externalities (Young and McColl 2002).

Administrative efficiency was also found to be very low. In fact, so low that brokers who had a vested interest in having the scheme expanded recommended in interviews that this dimension of the trial be improved before it was extended. This was particularly interesting observation as some States and administrators were clearly of the view that the trial should be extended to cover all the southern

connected River Murray system, not just the southern core part downstream from Nyah (see Figure 6.1).

Bouncing of water into the trading area was an issue. Strictly, the trial was to be limited to a subset of the River Murray system. Irrigators, however, soon worked out that they could trade water from outside the trial interstate pilot trading area from another part of the State and then use the new interstate trading arrangements to move it to another state. This became known as “bouncing” and was quickly opposed by the State administrators involved. While broker collusion was also evident, all agreed that they would not support the practice. Economic analysis undertaken during the assessment, however, quickly revealed that the invisible hand of the market was at work. While overt bouncing was stopped, within-state trading arrangements in Victoria in particular meant that price signals were being transmitted from central Victoria to the river mouth. The market was bouncing that which administrators had sought to stop.

Comparison of Results with Other Evaluations

Comparison with other assessments of interstate water trading was not possible. Comparison with assessments within States, however, revealed many administrative inefficiencies. If the goal is a perfectly competitive market, then the baseline hypothesis is that the cost of within and interstate trading should be similar. Typically, interstate trades took much longer than within-state trades. The report said:

“Illustrative of the complexity of the current system, trade documents can easily spend a total of 32 days in the post moving from one location to another.

“Settlement procedures could be made consistent with best practice in dealing with property transactions. A system more like the existing land registration system would give buyers and sellers much greater confidence – the examples are right in front of us.

“Current water dealing practices and procedures are much more lax than those that apply to land - even though many of the transactions are of similar value and implication. There are significant opportunities to improve licence registration arrangements and record keeping procedures. Face-to-face meetings among the officers who actually manage the process could assist to this end. Many of the problems result, however, from complexities in State record keeping systems. The problems are State and Basin wide, not inter-state specific.”

VII. Concluding Comments

As a result of the review it was decided by the Murray Darling Basin Commission to continue the trial and work to improve trading arrangements and, in particular, to improve administrative processes. As a result, several new projects were set up to improve assessment of the salinity impacts of inter-state and within-state trading.

From a tradeable permit system design perspective, the general literature emerging from Australian research draws attention to:

- The importance of designing the access entitlement and allocation system to facilitate efficient trading;
- Trading systems that simply grow out of a regulatory environment result in unnecessary and administratively complex arrangements;

- The value of trialling or piloting new trading arrangements;⁶ and
- The importance of considering more than just economic efficiency criteria.

At a more general level, questions must be asked about the purpose and cost of ex-post reviews. Two different styles of review can be identified – those that seek to improve the efficiency of future reforms and those that seek to improve reform processes currently in place. For the latter, it is our observation that there is a strong case for independent review at a very early stage so that errors are identified and corrected as quickly as possible. Structured qualitative interviews with participants in the process can be particularly valuable as those who bear the costs as well as the beneficiaries of the reform process can often see and draw attention to issues and problems not easily identified from official data.

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6. Australian Governments are in the final stages of funding \$5 million of trials for market-based instruments (MBIs). The essential criteria are:
- Fill gaps in knowledge needed to design, apply and/or evaluate MBIs;
 - Address salinity, water quality and/or biodiversity and run in NAP region/s;
 - Be conceptually robust – well-grounded in economic principles and good quality biophysical information, and a sound understanding of biophysical processes;
 - Engage relevant agents, such as the private sector, institutions, landholders and government;
 - Incorporate a sound implementation strategy that identifies and manages risks in implementation (including consideration of any adverse or perverse effects);
 - Incorporate appropriate strategies for evaluation and knowledge transfer; and
 - to overcome impediments to the practical application of MBIs “on the ground”;
 - Develop cost-effective solutions to the particular national resource management problem when compared to or combined with more traditional approaches (such as regulations, zoning, compliance or penalties and charges); and
 - Provide for joint collaboration across a number of jurisdictions.

APPENDIX 6.1

EXECUTIVE SUMMARY OF “INTERSTATE WATER TRADING: A TWO YEAR REVIEW

A6.1. Inter-State Trading Arrangements

The pilot project has allowed 51 trades involving 9.5 GL of water to move across the borders of South Australia, Victoria and New South Wales. Collectively, these trades are worth over \$9.9 million. Three trades were worth more than \$1 million. In net volume terms, more than 90% of this water has moved to South Australia.

The approach taken by those responsible for implementing the trial, as would be expected, is to learn by doing. Changes are, and have been made, along the way. Provided that this spirit of adaptation and willingness to solve problems as they emerge continues, we see no reason to stop the trial.

In terms of the scale of irrigation from Nyah to Murray Mouth, the total volume of water involved so far is less than 1/100th of the total water applied in the area. Too much attention could easily be given to perfecting inter-state trading arrangements rather than the intra-state trading arrangements.

The inter-dependent nature and scale of the intra-state market is such that market signals are being sent from Shepparton to Lake Albert. Similar signals are being sent from the Murrumbidgee Irrigation Area in NSW to Lake Albert. The invisible hand of the market is encouraging water to trade into the trial area, others trade only within it. Since the mid 1990s, mostly in the last two years, over 30 GL has traded into the Sunraysia District from the rest of Victoria while less than 2.9 GL has moved into NSW and South Australia.

Intra-state trading is driving the market for water. Inter-state trading arrangements keep the various markets in place.

A6.2. Administration Costs and Burdens

A surprising recommendation from brokers is that the official trial area should not be expanded until administrative arrangements are improved significantly. Illustrative of the complexity of the current system, trade documents can easily spend a total of 32 days in the post moving from one location to another.

Settlement procedures could be made consistent with best practice in dealing with property transactions. A system more like the existing land registration system would give buyers and sellers much greater confidence – the examples are right in front of us.

Current water dealing practices and procedures are much more lax than those that apply to land - even though many of the transactions are of similar value and implication.

There are significant opportunities to improve licence registration arrangements and record keeping procedures. Face to face meetings among the officers who actually manage the process could assist to this end. Many of the problems result, however, from complexities in State record keeping systems. The problems are State and Basin wide, not inter-state specific.

The South Australian legislation allows people to hold water access entitlements without owning land that can be irrigated. New legislation in NSW will allow a similar mechanism to be set up in this

State. Separation of volumetric trading from access or environmental considerations simplifies administrative procedures. The question of what environmental impacts, if any, are allowed then becomes a matter for State consideration and reporting to the Commission.

A6.3. Environmental, Economic and Social Impacts

From an economic impact perspective, and before any account is taken of environmental or social considerations our conclusion is an unequivocal one. Inter-state trading is increasing the value of water use in the Murray-Darling Basin. Virtually all (99%) of the water sold was not being used by sellers. During the first two years, virtually all water has gone to high value uses. Around three-quarters has gone into new irrigation development using state-of-the-art technology. The value of some of the transactions involved exceeds \$0.5 million and in some cases over \$1 million.

From a social impact perspective, inter-state trading during the two year trial period has had no measurable adverse social implications for the districts that have sold water inter-state. The water, in all but 0.4% of cases was not being used by the seller. In a significant number of cases, the revenue resulting from the sale has been used to finance installation of water-saving irrigation systems. We do acknowledge, however, that there is a slight diffuse impact as the presence of unused water makes it possible to allocate slightly more water to all other water users in the Basin.

From an environmental perspective, our findings are qualified. The environmental flow impact of inter-state water trading has probably been positive but, in all reality, the 10 GL is so small in flow terms that this gain is probably impossible to measure.

States admit that monitoring and enforcement of plans and licence conditions is a problem. From a salinity perspective and in the long-run, inter-state trading can be expected to have a negative impact on river salinity. Most water is being transferred to South Australian land that has not been previously irrigated with the consequence that river salinity can be expected to increase. South

Australia, however, is aware of this and is in the process of putting in place arrangements to ensure that salinity impacts remain within acceptable levels. The new salinity strategy is expected to form the backbone of this initiative. As part of this process, it will be critical that salinity prevention obligations, and their equivalent in other States, are implemented and maintained. If adequate arrangements are put in the place, then the long-run net effect of recent trades could be neutral. For this to occur, end of valley targets will have to be set, and appropriate incentives and administrative arrangements put in place.

Experience with trades from High Impact to Low Impact areas in the Nyah to Victorian Border region, one way of simplifying environmental assessment procedures would be to establish a set of salinity exchange rates for transfers from one river reach to another.

With regard to environmental degradation at each trading destination, our conclusion depends upon the degree to which plans are enforced and the adequacy of the standards they set. All States express problems in monitoring compliance with these plans, so one can not be confident that the final outcome will be consistent with the goal set - "a procedural framework and a set of standards that ... does not result in increased levels of salinity, reductions in environmental flows or degradation of the natural environment."

A6.4. Opportunities for Improvement

Markets place pressures on all administrative systems. The pilot project provides an opportunity to see where the processes need to be revised and reworked.

Protecting water quality is a long-term goal and the fundamentals need to be put in place today while the volume of inter-state trade is low. There is an opportunity to put in place a system that defines obligations over the long term to the environment. The Salinity Strategy is one initiative that is defining the long-term policy goals. Another part of a long-term strategy will be a system of salinity debits and credits and a register of salinity prevention obligations and attach these conditions to the water. Where salinity impacts are not going to be felt until far into the future, long-term provisions need to be made to address or mitigate these impacts.

Exchange rates are poorly understood by the buyers and sellers in the market. If market forces change in the future such that incentives exist to move water from SA to NSW, the exchange rate will act as a tax on trade. Is an exchange rate the right mechanism? If it is, it will be important to communicate this and provide a justification for the numbers being used.

All States need to move to improve the mechanisms for enforcement. For example, SA has been requiring Irrigation and Drainage Management Plans for some time but now these plans are being linked to the Water Allocation Plans. As Water Allocation Plans become statutory documents opportunities for enforcement increase. A related issue is the legal status and long-term enforceability of Salinity Prevention Obligations and their equivalent in other States.

To ensure long-term delivery of the trial's environmental objectives, Salinity Prevention Obligations should be recorded on the licence. Failure to comply with the obligation should result in the sale of sufficient water to finance restitution of the obligation. In cases, where the purchaser is required to set aside money, this money should be put aside in a trust account.

Our review, also makes it clear that there is inconsistency in the pricing and charging mechanisms in use across the Basin. In many cases, South Australia's salinity prevention obligations have a similar effect to Victoria's salinity levy, but is less transparent. However, attention needs to turn to raising fines and penalties in SA because as it stands, it is cheaper to pay the fine for exceeding one's allocation rather than buy water on the temporary market.

Finally, the administrative systems need to be stream-lined and harmonised over the long term. A first step is for the water licensing officers to meet and identify the simple things that can be done to improve the processes of approving trades such as adopting common forms and guidelines where possible. During this meeting, we suggest that they explore the option of deeming that all permanent trades take effect from the 1st July after the date of settlement. Where appropriate, a permanent interstate trade should be accompanied by temporary inter-state trade.

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Chapter 7

COMPENSATING FOR THE IMPACTS OF WETLANDS FILL: THE US EXPERIENCE WITH CREDIT SALES

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I. Introduction

Wetlands ecosystems are characterised by recurrent shallow inundation or saturation at or near the surface of the soil (NRC, 1995) and, based on the hydrologic regime and vegetative cover, can be categorised into wetlands types such as forested, shrub-scrub and bogs (Cowardin, et. al., 1979). Wetlands also can be described as a natural capital asset that may provide hydrologic services of flood and drought remediation, water quality services of sediment and nutrient assimilation and wildlife habitat services such as nursery and feeding area. The level of services from any particular wetlands depends on the wetlands type and its location in the landscape. (Mitch and Gosselink, 2000; NRC, 2001).

Analyses conducted during the 1970s found that about 50% of the United States' original wetlands had been converted to agricultural and urban land uses, making wetland loss a matter of policy concern (OTA, 1984; Dahl, 1990; Dahl, 2000). While not all wetlands provide all the possible services, the concern about historical wetlands loss led to a stated national goal of no net loss (NNL) of wetlands acres and services from future development. (Conservation Foundation, 1988; NRC, 2001) NNL has become the foundation for the nation's principal regulatory programme governing wetlands alterations.² This federal programme, created by Section 404 of the Clean Water Act, is administered

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1. The views expressed in this chapter are those of the author and do not necessarily reflect those of the institute with which the author is affiliated, or the OECD. Partial support for this work was provided by the Resources for the Future Fellowship in Environmental Regulatory Implementation. I wish to thank Paul Scodari for his critical review of this manuscript and assistance in its development.
 2. There is a much richer mix of wetlands policies than just the federal regulatory programme. For example, US agricultural policy includes a provision that denies farm operators access to price and income support programmes if they drain or fill a wetlands on their property. Also, the Wetlands Reserve Program purchases perpetual easements from farmers who agree to stop farming former wetlands areas and return the areas to wetlands status (Heimlich, et. al., 2000).

by the U.S. Army Corps of Engineers (Corps) with oversight by the US Environmental Protection Agency (EPA) (Strand, 1995; GAO, 1991). In addition, individual states have wetlands permitting authorities that either complement or expand upon the federal permitting programme (Zinn and Copeland, 2001).

Under Section 404 anyone wishing to place fill material in an area delineated as a wetlands, and that falls under the legal jurisdiction of Section 404, is required to secure a permit from the Corps.³ If a permit is issued, the permittee has the legal and financial responsibility to restore degraded wetlands not affected by the permitted fill or create new wetlands from uplands.⁴ The expectation is that this “compensatory mitigation” will offset the permitted loss of wetlands and result in no net loss of wetlands area and services. These mitigation wetlands, called wetlands “credits”, are measured as acres of a wetlands type or as a weighted-index of wetland services.

For the most part, the creation of credits has been the responsibility of the permittee. For reasons to be described below, some permittees did not provide either the quantity or quality of credits necessary to secure the NNL goal, especially for the habitat service. To address this problem, wetlands programme administrators at first cautiously allowed and then later actively encouraged private entrepreneurs to make investments in producing wetlands credits for sale to permittees. The logic was as follows. Regulatory programme administrators would certify the ecological - usually habitat - quality of private entrepreneurs credits before they could be sold for mitigation. Then, these same regulators would assess the wetlands acres and services lost when they issued a fill permit, and allow permittees to purchase credits from certified sellers to offset the wetlands loss. In this way, the NNL goal would be served. In conceiving this programme, private sellers were expected to comply with credit quality assurance requirements when seeking regulatory certification as a seller. They would then compete on the basis of price when offering credits to permittees. With the minimum acceptable credit quality assured, credit price would fall toward the long run average cost of production. If the suppliers were private entrepreneurs, then the full cost of credits supplied would be included in the credit price, including the opportunity cost of invested capital and returns to management. Competition among credit sellers would drive credit prices towards minimum cost, but there would be no cost subsidy to recipients of fill permits. (For an early paper that makes these arguments, see Shabman, Scodari and King, 1994).

The production and sale of wetlands credits has been cited as an example of a market-like approach to environmental management (EPA, 2001; NAPA, 2000). In fact, the effort to improve the availability and quality of mitigation credits through this sales programme has had some success, where mitigation has been secured using private sector credits. However, the early hope for vigorous

3. Much that remains under debate is embedded in this apparently simple and definitive sentence. Section 404 does not use the term “wetlands”, instead referring to “waters of the United States”. However, court rulings have established that waters of the United States included “wetlands”, as that term came to be understood after passage of the Clean Water Act in 1972. These rulings led to other questions: “what are wetlands?” and “which wetlands can the Corps regulate?” Agreement on what a wetlands is took over 15 years to achieve and required a National Academy of Sciences panel report (NRC, 1995). However, the popular image of wetlands is a place with visible water at the surface of the land. This image conflicts with a scientific definition in which wetlands may be places where water rarely reaches the surface. This conflict between the general image of a wetlands and the NRC definition remains a source of contention about the scope of the permitting programme. Meanwhile, the scope of wetlands (however defined) subject to S404 regulatory jurisdiction remains in dispute. As recently as 2001 the United States Supreme Court decision limited the Corps permitting jurisdiction over so-called isolated wetlands (NRC, 2001). A related ambiguity affecting S404 implementation is the definition of “fill”. Recall that S404 only regulates the placing of fill material in waters of the United States, but recent court decisions (NRC, 2001) have invalidated the Corps definition of fill. For a comprehensive review of the origins and interpretations of Section 404, through 1995, see Strand (1995).
4. Recent reviews have argued that restoration of degraded wetlands, for example re-flooded drained farmland, is always preferable to creating wetlands from uplands (NRC, 2001).

price competition has not been realised. Regulatory conditions have created barriers to market entry and have created thin markets characterised by little price competition. As a result, today only a small amount of the total compensatory mitigation required by regulatory programmes is provided by private credit sellers.

This paper is a review of the U.S. experience with the private credit sales programme and factors that have limited its development and use. That experience suggests an alternative institutional structure could widely expand the reach and usefulness of this mitigation option. The origins and current status of the wetlands credit sales programme are first described. This is followed by an explanation of how the regulatory rules governing the overall wetlands permitting programme, as well as specific rules governing credit sales, have prevented the development of a robust market in buying and selling of wetlands credits. The paper concludes with proposed reforms that would rely on market-like principles to expand the quantity and lower credit prices while assuring that wetlands credit sales move the nation toward the no net loss goal.⁵

II. Wetlands Credit Sales: Origins and Current Status

Before a fill permit is issued by the Corps of Engineers, regulators must conduct a review called “sequencing”.⁶ In this review the regulator determines whether the proposed activity has avoided impacts upon existing wetlands to the maximum extent practical. If avoidance is not deemed possible, then minimization of wetland fill must be achieved. If the regulator determines that the proposed development project is in the public interest, but that there will be unavoidable wetlands losses after all practical impact avoidance and minimization is secured, the permit applicant may be asked to provide wetlands credits as compensation for the permitted loss. The Corps regulator is instructed by a 1990 Memorandum of Understanding (MOU) between the Corps and the EPA to require that credits be similar to the wetlands types lost to the permit (“in-kind”). In addition, the MOU stresses that credits should be located as close as possible (“on-site”) to the permitted fill.⁷ Implicitly, the in-kind requirement is an effort to use wetlands type (forested, scrub, etc.) as a proxy for habitat services. The on-site requirement is recognition that the values of the hydrologic and water quality services of a wetlands area are site-dependent. For example, existing floodwater storage and water quality services accrue to the areas downstream of the wetlands permitted for fill. If the wetlands credits are located at a distance from the area filled, these services might be moved to a location that does not benefit these downstream areas.⁸ Typically, permittees contract with wetlands consultants to plan and construct such “permit-specific” mitigation, but permittees retain legal responsibility for project implementation and success.

As the regulatory programme matured, critical reviews found that NNL for all services was not being secured, and several arguments were made for increased use of “off-site” mitigation--or mitigation projects located away from the site of the permitted fill. First, in some instances, a promised credit may not have been provided at all, largely because of limited agency resources available for enforcement of the mitigation requirement. This suggested that enforcement might be more effective if multiple and spatially dispersed mitigation projects were consolidated in one more easily monitored “off site” area. Second, at times the permit recipients supplied credits when they or their consultants

5. See Appendix 7A for a discussion of the approach used for this study of wetlands credit sales.

6. Generalisations about the wetlands regulatory programme are possible, even though the administration of the regulatory programme differs among the regional offices of the Corps. Also, some states have programmes that affect the execution of the federal programme, while other states have no programmes.

7. Memorandum of Agreement between the Environmental Protection Agency and the Department of the Army Concerning the Determination of Mitigation under the Clean Water Act Section 404(b)(1) Guidelines. (February 6, 1990).

8. For an analysis of the effect of landscape location on wetlands services see Boyd and Wainger (2002).

had little knowledge of, or ability to apply, proven wetlands credit creation methods.⁹ Sometimes the resulting credits were in the form of open water ponds (a type of wetlands) that did provide hydrologic and water quality services (NRC, 2001), but did not replace lost habitat services. Also, if compensation wetlands were located near the permitted fill, the habitat service would be compromised by polluted runoff and adverse changes in hydrologic regimes from surrounding development. The wetlands would become storm water ponds that continued to provide hydrologic and water quality services, but the habitat function was lost. For habitat services to be replaced, compensation wetlands need to be located within favourable landscape settings, and these locations were likely to be at some distance from the permitted fill. Third, even when projects were initiated using sound mitigation practices, there was a multi-year time lag causing a temporal loss of habitat services between the time of the permitted fill activity and the maturing of the mitigation wetlands. It was argued that advanced investment at a consolidated, off-site mitigation area, even without certain knowledge of the location of future fill permits, could address the temporal loss problem. Finally, in some cases the rules of the regulatory programme did not require compensation credits as a permit condition for “small” fills.¹⁰ Without credits being required, NNL would be compromised. This suggested a need to have a mechanism where even fills of small areas would be required to provide compensation. In concept, if a large area of mitigation wetlands was developed, then permittees who filled small areas and could not develop credits on-site might be required to purchase credits from a consolidated site.

By the 1990s, many observers of the wetlands permitting programme agreed that there was too much on-site, permit-specific credit creation with inadequate assurance that the habitat service would be replaced. The desire to secure habitat functions, to assure that credits were available at the time fills are permitted, combined with a desire to improve regulatory oversight of wetlands credits implementation, lent support to single-user mitigation banking. A single-user mitigation bank is created when a single entity (such as a road construction agency) creates a large and readily monitored mitigation project away from the area of the permitted fill activities it serves. After an assessment confirms that the mitigation wetlands are likely to be ecologically successful, this entity is then awarded credits it can use as mitigation for future permitted fills from their own development projects. These off-site projects represent a “deposit” of credits and the deposit is drawn down as wetland fills requiring mitigation are permitted.

Moving the location of the credits to consolidated sites in landscape settings more conducive to the habitat service made sense. However, if the wetlands credits were moved very far from the fill location to favour the habitat service, then the site dependent hydrologic and water quality services would be lost. Because mitigation requirements were being defined in terms of the wetlands asset, and not separable wetlands services, there was an implied tension over which services to favour in regulatory decisions about acceptable mitigation options. In the context of single-user banks, this tension was addressed by requiring bank projects to be located within the same (usually small) watershed area where the fill permits it was developed to serve are located. These “sales areas” for other types of off-site, consolidated mitigation options (discussed below) have been established in the same way.

Clearly, the single-user mitigation bank approach to compensation can only work for permittees who expect to have a number of future projects and who had access to the initial capital to make the investment in advance credit creation. Recognising that this mitigation option is not viable for permit recipients with only one fill, some local government agencies began charging a mitigation fee, in-lieu

9. Credit creation is technically challenging, but has been shown to be feasible if proper planning and construction methods are employed (NRC, 2001).

10. At present an individual permit is required for a wide range of activities (roads, dams, etc.) for fills of multiple acres as well as small as 1/3 acre. This was not always the case. In the permitting programme’s early years areas a large as 10 acres and many activities were exempted from either permit review or from providing mitigation.

of on-site compensation, before issuing a permit.¹¹ These “in-lieu fees” (ILF) were held until accumulated funds were sufficient to undertake a single large wetlands mitigation project. ILF programmes would assure that experts in the environmental agencies would select the construction practices and locations for the credits, and then execute the projects. Also, ILF programmes were promoted as means for securing compensation for even the smallest permitted fills. However, ILF programmes were faulted for excessive lag times before mitigation projects were initiated and for not charging fees that covered the full cost of those projects. If fees were below cost, the result would be either that fewer credits than required for NNL would be created, or that public funds would be needed to complete the projects. In effect, a failure to properly account for costs was either working against NNL or, if public funds were used, there was an implicit public subsidy to permit recipients. The interest in ILF programmes waned as the prospects for private credit sales grew.

The early ILF programmes did establish a precedent of transferring the legal and financial responsibility for creating the credit from the permit applicant to another party – an ILF programme administrator – in return for a cash payment.¹² It was the affirmation by regulatory authorities that they would allow this transfer in return for a cash payment that helped motivate private investors to produce credits for sale. In turn, to encourage private sector sellers to make such investments, the federal government issued mitigation banking guidelines in 1995.¹³ These guidelines were intended to assure skeptical environmental activists that the wetlands resource and the NNL goal could be secured by relying on a profit motivated private sector. At the same time, the guidelines were expected to reduce the uncertainty that investors might face when assessing the financial viability of entering this business by specifying federal requirements for the establishment and use of mitigation banks (Scodari, Shabman and White, 1995).

Today, private sector entrepreneurs with access to private capital (borrowing or internal firm equity) have acquired land in areas away from intensive development and made investments in creating wetlands credits for sales to future permit recipients. The credits may be defined in terms of acres of a wetlands type (e.g. vegetation type and appropriate hydrologic regime), in effect assuming the pre-restoration site had little wetlands service value. In some cases a process called functional assessment seeks to create a metric that measures the change in service levels from a wetlands restoration project. It is this change, that is used to assign the credits. These sellers and their credits are subject to “certification” before they can be sold. Certification requires either that ecological success criteria are met or financial assurances (e.g. bonds) are provided.¹⁴ Monitoring and reporting of credit performance is the responsibility of the credit seller and the regulatory authority can exercise random audits and inspections of the consolidated project site. In addition, permanent easements, deed restrictions or transfers to public ownership are expected to insure that sites retain their wetland status in

11. In some cases state or local government agencies secured government funds to establish mitigation banks for general use. Largely due to the difficulty in securing public funds to capitalise banks, however, this type of public venture never matched the spread of public ventures capitalised exclusively with mitigation fee revenues. (see: Scodari, 1996).
12. Prior to the development of the credit sales programmes the Fina Oil Company developed a mitigation bank in coastal Louisiana. After the bank was developed the company determined that they had no prospective use for some of the credits they had created and proposed selling them to other companies who needed credits. At the time this was an untested concept and the effort to set regulatory rules to govern this credit sale contributed, along with the emergence of the ILF programmes, to the development of the private credit sales programme. (Environmental Law Institute, 1994)
13. “Federal guidance for the establishment, use and operation of mitigation banks; notice.” *Federal Register* 60 No. 228, p. 58605, November 28, 1995.
14. In the event of credit failure, money from the financial assurance would be used to pay for the re-development of the promised wetland credits. The amounts are set so that any bond collections would be sufficient for repairing the failed site. The use of financial assurances recognizes that the profitability of an investment in credit sales requires the opportunity to sell some portion of credit capacity prior to achievement of specified ecological success criteria. Once credit performance has been secured the bond is returned (see Scodari, 1995).

perpetuity and receive long-term management. In some cases, an endowment fund under the control of a resource agency or non-profit conservation group, with interest dedicated to perpetual management must be established.

For wetlands regulators, the credits provided by private investors promise to be successful, relative to credits provided by fill permit recipients.¹⁵ The access to private capital makes sure funds are available to initiate credit creation in advance of fill impacts. Quality control rules, combined with the serious attention to cost accounting and pricing in the private sector, were expected to result in prices for credits that reflected the full costs of wetlands credit production. Private credit sellers have the flexibility to locate mitigation sites in areas that favour long-term ecological success, and monitoring and enforcement of relatively fewer mitigation projects and responsible parties is more readily accomplished. And the ready availability of commercial credits may reduce the chance that regulators might not require compensatory mitigation for small fills where on-site mitigation or other permit-specific mitigation is not feasible or practical.

At the same time, fill permit applicants and private credit suppliers have been enthusiastic about the potential advantages of the credit sales programme. Permit applicants are relieved of the need to dedicate land at their development site to wetlands compensation credits. Wetlands consultants who help permit applicants to navigate the permit process have recognised the profit potential of adding consolidated credit supply to their business models. In fact, the number of private commercial sales ventures has expanded across the nation (Environmental Law Institute, 2002). The sellers have formed a professional business association, have their own annual conference and have association activities that promote the business (<http://www.mitigationbanking.org>).

Private credits sales were promoted as a way to address the causes of credit failure and so help to secure NNL. However, private credit sales still constitute only a fraction of the total mitigation required by regulators. Today, private credit sellers provide no more than 10 - 20% of the total credits required by fill permits in any region of the country, and in many regions the proportion is much less.¹⁶ A number of factors, many under the control of the regulatory authorities, limit private sector investment in credit creation and sales. Therefore, because almost all required mitigation continues to be provided by the permit recipients credit sales can not significantly contribute to the achieving the NNL goal. The next section discusses the current credit sales situation, and whether the gaps in securing NNL are being filled in other ways.

III. Wetlands Credit Sales in their Regulatory Context

Private credit sellers, in seeking a competitive return on invested capital, must make credit sales sufficient to recover all costs of production. Costs include cash outlays for all purchased inputs, the opportunity cost of invested funds, the opportunity cost of owned assets, land and labour, charges for management time and a return to investment risk. This listing of possible costs is not surprising to economists. However, the regulatory authorities who designed the rules governing the credit sales programme and who administer the programme often are insensitive to, or ignorant about, the full

15. This assertion that private sellers are offering higher quality credits cannot be documented by available data. There is only limited monitoring to track wetlands permitting and mitigation activities (NRC, 2001; Corps, 2001; Corps, 2002). Furthermore, there are no case studies of mitigation credit provision that compares permittee provided credits with those provided by credit sellers. What can be compared are the legal and other requirements for quality assurance imposed on permittees versus commercial credit sellers. Few of the quality assurance requirements described for the private sellers apply to permittees who do their own mitigation. On this basis most observers believe that private sector mitigation credit sales do yield greater assurance of ecological success (NRC, 2001).

16. This estimate is based on interviews with staff at some of the Corps field offices and policy analysts at the Corps of Engineers Institute for Water Resources located at Fort Belvoir Virginia.

range of costs that might be relevant to a private investor. The result is that regulatory rules appear to have raised the cost of supplying credits beyond that necessary to ensure ecological success. Meanwhile, uncertainties about the wetlands permitting programme, the continuing opportunity for permit recipients to provide mitigation on-site, and new forms of off-site mitigation make the expected future demand for credits highly uncertain.

First, consider how regulatory approval procedures affect the costs of private credit supply. An intense scrutiny of the credit certification and sales plan of the seller has raised the administrative costs of gaining approval to sell credits.¹⁷ It is not uncommon for credit sales approval to take several years and require extensive expenditures for legal counsel. Securing assurance that the restored or created areas will have the hydrologic and soil and vegetative characteristics of wetlands has led to significant delays in the approval process. Land must have been acquired or a purchase option secured. There often is an intensive regulatory review of the techniques that will be used to create the wetlands. Then performance criteria are established for mitigation wetlands. Regulators then require sellers to post performance bonds or other financial assurances for constructed wetlands for which performance criteria have not yet been fully met. In effect, in seeking to assure ecological success of the credits, regulators impose significant and redundant costs on the seller: opportunity costs of funds invested in land while approval is pending, costs for the engineering design and documentation of how the project will be developed, and opportunity costs of the funds used to post performance bonds. Meanwhile, regulatory review must establish the number of credits that can eventually be sold. If the credits are represented as acres of a wetlands type, then the credit quantity definition is rarely a matter of dispute between the prospective seller and the credit certification authority. However, where functional assessments are used there is often a protracted disagreement over how many credits will be certified. Functional assessment is process that relies heavily on the professional judgment of biologists who “score” and then place weights on hydrologic, habitat and water quality services of the wetlands. Differences of view between and among biologists in the regulatory agencies and the seller are common and how such differences are resolved can mean significant changes in the revenue potential for the seller.

Uncertainty about the future demand for credits – and the ability of credit suppliers to fulfil credit demand that does arise – also affects supply expansion. Investors won’t make an investment to produce a product if they are not sure when or if they will be able to sell that product. Difficulty in predicting future land development patterns that may affect wetlands filling -- and therefore the future demand for credits -- is always present. However, greater sources of demand uncertainty are embedded in the regulatory programme itself. First, there is uncertainty created by the ambiguity of national wetlands policy. When Congress wrote Section 404 of the Clean Water Act its intent was unclear. As a result of the ambiguity of legislative intent, there has been persistent policy disagreement over matters as basic what constitutes a wetland, what constitutes fill, what constitutes waters of the United States for defining the limits of federal jurisdiction, what constitutes an activity significant enough to warrant intensive regulatory review, and what constitutes appropriate mitigation for a permitted fill (Zinn and Copeland, 2001). In the absence of legislative clarification, the goals and structure of the permitting programme have been defined by executive orders, administrative rule making and rulings by the United States Supreme Court, as well as lower courts (Strand, 1997; NRC, 1995; NRC, 2001). If prospective private sales ventures believe that future regulatory requirements may not require credits for permitted fill, or may limit the wetlands areas or wetlands types for which credits will be required, they will discount the possibility of making credit sales in future years.

17. The 1995 mitigation banking guidelines give the EPA and the Corps, as well as the other federal and state agencies and environmental groups, access to and oversight on the process of approving a private credit seller’s business plan. Some of these agencies and some environmental groups remain skeptical of private enterprise as a reliable provider of credits (see Corps, 2001a).

Second, recall that wetlands, as a natural capital asset, may provide hydrologic, water quality and habitat services depending on the wetlands type and its location. Also recall that the values from the hydrologic and water quality services of existing wetlands are site-dependent; therefore, credits to compensate for these services should be near the permitted fill. Habitat services are less site-dependent and in fact might be enhanced if a compensation wetlands was moved away from a developing area (for example, to adjoin a nature preserve). Requiring that the credits be at or near the location of the permitted fill best replaces hydrologic and water quality services, but habitat services may best be replaced at more distant locations. Because the wetlands permitting programme has been organised around the wetlands asset, and not the three services, there has had to be a compromise on location of the wetlands credits. The compromise has been to limit the area in which wetlands credits can be sold; this is termed the “sales area”. Sales areas are currently limited to spatially small watersheds with the effects of both restricting the potential demand for credits for a private seller and limiting the ability of sellers to compete for sales across sales area boundaries.

Third, the sequencing process that governs the fill permitting process creates credit demand uncertainty. Recall that the sequencing process first requires permit applicants to avoid and minimise wetland impacts as conditions for permit issuance. Then if a permit is issued, the regulator will first examine and give regulatory preference to available opportunities for the permittee to provide on-site mitigation for permitted wetland impacts (even though, as noted earlier, regulators have been increasingly concerned about the quality of on-site mitigation). However, there is no standard analytical protocol that is followed in applying the sequencing steps, leaving the specific regulator handling any particular permit application wide latitude in the process. Thus, the regulator’s judgment determines whether the proposed activity is water-dependent, whether the wetlands can be avoided and whether on site mitigation is practical, and such judgments can be highly variable among regulators. Then, if the regulator does allow the permittee to seek credits off-site, the kind and location of wetlands credits that will be required is determined for that permit and may not be at the location or be for the type of wetlands that have been created by a credit seller. So even if a permittee wants to use a specific credit seller (having come to agreement with a seller on credit price), there is no guarantee that the regulator will decide that the proposed credit trade can be used to satisfy the permittee’s mitigation requirement.

Despite the high regulatory approval costs and credit demand uncertainty, some investment in credit creation by private sales firms has occurred in some areas since the mid-1990s. How were the significant market entry costs and demand uncertainty risks managed by these market entrants? Some evidence points toward a plausible risk management process (Shabman, Stephenson and Scodari, 1998.). First, before seeking approval as a credit seller, prospective sellers identify prospective applicants for fill permits at specific sites. A prospective permit applicant and credit seller might then reach an informal understanding with the regulatory agency, or conclude based on previous regulatory decisions, that the permit applicant will be allowed to meet its compensation requirements by purchasing credits from that particular seller, if the seller’s credits are certified. Included in this understanding is the kind of credits that will be required to compensate for wetland losses at the fill site and the acceptable location for credits – the sales area.¹⁸ Second, regulatory approval of a credit seller typically allows a limited share of

18. In fact it may be more accurate to say that these understandings are a result of the tacit knowledge of credit sellers about the demand for fill permits and the likely compensation requirements of the regulators. Credit sellers are closely linked to the land development sector, and in fact may have business interests in land development and have been through a permit application process. This is how the seller can anticipate a demand for wetlands credits and be expert in navigating the often-confusing regulatory process both in securing permits for fill and in getting their credits approved for sale. Meanwhile, as the credits sales programmes have been more common there has been increased communication among sellers and regulators, such as at the sellers annual meeting. Presentations at these conferences, as well as informal discussions, allow credit sellers to infer what will be required to secure approval as a credit seller. The credit price charged is a return to this tacit knowledge as well as for the land, labor, energy, materials and know-how used for credit creation.

the created credits to be sold before mitigation wetlands have been certified successful in accordance with performance criteria, in return for the posting of a financial assurance. With these two considerations in mind, the credit seller and the permit applicant could negotiate a credit price that was high enough for the seller to recover a significant share of its costs for the whole credit venture, even if only a fraction of the credits created (those required by the pending fill permit) are eventually sold. Because there were few if any competing sellers, the credit prices paid were based on the permit applicants avoided cost from buying credits, rather than different sellers' cost of production (including a competitive return on invested capital). The fill permit applicant's avoided cost includes avoided expenditures for creating and maintaining on-site credits, avoided opportunity costs of dedicating part of the development site to mitigation, and avoided legal or administrative costs for securing the fill permit (because the seller typically includes the cost of this permitting assistance as a part of the credit price).

This market entry and credit pricing strategy served all concerned parties. Permit applicants who purchased credits benefited for the reasons noted above. The intense review required for regulatory approval of credit sellers, combined with requirements for financial assurances and long term site protection and maintenance, provided assurance to the regulatory agency that the created credits will be ecologically successful. Meanwhile, the private seller earns an acceptable return on its entire investment in the credit venture from the immediate sale of just a portion of its credit supply. Hence, past and future costs were covered and the potential problem of future demand uncertainty was effectively addressed (Shabman, Stephenson and Scodari, 1998). Investment in credit production in anticipation of making future sales is a high-cost with high-risk business, and the wetlands regulatory programme creates the risk. The market entry and credit pricing strategy adopted by credit sellers minimises the financial risk of uncertain future credit sales. Note that this strategy requires entrepreneurial and business management skills as much as wetlands restoration expertise. As a result, the wetlands credit sales business attracted sellers with certain characteristics conducive to participating in this market. The sellers have access to capital markets for borrowing and often already own tracts of land in locations and in a condition suited to wetlands credit creation. Credit sellers are closely linked to the land development sector, and in fact may have business interests in land development; this is how the seller can anticipate a demand for wetlands credits. Sellers are expert in navigating the often-confusing regulatory process both in securing permits for fill and in getting their credits approved for sale. The credit price charged is a return to this management skill as well as for the land, labour, energy, materials and know-how used for credit creation.

Nevertheless, the regulatory conditions embedded in the wetlands permitting programme have limited widespread market entry throughout the country, thus limiting competition. There are many areas where no, or only a few, private sellers are present and the credits they have been certified to sell are limited to a small sales area. Moreover, in places where private sellers are present, they have to set credit prices to recover not only the costs of credit production, but also the regulatory costs of gaining credit sales approval and the risk costs associated with future demand uncertainty. As a result, credit prices may exceed what many permit applicants are able to pay for mitigation.

It is essential to stress here that, as alluded to above, the regulatory agency (the Corps or a state programme) that issues a fill permit determines the acceptability of a permittee's mitigation proposal. There is no opportunity for a credit sale to be made between a buyer and seller without this regulatory approval. More importantly the regulatory agency may approve other compensation forms that are alternatives to the purchase of private credits. In fact, given high prices and limited availability of private seller credits, permit-specific on-site credits remain a necessary option for permittees and the regulator. Hence, the problems of compensatory mitigation failure from permittee-sponsored mitigation, and the failure to require compensation for wetland services lost by some fills, remain concerns for the permitting programme (NRC, 2001). As a result, a continuing interest in fostering off-site mitigation, in the face of the limited quantity and high price of credits from private sellers,

reinvigorated interest in in-lieu fee programmes in the late 1990's (Environmental Law Institute, 2002; Scodari and Shabman, 2000). Recall that in an ILF programme, after the sequencing review, the permit recipient pays a fee to an ILF administrator (typically a conservation organisation or government resource agency designated by the Corps to receive and use fees) instead of creating their own credits. The collected fees are accumulated until used by the ILF administrator for a wetlands mitigation project. However, various reviews (Scodari and Shabman, 2000; GAO, 2001) have concluded that ILF programmes may not provide the same quality assurances as the private sector, in part because the fees may not be adequate to cover the costs of credit production. Also, ILF programmes result in a time lag between the fill activity and the mitigation being initiated. Meanwhile, other forms of securing off-site compensation also have been approved as compensation with increasing frequency. Cash donations to ongoing restoration projects by non-governmental organisations and government agencies have been accepted as mitigation. Also, permittees have been allowed to conduct mitigation off-site on land owned by such organisations. Many of the same criticisms of ILF programmes apply to these options.

From the perspective of private credit sellers, on-site credit creation, ILF programmes and cash donation programmes provide inferior wetlands mitigation and unfair competition. They point out that these mitigation options are held to lower quality control standards do not bear the costs of credit sales certification and credit demand uncertainty, may not employ accurate cost accounting, and may not reflect some costs (ex. donated labour and land) of production in their prices. The result has been political activity by the private sellers seeking legislation that would require regulators to give preference to the use of available private credit supply for compensatory mitigation, thus limiting the discretion of regulators to determine appropriate mitigation for permit fills. Credit sellers have already encouraged regulators to issue guidance to agency field offices that impose stricter requirements on ILF programmes and that establish a hierarchy of mitigation options that favours private credit suppliers over ILF mitigation.¹⁹ They have also lobbied for on-going efforts to raise the quality assurance requirements and enforcement activity applied to on-site mitigation projects.²⁰

IV. Toward a Market-like Credits Sales Programme

Based on the experiences with both private credits sales and ILF programmes, a new programme could be designed that will provide more affordable and high quality credits, initiated in advance of fill permits, in many more watersheds. Three basic changes in the approach to wetlands mitigation would form the foundation for this new programme. First, a new programme would require a designated entity responsible for wetlands mitigation (analogous to the ILF administrator) that would be given the authorities and initial funding to act as a both a credit purchaser and re-seller of the purchased credits. If properly designed and administered this programme will secure the supply, quality and price advantages of competitive wetlands credit market (numerous sellers competing for the business of permit recipients).

The core conceptual idea is taken from the economics literature for managing natural monopoly and is termed "competition for the field" (Demsetz, 1968). In the general case of a natural monopoly, a government body holds a competition where firms compete for the exclusive right to offer a product in a specified area. The winner of the competition would sell the product directly to the consumers and would be responsible for bearing and managing any demand risk. Presumably, they would include recognition of this risk in their bid to provide the service. For example, if a highway were to be built, firms would compete for the right to build the highway and then charge tolls that would recover costs. The key to this programme is that the government would assure the winning firm that

19. "Federal guidance on the use of in-lieu-fee arrangements for compensatory mitigation under section 404 of the Clean Water Act and section 10 of the Rivers and Harbors Act; notice", *Federal Register* 65 No. 216, p. 66914, November 7, 2000.

20. See: <http://www.mitigationactionplan.gov>

no other competing roads would be built in the area for a specified period of time (Engel et.al, 2002)

In the case of wetlands mitigation, the “field” is the sales area and the product to be sold is wetlands credits. However, for this application there must be an accommodation of the demand uncertainty created by the wetlands regulatory structure. Specifically, the mitigation agency would purchase the credits produced by the winning bidder(s) on a defined payment schedule. This has the effect of shifting the credit demand risk from the private to public sector, and in so doing encourages credit suppliers to compete to provide a product with an assurance that there will be a demand for the product.²¹ By removing the risk from the private sector and by fostering competition among prospective suppliers, and by eliminating the costs of gaining regulatory approval associated with the existing credit sales programme, credit prices should move toward costs of production.²²

The process would work in the following way. The wetlands mitigation agency would make an estimate (for say 10 years) of the number and type of credits they anticipate being required to meet the no net loss goal in a defined sales area. With the estimate, the agency issues a Request for Proposals (RFP) from potential suppliers of credits. The low-cost bidder who also provides ecological success assurances such as those now imposed on private sellers would receive the credit supply contract.²³ The price paid for credits by the mitigation agency would reflect the full cost of producing and then maintaining ecologically successful credits. The RFP would include a requirement for long-term site protection and maintenance.²⁴ The winning bidder immediately begins the mitigation project and payments would be made on a defined schedule tied to project construction milestones and the attainment of performance criteria (and not, as under the current programme, to prospective and uncertain future credit sales to individual permit applicants). If the watershed authority overestimates its demand the result will be that public funds have been used to buy credits that would in any event contribute to wetlands restoration. If the demand is underestimated, the agency can immediately issue a new RFP.²⁵

For this system to work, the entity issuing the RFP must have access to a fund that can be drawn on to pay the winning bidders on the agreed-to schedule. In turn the agency would re-sell the acquired credits to permit recipients at a price that recovers its credit acquisition costs and repay the fund until another round of bidding is required. The price charged for credits by the mitigation agency might be thought of as the in-lieu fee, but this fee is based on real projects and real costs elicited from the bidding process.

Second, the possibility for bid programme success will be enhanced if compensation requirements are established for wetlands services and not for the wetlands asset itself. Then, the RFPs could be issued for the wetlands habitat service, separate from the other services. Earlier it was recognised that there were sound arguments for limiting the sales area for selling hydrologic and

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21. Recall that the public creates investment risk by its uncertain approval process for sales ventures and by the continuing ambiguities in the wetlands permit programme.
 22. Simulations suggest prices might fall as much as 75% in some places (Shabman, Stephenson and Scodari, 1998).
 23. Preferably, the quality assurances would be based posting of bonds that are held until performance goals for the mitigation wetlands (e.g. attainment of a certain hydrologic regime) are achieved.
 24. The low-cost bidder might be a government agency or an NGO. However, the NGO would need to do planning and initiate credit creation work up-front and would need to have the same quality assurances as all other bidders. If the NGO does not charge for all costs, for example interest on capital or on volunteer labor, it would have a competitive advantage. Unless NGOs are prohibited from being bidders this advantage can not be avoided, but the implicit subsidy to wetlands filling would be present.
 25. A similar competitive bidding process has been used in the Colorado River Salinity Control Program of the US Bureau Of Reclamation. The Bureau publishes RFPs for salinity reductions. The winning bids are selected based on lowest unit cost and assurances that the salinity reductions will be realised, including the bidder's willingness to post a performance bond (NAPA, 2000).

water quality services of wetlands. It was also the case that a larger sales area might be best for sale of habitat services. However, it has been the more limited area for hydrologic and water quality services that has been the basis for the sales area limits currently placed on private credit sellers. In turn, the limited sales area has created two problems that could also plague a bidding programme. One problem has been a thin market; in effect there is often only one seller, if any, in almost all sales areas. A second problem has been that the limited sales area also limits the land parcels suited for a wetlands mitigation project that can provide all the wetlands services. Landowners have recognised the unique asset they hold and have demanded land prices that have resulted in escalating credit prices. One solution would be to expand the sales area so that more land parcels would be suitable mitigation sites.

The need to expand sales areas suggests a third programme change to secure mitigation for each service through different regulatory programmes. The habitat service is the one that must be replaced by another wetlands, but is also the service that can be mitigated farthest from the site of the permitted fill. If the bidding and resale programme was applied for wetlands as habitat, the sales area could be greatly expanded and available land for mitigation projects expanded as well. The hydrologic and water quality services are more site-dependent, but can be provided in ways other than by wetlands. Site design changes, storm water ponds, pervious pavement, riparian buffers and a host of other methods can be substitutes for the water quality and hydrologic services of a wetlands (Center for Watershed Protection, 2000) and can be implemented near the site of the permitted fill. Mitigation requirements for lost hydrologic or water quality services might be met by one or more of these non-wetlands alternatives. Significantly, there are a variety of local and state regulatory programmes governing land development that already require mitigation for hydrologic and water quality effects of land development.

V. Conclusion

The goal of No Net Loss of wetlands due to fill permitting is being compromised when a permit recipient provides low quality credits to offset the loss of wetlands services. A wetlands credit sales programme, which has been understood as an example of a market-like approach to environmental management, has been instituted to address this problem. The programme has had success in raising the ecological quality of credits and in initiating credit production before fill is placed in wetlands. However, barriers to widespread market entry have limited investment in credit creation and, where credits are available, high credit prices have discouraged regulators from requiring their purchase. As a result, most credits continue to be provided by the permit recipients.

Wetlands mitigation authorities, appropriately funded and administered, could still rely on the private sector for supplying quality credits. These private sellers would compete with each other to provide credits that would be purchased by the mitigation agency and then re-sold to permit recipients at prices sufficient to recover costs of credit procurement. In this way, the benefits of competition can be secured even when there are few buyers and sellers of credits. In addition, the areas where quality credits would be available would be expanded and the costs of credits should fall from current levels making it practical to require mitigation for all permitted fills. For this programme to be effective it would be desirable to define mitigation requirements and credits in terms of the services provided by wetlands and not in terms of wetlands area and type.

APPENDIX 7A: THE APPROACH TO PROGRAMME EVALUATION

Pattern modelling (Wilber and Harrison, 1978) is a structured, but not rule-based, approach to understanding a complex system. The product of pattern modeling is, unapologetically, a narrative (a story) drawing from case examples, interviews, and available data and existing literature.²⁶ The construction of a narrative begins with a conceptual model of the system of interest. The narrative is developed over time and at several stages all or part of the story is tested for validity through critical reviews by participants described in the narrative and, if possible, by data analysis and simulation modeling. Of special note is that pattern modeling puts a premium on incorporating details of the legal and organisational setting and minimises the use of simplified assumptions about behavioural motivations of organisations and individuals.

The story of the U.S. wetlands credit market in this paper has been written and re-written by the author over the past decade. During that time I have, at different times, been an observer/analyst of the wetlands programme and at other times have participated in the design of the rules governing the programme. For this reason the paper is “lightly referenced” because many of my observations are impressions from over 300 interviews, attendance at numerous conferences and Congressional hearings, and reading legal briefs, agency memos and newspaper articles.

Several years ago an initial conceptual model was developed of a then nascent proposal for a market for the buying and selling of wetlands credits. Hypotheses about market and policy determinants of the demand and supply for credits were developed. After considering these determinants it was concluded private investors would enter such markets to sell high quality mitigation credits in watersheds across the nation (Shabman, Scodari and King 1994). The written products from the study were not meant for a professional economics audience; instead the goal was to communicate the necessary conditions for a market approach to regulatory staffs that were educated as biologists and lawyers (Shabman, Scodari and King, 1996). The resulting narrative made an instrumental contribution to the design of the guidelines that were issued in 1995 governing the initial operation of this market. It was the story, and not an abstract model, that was essential to this successful policy entrepreneurship (Majone, 1989).

Over time, as the anticipated results from wetlands credit markets did not materialise, I had to again consider the interplay between U.S. wetlands law and its implementing rules, regulator decision-making to implement the law, and private entrepreneurs decisions to invest in wetlands credit production. A revised story needed to be constructed to explain the existing situation and to suggest ways to expand the quantity and quality of wetlands credits. The reliance on interviews, testimony and other written materials was again important to this endeavour. It is especially worth noting that valuable insights were gained from my participation in meetings about the wetlands credit market with wetlands regulators, fill permit recipients, private credit sellers and environmental groups. During these meetings the critique, by those closest to the wetlands programme, of my

26. McCloskey (1994) suggests that story telling has an unrecognized, but central significance, in the economics profession.

assumptions and causal arguments made for a more complete understanding of the current and prospective future state of the wetlands credit market. In addition, because there was now some experience with credit sales, it was possible to build an investment simulation model to test the validity of the arguments about the factors that were discouraging robust credit markets (Shabman, Stephenson and Scodari, 1998). It was from these activities that the reform proposal described in this paper was developed. The basic concept was presented to and included in a report of National Research Council Committee (NRC, 2001) and was endorsed in that report. At the same time wetlands programme administrators in North Carolina embraced the concept and they began to experiment with a credit purchase and resale programme. My recent activity has been to advise those programme administrators, while continuing to interview private credit sellers and other participants in the North Carolina wetlands programme. Lessons are currently being learned from this experiment about how to write a request for proposals and about the need to mitigate for wetlands services instead of types of wetlands acres.

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Chapter 8

The Management of Ex Post Public Policy Evaluations

by

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I. Introduction

The previous chapters have revealed that the ‘management’ of policy evaluation extends well beyond the specific technical tasks associated with undertaking the evaluation exercise. Indeed, for evaluation to be effective it must be integrated in the public decision-making process more generally. As such, in this concluding chapter the focus extends beyond the evaluation process itself, to a broader discussion of how public policy evaluations fit into public administration more generally. An effort is made to link some of these issues to the case of the ex post evaluation of tradeable permits. It draws upon the case studies in the preceding chapters, as well as the aforementioned ‘policymakers’ roundtable’, and a review of the secondary literature.

II. Policy Evaluation as Performance Management

It is important not to see ex post policy evaluations as a technical exercise which is a substitute for political decision-making (CEC 2002). The role of policy evaluation is both narrower and broader than this would imply. It is narrower in the sense that evaluations are only one input into the decision-making process, and it is broader in the sense that evaluations should be seen to encompass more than just a technical exercise.

It has been argued that undertaking evaluations has the following three broad benefits for public authorities (OECD 1999b, CEC DG Budget 1997):

- Improving the effectiveness and economic efficiency of those public policies which are already in place;

1. The views expressed in this chapter are those of the authors, and not necessarily those of the OECD or its Member countries.

- Discouraging the introduction of future policies which are likely to be ineffective and inefficient; and,
- Increasing the accountability and transparency of public decision-making, including communication with stakeholders.²

Thus, in addition to being an efficient tool of decision-making, systematic policy evaluation is also an important element of performance management within public administration (see OECD 1999b). By encouraging policy-makers to see their role as defending the broad public interest in a manner that is subject to a transparent assessment, the systematic use of policy evaluations can result in a culture shift within public administration. Indeed, the Australian Government has argued that the process of evaluation in general can be of more far-reaching benefit than any precise estimate that arises out of a particular evaluation (see Holmes and Argy 1997).

However, incremental and instrumental uses of evaluations need to be distinguished. In some cases the results of evaluations may translate directly into policy recommendations and, hopefully policy reforms. Such cases are likely to be exceptional. Instead, the ‘culture’ of evaluation may make incremental processes of reform more likely (see Boyle 1993). Changes in the direction of public policy arise out of a continuous process of collection and dissemination of evidence of the desirability of change.

III. Timing of Evaluations to Encourage Policy Reform

In order to have a policy impact, the evaluation procedure can not be divorced from decision-making processes more generally (Deighton-Smith 1997). In temporal terms this means that in order to have an impact on policies, evaluations need to be linked with the policy cycle. In the absence of any window of opportunity for reform for bureaucratic or political reasons, evaluations – no matter how revealing - are unlikely to have significant influence on policy development (OECD 1999b).

In some cases, it is felt necessary to pre-programme the schedule for evaluations. For instance, in France the “Law on Interior Transport” requires that large publicly-funded infrastructure projects must be evaluated at least 5 years after they commence operations (Delache 2003). In Japan, it is recommended that evaluations are undertaken between five and ten years after the introduction of a policy (Japan MPHPT 1997, Article 7). In the United States, U.S.C. Section 7612 requires the EPA Administrator to conduct a comprehensive analysis of costs and benefits of all major provisions of the Clean Air Act (including, but not limited to, Title IV which covers the SO₂ allowance programme). An updated report has to be submitted to Congress every 24 months. Some programmes have specific pre-specified conditions that trigger an evaluation. RECLAIM, for example, required an evaluation of the programme if and when the permit price hit a particular trigger. When the condition was met, the evaluation was conducted and reform actions undertaken (Harrison 2004).

It has even been proposed to include ‘sunset clauses’ with the continuation of policies – particularly those with important implications for public expenditures – dependent upon the outcome of evaluations (see OECD 1997d). More generally, linking evaluations with the budget process can further encourage impacts on policy development (OECD 1999b). For instance, in Australia a ten-year reform programme has been put in place under the National Competition

2. The OECD’s Public Management Service has also argued that it can be an effective means of improving co-ordination across public sector departments (OECD 1997).

Policy with three formal reviews. To access these funds (billions of dollars), State and Territory governments have to meet performance targets relating to aspects such as pricing, separation of regulation from service delivery, and the development of competitive markets (Young 2004).

Some countries have gone even further, not leaving implementation of evaluation recommendations to chance. For instance, in Japan the Ministry of Public Management has a legislative responsibility to ensure that ministries and agencies 'act upon' the results of evaluations undertaken. Thus, if an evaluation is undertaken which indicates significant flaws and efforts are not made to rectify these faults the MPHPT has a responsibility to ensure that policy changes are introduced (Japan MPHPT 2002, Article 17).

In the case of tradeable permit programmes, many workshop participants emphasised that most of these programmes evolve considerably over their lifetime. Apparently both participants and administrators experience a considerable amount of "learning by doing". Thus in many cases it is important to allow potential for adjustments in programme choices. However, the necessity to provide flexibility for evolution can conflict with the need to make the programme rules sufficiently stable that investments in pollution abatement and resource exploitation, etc. can be justified. The effects of sunk costs (both public and private) need to be considered. Thus, the trade-off between the need for discretionary power to make adjustments, and the benefits of stability of a system creates a tension that needs to be resolved on a programme-by-programme basis.

One solution is to assure that the initial programme is not so laden down with restrictions that it can not effectively evolve. While the evidence seems to suggest that most restrictions diminish over time as familiarity increases, as the wetlands credit trading example suggests (Shabman 2004), when the initial design deviates so far from an effective market, the transition may be very difficult and inefficient. Indeed, sunk costs associated with the introduction of the programme may make reform more costly than adopting a new policy altogether.

Thus, relatively early targeted evaluations can prove to be very useful. Many countries introduce tradeable permit schemes for which they have specific concerns *a priori*. No doubt the Danish administrators were worried about market power in their CO₂ trading programme (since they only had eight firms) and British administrators were concerned about environmental effectiveness (since they had some permits denominated in relative terms). As such, with limited resources countries should focus their mid-course efforts on the elements of greatest concern.

In Australia, legislation mandating an assessment of the water entitlements trading programme at the end of two years was a clear indication that interstate water trading was a new area for States to enter into, and that regulatory authorities were concerned about its effects (Young 2004). In particular, since each State defined water entitlements in a different manner, they were concerned that the system could result in serious problems. Use of this cautionary approach enabled the States and the Commission to support the process without having to make a long-term commitment. The Australian national water reform process committed States to progress towards development of an open market and this mechanism enabled them to explore the implications of moving to a Basin-wide market.

The value of periodic evaluations is clear, particularly since mid-course corrections seem more common in the area of tradeable permits. For instance, the nature of the harvesting right was changed in the New Zealand fishery (Kerr 2004), banking was added to the lead banking programme (Nussbaum 1992) and, an additional 10% reduction in swine was added to the Dutch nutrient quota system (Wossink 2004). Since evolution is so common for tradeable permit programmes, a strong case can be made for thinking of mid-course corrections as routine.

Satisfying the special information requirements of the mid-course requirements implies the need for necessarily more limited, but more focused, evaluations at a relatively early stage in programme development.

IV. Ensuring Adequate Data Collection

Ex post evaluation depends crucially on built-in mechanisms for data collection (OECD 1997a). Data collection should be closely linked with the implementation of the programme. Indeed, there should be an 'in-built system' in place which ensures that data collection necessary for ex post evaluation is defined from the outset (see OECD 1997a). Ideally, the monitoring system should be the first source of information for the ex post evaluation of public policies (CEC DG Budget 1997). In addition, there should be strong links with the data used in ex ante evaluations. In effect, data necessary for ex post evaluation should be identified at the point at which the policy is initially appraised, and certainly prior to introduction of the policy (HM Treasury 2003).

Tradeable permit programmes present specific issues for data collection requirements. For instance, for programmes where permits are distributed gratis (rather than auctioned off), both the identification of the eligible participants and the basis for their allocation (typically based upon historic use) must be established before the initial allocation can be made. Monitoring and enforcement of the operating programmes requires the routine collection of data on transfers and actual emissions or use. Other important data, such as prices, may not be routinely collected, but may be easily obtainable from other sources such as brokers.³

Three of the case studies provide rich examples of how very useful data became available once the system was up and running. New Zealand's ITQ programme (Kerr 2004)⁴, RECLAIM (Harrison 2004) and the SO₂ allowance programme (Ellerman 2004) all report on rich data sets that were able to be assembled from data gathered by the normal administration of the program.⁵ In such cases the costs of performing an evaluation, relative to other instruments where dedicated data collection exercises are required, are likely to be much less.

However, other important data may only be become available as a result of special data-collection efforts. Examples in this category could include surveys of participants to discover the sources of transactions costs or motivations for trade. For full benefit/cost analysis it could also include the monetised environmental benefits or damages. Finally some unobserved data can be

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3. Prices were routinely collected in the New Zealand ITQ system, but not in the SO₂ allowance programme. In the SO₂ allowance programme, for example, trades can either take place at the annual auctions conducted by the Chicago Board of Trade or privately. While the former prices are routinely made available, the firms who broker the private trades have made aggregate price data available for those trades as well.
 4. For example, Kerr (2004) reports that her group was able to assemble a considerable amount of information on the New Zealand ITQ system over the period 1986–1999. Their data set includes information on the name of each fish stock, quota transactions (i.e., prices and quantities of quota leases and sales), the export prices of fish species covered by the ITQ system, quota ownership, the total allowable commercial catch (TAC) and actual catch for each fish stock, biological information on fish species, climatic variation, and interest rates.
 5. A fourth case (Pedersen 2003) had the opposite experience. Denmark had 10 years of good data available before the implementation of their programme and so it has not felt the need to move to continuous emission monitoring.

inferred from observed data. This would include marginal control costs for pollution abatement, which can be inferred from the prices revealed by transfers.⁶

Special features of a tradeable permit programme may also affect the quality of the data. Ellerman (2004) and Harrison (2004) both point out that continuous emission monitoring was required of all (the SO₂ allowance program) and some (the RECLAIM programme) of the sources. Though some initial problems were encountered in the case of RECLAIM, the current data seem highly reliable, although the means by which it is disaggregated temporally and spatially does complicate undertaking some kinds of evaluations.

An unexpected, but occasionally significant, quality impact may come from improvement in the emission inventories used in programmes which are voluntary, or baseline-and-credit schemes. Since participation in the programme can confer some significant economic benefits, sources want to validate their inventories in order to participate. This was cited as one of the major benefits of the introduction of a programme to control particulates in Santiago, Chile (Montero et al., 2002).

While the data specifically needed for administration of a tradeable permit programme (i.e. transactions) are likely to be routinely gathered, the same cannot be said for all data. McLean (2003) points out that the collection of data deemed not to be essential for running the programme – but which are likely to be useful for evaluation – are frequently the first thing cut when budgets are tightened. Inconsistent collection over time could leave important holes in any time-series data sets. How serious a problem is posed by missing data will depend upon the elements to be evaluated, and the means of evaluation.

V. Providing Access to Data for Evaluators

Access to data is also important (Smith 2002). This is one area in which there are significant variations between countries. It would appear that it is only in one OECD country (the United States) where it is possible to undertake comprehensive evaluations of policies without requesting access to official data. Indeed much of the data necessary to undertake cost-effectiveness studies of a wide variety of policies are available on different government websites.⁷ In other countries, evaluators will be required to request access to official data. This is one of the reasons why internal evaluators are preferred in some countries (see below).

Thus, the fact that programme administrators may gather data, of course, does not mean that it is always made available to independent researchers or even other arms of the government responsible for evaluation. The case studies themselves provide considerable support for the view that independent evaluations are an important component of ex post evaluation and therefore it is important to provide data access if outside objective reviewers are to contribute.

6. Theory suggests that permit prices will be equal to marginal abatement costs. In markets with high transactions costs and few trades this will be a very unreliable indicator.

7. Indeed, due to the availability of data, many CBA's and CEA's have been undertaken in the United States without the regulatory authority even being aware of their existence. Such 'openness' can bring incalculable benefits in terms of improved policy design if policymakers incorporate their results in their overall assessments of a policy.

For emissions data, the SO₂ allowance programme represents the 'gold standard'. No other programme provides the amount and quality of data available to the general public.⁸ The programme also provides researchers with a high degree of access to other data such as basic heat input and permit transactions. Ellerman (2004) reports that his MIT team had no difficulty obtaining any of the data necessary for his team to do their evaluations from the Environmental Protection Agency, the Department of Energy, and the Department of Commerce web sites.

Granting access to such data without requiring permission of the authorities seems to be exceptional, but very productive. Discussions at the workshop suggested that this proclivity for providing very transparent systems may be a main reason why American environmental policies (CAFE, Green Lights, SO₂ allowances, etc.) are so much more thoroughly evaluated. The EPA can benefit (and has benefited) from these independent ex post evaluations despite having little control over them. The costs savings can be considerable. Indeed, a simple review of the academic literature may give regulatory authorities the basis they need upon which to evaluate at least some aspects of environmental policies without commissioning the work themselves.

Other programmes have made an impressive amount of data available to at least some independent researchers with mutually beneficial results. The New Zealand experience is instructive (Kerr 2004). Their analysis was done in two stages. The two cooperating research groups funded the first stage out of internal funds. During this phase they negotiated a confidentiality agreement with the Ministry of Fisheries that not only provided access to some of their confidential data, but also made available a considerable amount of indirect support through access to staff. In the second stage their work has become partially funded by the Ministry of Fisheries. This closer relationship has reportedly further facilitated data access and the researchers no longer pay even nominal sums for data.

According to Kerr (2004) the fact that the Ministry of Fisheries did not initially fund their effort made it easier for their research team to gain the confidence of both industry actors and the environmental community. Thus, as noted above, sources of funding are important irrespective of the institutional location of the evaluator. Moreover, the New Zealand experience also supports the view that if evaluations are to be credible they must not only be independent, but they must be seen to be independent. And finally, since the policy has been widely perceived to be a success and most of the players involved are quite proud of it, administrators were very supportive of research that could rigorously show its value.

A third example is provided by Wossink's (2000) study of the effect of policy uncertainty in the Dutch nutrient quota program. Some of the data used in that analysis were not publicly available, but were obtainable for this independent study from both public and private sources. Since this increases the cost to the researcher, the probability of evaluations being undertaken more widely are correspondingly less.

Sharing information about basic data facilitates independent ex post evaluation, particularly if rather detailed, disaggregated information is made available. However, making public such detailed information may compromise legitimate commercial concerns raised by market participants. One response to this is to aggregate the data in such a way that commercially sensitive information can not be traced back to individual participants. This protection comes at a cost, however, since aggregating data removes a great deal of variation that might prove analytically important in understanding the actual workings of the system.

8. McLean (2003) points out that the public can now have real-time on-line access to CEMS data from any plant in the SO₂ allowance programme. This provides a degree of public accountability that seems very conducive to good policy.

Retaining the basic data within the government and not making it available to the public would, of course, also protect privacy concerns. Although that would preclude completely independent evaluations (such as those performed on the SO₂ allowance programme), it would not prohibit independent evaluations that were approved by the government and therefore entitled to have confidential access to the data (such as those performed in New Zealand and Australia).

In general, it appears that data accessibility could be improved for most of the programmes. Although the confidentiality of proprietary information is a legitimate concern, this should not be a binding constraint for most programmes. The Danish programme, which involves only eight participants in the market, illustrates the type of very small programme that does face the trade-off between accessibility and confidentiality (Pedersen 2003).

VI. Selecting the Institutional Location of Evaluators

The central issue associated with the choice of evaluator is the decision about whether or not to choose an 'internal' or an 'external' evaluator. An internal evaluator is usually someone who is directly within the agency or ministry which is responsible for implementing the regulation, while an external regulator is usually someone who has been contracted (e.g. an academic or a management consultant) to undertake the evaluation. However, the distinction can be one of degrees and not kind. For instance, in many cases there may be a separate evaluation unit within the Ministry which is responsible for the policy to be evaluated (OECD 1999b and OECD 1997a). In ascending order of the degree to which the evaluator is described as being 'internal' to the authority whose regulation is being evaluated, the following possible classifications can be considered:

- organisation evaluated
- special evaluation unit within ministries
- central management agencies
- commissions
- management consultants
- academic or research institutions
- community and consumer groups
- national audit offices

However, institutional location is not the only determinant of the degree of 'autonomy' enjoyed by the evaluator. For instance, in the OECD Public Management Service's classification set out above national audit offices are presumably considered to be even more independent than consultants or academics since no contractual relationship is involved in the former case. However, this will depend upon the political culture within the country. No one-to-one relationship exists between the institutional location of the evaluator and the degree of autonomy which the evaluator enjoys.

There are advantages and disadvantages between choosing internal or external evaluators (OECD 1997a and Smith 2002). The use of internal evaluators runs the risk of

significant bias, sometimes not even recognised by the officials themselves. If government officials have a vested professional interest in the outcome of the evaluation (i.e. continuation of the policy), they are less likely to be critical. One economist who oversaw a review of the US Clean Air Act stated that she had to “strong-arm EPA staff to get them to conduct a sensitivity analysis that generated lower benefit estimates” (Lutter 2003). More subtly, they may have a feeling of ‘ownership’ over the project which makes it difficult for them to distance themselves from the policy in a way which allows for independent assessment (Lutter 2003, CEC DG Budget 1997). Moreover, they may not have the specialised technical capacity needed to undertake methodologically complicated evaluations. For these reasons, external evaluators are often recommended.

On the other hand, external reviewers may be insufficiently familiar with many of the programmatic details of the policy to undertake an effective assessment, perhaps underestimating some of the costs associated with policy implementation. In addition, if data confidentiality is important they may not have access to all of the data required in order to undertake an evaluation of the programme. And finally, the use of external evaluators may restrict the potential for ‘learning by doing’ and downstream policy reform within the regulatory authority.

However, clear advantages are associated with the use of external evaluators. In the United States for example, the external ex post evaluations of the SO₂ allowance programme published by Resources for the Future and MIT have served the public, the academic community and the programme managers well. Since they approached the evaluations from different perspectives each review was helpful in its own right. These independent external evaluations were made possible by the public availability of all the key data about the programme.

These trade-offs are ‘reconciled’ in strikingly different ways in OECD Member countries. For instance, Japan stresses the value of internal understanding of the policy process, greater potential for policy reform, and ‘learning-by-doing’ in supporting the use of internal evaluators (Japan MPHPT 2002). While the European Commission generally recognises the importance of such factors, a stronger preference for the use of external evaluators is evident (CEC DG for Budgets 1997). In France, it is suggested that a combination of internal and external evaluators may be optimal (France Conseil Scientifique de l’Evaluation 1996). In Ireland, an interesting case arose when Ministry of Finance officials were seconded to the academic institute (the Economic and Social Research Institute) undertaking an evaluation on the efficiency and effectiveness of the use of EU Structural Funds in Ireland (Boyle 1993).

Thus, a key question in ex post evaluations is identifying the most appropriate institutional location of the evaluators. This is as true of tradeable permit programmes as other public policies. The workshop participants identified many existing models. As noted above, evaluations can be conducted internally by the programme managers or by professional evaluators elsewhere in the government, or by independent evaluators either under contract to the government or acting completely on their own.

Does the institutional location of the evaluator also affect the type of evaluation undertaken? Apparently it can. Young (2004) points out that in Australia assessments prepared for governments typically tend to be broader in their focus than those undertaken primarily from an academic research perspective. Furthermore, for the evaluation of the Australian water entitlements trading programme, since several States and the Commonwealth were involved, it seemed obvious to administrators and participants that this particular review would need to be independent.

In the case of RECLAIM, the initial audit was undertaken by RECLAIM staff, but it was overseen by an Advisory Committee which included regulated firms, academics, environmental

NGO's, and regulatory agencies. This provided some quality control, a point to which we return below. Moreover, since dissenting opinions were published in the report, this allows for a degree of independence which is important for the credibility of the evaluation (Smith 2002).

Discussions during the workshop revealed the existence of a difference between countries in the reception accorded to evaluations from external sources. The reception given to the outside evaluations in both New Zealand and the United States were extremely welcoming and co-operative. Yet the process was not severely tested because the evaluations were not only generally very positive, but were expected to be so. It is not clear that the reception would have been quite as welcoming or the relationship quite as co-operative if the evaluations were negative.

VII Providing Appropriate Personnel Incentives

The question of the choice of evaluator is closely linked with more general issues associated with the division of responsibilities in the evaluation process, and incentives for personnel in the implementing agency. In particular, one issue requiring further investigation is whether differences in governments' receptivity to outside evaluations is systematically related to their personnel policies. One hypothesis would suggest that bureaucracies where personnel are so vulnerable that negative evaluations could place their careers in jeopardy would be less likely to welcome external evaluations. However, these same incentives would also make it much less likely that an internal evaluation would be unbiased.

For tradeable permit programmes – which remain at a relatively immature stage of development in most OECD countries – these issues are particularly important. Since tradeable permit systems represent a significant change from legalistic direct forms of regulation to more market-oriented economic forms of regulation, they will inevitably have significant implications for personnel within regulatory authorities. In some cases, the implications for agency staffing might result in pressures for emphasising the deficiencies of reforms (Smith 2002). For instance, a strong public sector union might have trouble with programmes that fundamentally change the nature of the bureaucracy, which tradeable permit programmes seem to do.

The issue of personnel management and organisational incentives could, therefore, be a key factor in determining the nature and effectiveness of ex post evaluations. In particular, evaluations could prove to be an effective means of resolving incentive-incompatibility problems within public administrations by ensuring that the incentives of those bureaucrats who are responsible for policy implementation are compatible with the underlying objectives of the policy. In the absence of external evaluations, upper-level management are relying upon information provided by people who have a vested professional interest in making the programmes appear to look good or bad (depending upon the relative importance of the points made above).

Thus, to some extent, the choice of evaluator may be less important than the management structure in which the evaluation is undertaken. In particular, it is important to have a clear delineation of responsibilities within the management of the evaluation process. Thus, Hahn and Litan (1997) have emphasised the need to ensure that responsibilities for regulatory analysis be distinct from rule-writing and regulatory enforcement. In such cases, even if the evaluator is 'internal' there may be a sufficient degree of autonomy and independence. Thus, the trade-offs between internal and external evaluators can be reconciled through an effective division of responsibilities within the regulated authority.

However, in all cases a body with overall oversight of evaluations seems to be key. In a study conducted in the late 1990's the OECD Public Management Service found that 10 out of 17 countries surveyed had dedicated bodies responsible for regulatory oversight (OECD 1997b). In

four cases these were located in industry, economics or commerce ministries, three were in budget or public management ministries, two were in central cabinet offices, and one was in a parliamentary auditing office.

For example, in Canada, the Regulatory Affairs Directorate of the Treasury Board is responsible for oversight of policy evaluations (Apogee Research 1997). In the United States, the Office of Management and Budget's Office of Information and Regulatory Affairs serves this role (Hahn and Litan 1997). In Japan, the Administrative Evaluation Bureau of the Ministry of Public Management has an analogous role (Japan MHPHT 20002).

Hard and fast rules concerning the optimal means of managing evaluations are not possible. However, it has been emphasised that two extreme positions are unlikely to be efficient (OECD 1997b), namely:

- full delegation to regulatory authorities themselves of responsibility for undertaking, overseeing, and acting upon the results of evaluations; and,
- full centralisation of responsibility for evaluation, without significant input from the agency or ministry whose regulation is being evaluated.

Not surprisingly, most OECD countries find themselves somewhere between these two extremes, although it would appear that the greater danger lies in too much decentralisation rather than too little.

VIII. Resisting Strategic Behaviour

Evaluations can upset vested interests. As such, issues of strategic behaviour and rent-seeking are endemic in the evaluation processes. Indeed, pressure on evaluators (particularly if internal) can be such that protection may be required in order to ensure that they are able to exercise their role effectively. Lutter (2003) has gone so far as to suggest the need to grant 'whistleblower' protection status for those who point out distortions which might have been applied by government officials in support of particular policies.

As noted above, those responsible for the policies that are being evaluated may present significant obstacles to effective evaluation if they have a vested professional interest in the outcome of the evaluation (i.e. continuation of the policy). Even if they are not directly responsible for the evaluation they may be unwilling to make important data and information available. Thus, resistance can often arise within the bureaucratic establishment (Morrall III 1997 and Smith 2002).

More fundamentally, regulators may see that they represent a particular constituency (often the beneficiaries of a policy) and not the community more widely (including those who bear the costs associated with the introduction of the policy). This can result in significant resistance to evaluation. Analogously, Deighton-Smith (1997) has pointed out that regulators may see themselves as representing a particular objective rather than social welfare generally.

The issue of regulatory 'capture' by those outside of the government who are affected by regulations is, of course, also important. This might manifest itself in the evaluation process itself since evaluations are never purely 'positive' exercises. The choice of criteria and methodology implies value judgements. The demand for 'better' regulation might in fact be interpreted as masking demand for 'less' regulation (see Deighton-Smith 1997).

In other cases, the beneficiaries may have significant influence. For instance, if a policy provides subsidies or other types of benefits, evaluations which might jeopardise their continuation will be seen as potentially damaging to their interests. For ex post evaluation, this may be particularly important since particular constituencies will have a stronger sense of 'rights' to benefits actually being received, rather than those being received prospectively in the case of ex ante evaluations (see Deighton-Smith 1997).

On the other hand, effective and credible evaluation can strengthen the government's hand relative to vested interests in their efforts to protect overall social welfare (see Holmes and Argy 1997). By providing reliable and analytical support for the need to undertake policy reforms, governments may be able to deflect some of the pressures from stakeholders with particular interests.

IX. Providing Adequate Quality Control

In order to be credible evaluations need to be seen to be of sufficient technical quality. More specifically, evaluations need to satisfy the following criteria (Boyle 1993):

- internal validity – i.e. are confounding factors adequately addressed, is the baseline plausible, etc.;
- reliability – i.e. is the analysis undertaken using appropriate methodologies, is the data reliable, etc.; and,
- objectivity – i.e. are the evaluation conclusions consistent with the outcomes of the empirical research.

In some cases, 'external validity' (i.e. are the results of the evaluations generalisable to other policy or sectoral contexts) may also be seen as an important attribute. In order to ensure such conditions are met, it is important to put in place mechanisms to oversee evaluation procedures. In this vein, the European Commission emphasises the importance of a steering group to oversee the management of evaluations (CEC DG for Budgets 1997). Interestingly, in the United Kingdom, Heads of Departments must sign 'Regulatory Quality Certificates' indicating that the evaluation was undertaken using satisfactory methodologies and sound data (UK Better Regulation Unit 1997).

Training and guidance can also be important means of ensuring good quality evaluations are undertaken. In Japan the Commission on Policy Evaluation and Evaluation of Individual Administrative Institutions provides guidance to ministries and agencies on undertaking evaluations (Japan MPHPT 2002). A dedicated 'Study Group on Policy Evaluation' was formed to provide specific guidance on evaluation methodologies. In Canada, the Regulatory Affairs Directorate of the Treasury Board is responsible for providing guidance on evaluation methods and management (Apogee 1997).

A large number of OECD countries provide guides for evaluators and managers of evaluations. Hopkins (1997) provides an overview of the contents of 12 such guides from seven different OECD countries. And finally, resources (human and financial) need to be available in order to ensure that evaluations are of a satisfactory quality. The EC's Communication on Evaluation estimates that budgets for evaluations will often represent in the region of 0.5% of total programme costs (CEC DG Budget 1997). However, this varies widely depending upon the nature of the evaluation and the context in which it is applied.

The final check on quality comes from having a transparent process that allows independent assessment of the evaluation. All ex post evaluations involve a host of assumptions and judgements. It is important that these be understood and evaluated by those who have the appropriate expertise and the interest. Lutter (2003) emphasises the importance of independent peer reviews of evaluations.

XI. Disseminating and Communicating Results

The need for effective dissemination is also key to ensuring that policy evaluations have an impact on the policy process. It is not enough to assume that ‘good science’ will lead to ‘good policy’. The results of the evaluation need to be communicated to those responsible for bringing about policy change in a manner which resonates (Hahn and Litan 2002). More generally, in order to have a wide impact, the results should be disseminated widely – a point made by the Japanese Ministry of Public Management (Japan MPHPT 2002). The need for effective dissemination of evaluation results has been explicitly mentioned in the OECD’s Council Recommendation on Improving the Quality of Government Regulation (1997b).

The widespread use of the World Wide Web has made the current possibilities for sharing information historically unprecedented. It is now possible to make vast amounts of information available to the public at very low cost. In many senses, tradeable permit schemes have been at the forefront of the effort of regulatory authorities to make use of modern technologies to disseminate information widely and inexpensively (See <http://www.epa.gov/airmarkets/trading/> for an example).

However, the existence of these technological possibilities also means that cost can not be used as an excuse for inadequate dissemination of evaluation findings. In general, the results of evaluations should be shared as widely as possible in as complete a form as possible. This should include not only the complete evaluation report with all supporting analysis, but also more synthetic documents which summarise the methodology and results. Others must be able to assess and comment on what was done in the evaluation. Since peer review has served the process of policy reform well for a long period of time, it should be empowered by the transparency of the evaluation methods and results.

The summary findings and non-technical presentation of the results should be disseminated as widely as possible for the non-specialist audience. While such reports should be intelligible to a wide audience – and particularly those affected directly by a regulation – the reports should be careful not to underplay the uncertainty and complexity of evaluation methodologies (see UK Treasury 2003).

Conceivably many different groups could be seen as entitled to have access to the information, including: programme managers, the broader bureaucracy, legislative committees or individual legislators, researchers in the academic community or independent research institutes and finally the general public. The proper management of ex post evaluations will have to not only identify who should have access to this information, but to whom the analysis should be targeted. These are not necessarily the same – even reports targeted at policy makers, may be made available to the public.

It is also worth noting that the form of the evaluation could have an impact on dissemination. Independent reviews are much more likely to be published in the journals that are regularly read and referenced by the academic community. Academics have a strong incentive to get everything they do into print. If, as generally supposed, academic journals are the most effective means for spreading the insights learned (particularly among foreign scholars), this

means that the choice of evaluation type (internal or external) may influence dissemination as well as content.

XI. Generating Demand for Evaluation

Analysis of the ‘supply side’ of policy evaluation is relatively well-developed. However, the demand side is perhaps more important. Those responsible for introducing, implementing and reforming public policies need to see the merits of policy evaluation. Perhaps more importantly, they need to face incentives to ensure that evaluations are undertaken and acted upon. Such ‘demand’ can be created through (OECD 1999b):

- sticks – such as mandatory requirements to undertake evaluations at specified points, or withdrawal of budget allocations if such evaluations are not undertaken;
- carrots – such as earmarking of funds for evaluations, or links with personal performance management systems; and,
- moral suasion – such as workshops and seminars which serve to point out the value of evaluation for efficient public administration.

The key point is that the internal demand for evaluations has to be genuine, and this must be reflected in the incentives faced by those who are responsible for implementing regulations. To some extent this runs counter to the bureaucratic culture which exists in many countries, encouraging those who are responsible for implementing particular regulations to feel ‘ownership’ over the programme rather than the fundamental policy objective.

If appropriate incentives are not in place, resistance to the evaluation process may arise, and the evaluation findings are unlikely to be implemented. In a sense, the incentives of the regulatory authorities need to be linked as closely as possible with meeting the social objectives of the programme writ large, and not the precise programmatic details of the instrument applied. This is, of course, a significant management challenge.

XII. Conclusions

While providing information and insights on technical issues associated with undertaking evaluations of tradeable permit schemes, the workshop also provided valuable lessons on the links between policy evaluation and policy-making generally. The main lessons in this area included:

- Policy evaluation should be seen as an integral part of performance management within public authorities, and not ‘add-ons’ which are discrete and isolated assessments of individual policies.
- The timing of evaluations is crucial, since this will determine the ‘scope’ for policy adjustments and reform. Since programmes typically evolve, mid-course evaluations can be particularly helpful in shaping that evolution. Mid-course evaluations will typically be more targeted and less formal than retrospective ex post evaluations.
- Much of the data that would be needed to begin to perform at least mid-course evaluations of these programmes are collected as a normal part of the monitoring and

enforcement process. When this is not the case, it is important to devote resources to the collection of data which is needed for evaluating a programme.

- Data access is at least as important as data collection. Wide access to the data necessary for undertaking evaluations reduces costs and increases the range of evaluations which the regulatory authorities can draw upon.
- The institutional location of the evaluator is key, with a strong case to be made for the use of external evaluators. However, it is also important to ensure that there is an appropriate division of responsibilities for the management of evaluations, irrespective whether the evaluation is conducted internally or externally.
- In a closely related vein, it is important that those within public agencies who are likely to be affected by the outcomes of policy evaluations face incentives which encourage full co-operation. Similarly, they must also be in a position to resist pressure from lobby groups who have a vested interest in the outcome of evaluations.
- Since a peer review of evaluations is so important, at a minimum complete reports on the evaluation methods and results should be transparent to the public. Where possible all the data used to perform the evaluation of the programme should also be publicly available.

In conclusion, while the specific results generated from individual evaluations are clearly important, some of the most important benefits associated with undertaking policy evaluations relate to the broader process of undertaking and managing evaluations and linking them with public sector reform more generally. Policy evaluation must be seen as an element of efficient public management in a broad sense.

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