Social Research Working Paper

September 2004

A Tradable Permit Program for Nitrogen Emissions to Lake Taupo

Geoff Kaine and Megan Higson

A Tradable Permit Program for Nitrogen Emissions to Lake Taupo

Social Research Working Paper 08/04

September 2004

Geoff Kaine and Megan Higson

Table of Contents

Acknowledgements

The authors wish to thank Ian Reeve, Institute for Rural Futures, University of New England, Armidale, New South Wales for allowing us to quote extensively from Kaine and Reeve (1993). We also thank John Casti, Tony Pleasants, Tanya Soboleva, Alan McDermott and Stewart Ledgard who contributed by way of many interesting and valuable discussions. Our thanks also go to Tony Petch, Justine Young and Sarah Mackay for their comments on drafts of this report.

1. Introduction

Our intention in preparing this report was to contribute to the discussions and debate concerning policies for managing nitrogen emissions from agricultural land in the Lake Taupo catchment. The views presented here are those of the authors and should not be interpreted in any way as reflecting the views, if any, of AgResearch on this issue.

In this report, we discuss some of the policy options that have been proposed for managing nitrogen emissions from agricultural land in the Lake Taupo district. Our main focus is on a proposal for a market in nitrogen emission permits. However, we do discuss other proposals such as an industry 'code of practice'.

The proposed market is intended to provide a mechanism which will facilitate structural adjustment such that nitrogen loadings remain within the assimilative capacity of the Lake as defined by current water quality targets (Environment Waikato 2003). For the purposes of this report we have assumed that the 20 per cent reduction in the nitrogen load entering the Lake that is needed to maintain water quality has been achieved. The purpose of the market is to facilitate the transfer of the allowable emission load between alternative land uses over time in line with changes in economic conditions and community and individual preferences over time.

Nitrate from both point and diffuse sources enter the Lake by means of ground water and streams. Emissions from point sources are relatively easily monitored and, at least in principle, point dischargers are capable of exerting control over the timing and quantity of their emissions. This means that changes in discharges due to actions taken by point dischargers are measurable, and that point dischargers are usually in a position to manage their emissions in both the short and long term.

Nitrate emissions from diffuse sources cannot be monitored directly. This means that the actual contribution of a particular diffuse discharger to the nitrogen load in the Lake cannot be identified. As a general rule, diffuse dischargers cannot exert control over the timing of their emissions but they can exert control over the quantity of their nitrogen emissions in the longer-term. Consequently, actions taken by diffuse dischargers to reduce their contribution to the nitrogen load in the Lake may only be measurable as a trend change over longer time periods.

The case can be made, on both equity and efficiency grounds that the adjustments entailed in reducing discharges from their current levels should involve point and diffuse sources. We believe that the market in emission permits that is described in this report

provides a framework which would allow for sharing of adjustment costs between point sources and diffuse sources. Furthermore, we believe that the type of market we describe has the potential to enable the water quality target to be achieved with the least cost combination of abatement and other measures.

In the next section the principal alternatives to a market in emission permits, namely regulation, education or moral suasion, and taxation are presented and critically evaluated. This is followed in the third section by a discussion of property rights and the issues involved in changing property right structures. After some discussion of general issues relating to the initial allocation of permits in the fourth section, the market in emission permits is described in some detail in the fifth section. This is followed by some further discussion of the issues relating to the initial allocation of permits. The prospects for implementing a market in emission permits are discussed in the final section.

2. Approaches to natural resource management

There are a number of alternative policy instruments that might be considered in achieving the policy objective of limiting discharges of nitrogen into Lake Taupo. Apart from the conventional instruments involving education, moral suasion and regulation the other major alternatives are fiscal instruments such as taxes and charges, subsidies and market instruments. In this section, the conventional instruments are briefly described and evaluated. This is followed by an evaluation of fiscal instruments.

2.1 Education, moral suasion and regulation

Education and moral suasion are important components of conventional approaches to protecting water quality, either as policy instruments in their own right or in conjunction with regulation. While the use of education and suasion to promote voluntary compliance with environmental standards has excellent short-run political appeal, overseas experience (Abler and Shortle 1991; Bohm and Russell 1985; Bradsen 1988) indicates a low likelihood of success. In the case of emissions from agricultural land, landholders have been effective in deflecting moral or altruistic appeals for practice change by mounting a defence based on their inability to stand increased costs, the contribution of agriculture to exports and the local economy, and historical precedent.

A feature of many existing systems of regulation is the granting of emission licences conditional on the implementation of 'best' management practices or the adoption of 'best' available technology, and subject to the provision that the practice or technology be 'economic'. In this context 'best' is that practice or technology which is believed to result in the minimum level of emissions. The meaning of 'economic' in this context is not always entirely clear. A strict interpretation may be taken to mean that the additional costs entailed in adopting the abatement measures should not threaten the viability of a technically and allocatively efficient enterprise. A more generous interpretation would be that the additional costs entailed in adopting the abatement measures should not threaten the viability of the enterprise under consideration. Typically, the latter interpretation prevails.

Usually, the authority responsible for the issuing of emission licences negotiates the renewal of licences periodically with each licence holder individually. In negotiation the individual discharger and the authority determine a program of abatement works which are believed to be 'best' and 'economic' at that time for that discharger. Since the renewal of emission licences is conditional on the implementation of abatement measures then, theoretically, the environmental standard will eventually be attained as dischargers will be required to progressively adopt increasingly stricter abatement measures. Given the progressive adoption of these measures is obtained through a negotiation mechanism the process is particularly sensitive to the economic and technical circumstances of dischargers and to the extent that the agency negotiator is prepared and able to act on threats of sanctions.

The success of this approach is predicated on the assumption that there is sufficient assimilative capacity in the environment to accommodate the level of emissions once the process is completed. If this is not the case then at least some dischargers must be forced to cease business if the environmental objective is to be achieved. And this raises the difficult issue as to the criteria which are to be employed by the authority in determining which licences will not be renewed, given that dischargers have complied with licence conditions. Depending on the extent to which compensation for cancellation is built into licence conditions, such situations can create victims who, with appropriate influence, can prevent termination by appeals to justice and national or regional interest.

The primary incentive for dischargers to participate in the negotiation process and to implement the agreed abatement measures is the threat that failure to cooperate or comply will result in either the non-renewal or revocation of licences. In many circumstances, for example where a licensed discharger has completed part of an

agreed program of abatement works, the penalty of non-renewal and closure of a discharger's enterprise may be seen to be excessive. This suggests that dischargers have some room to manoeuvre in attempting to minimise their expenditures under agreed abatement programs. Although they may be co-operative to some degree in the negotiation process, in order to minimise costs they may attempt to postpone planned abatement activities and expenditures and delay actual activities and expenditures.

Given that the authority may have to depend on dischargers to provide the information necessary to determine which practices and technologies are 'best' and 'economic', the opportunity exists for dischargers to overstate costs and understate revenues so as to delay action and expenditures on abatement for as long as possible. Furthermore, the authority may have to depend on dischargers to collect and provide information necessary for the authority to assess compliance. This may create a conflict of interest for dischargers.

In summary, the seriousness of the consequences arising from the non-renewal of licences may well neutralise non-renewal or revocation of licences as a threat to ensure cooperation and compliance. This, together with the reliance of the regulatory authority on the dischargers themselves for the provision of information, means that dischargers may well be able to minimise their rate of investment in abatement measures and delay the achievement of environmental standards.

In addition to the relatively slow rate of progress that might be expected with an approach based on the negotiated adoption of 'best' practices, another potential disadvantage of this approach is that environmental standards may be achieved at substantially greater cost than is necessary. A precondition for achieving a given level of abatement at least cost is that the marginal cost of abatement be equal across abatement programs (see, for example, Baumol and Oates 1975; Falconer 1998; Nicolaisen, Dean and Hoeller 1991; Parminter 2003; Rae 1997; Randall 2003; Stavins and Whitehead 1992). The ideal outcome under the 'best' practices approach is that each landholder adopts 'best' practice. If the cost of abatement per unit of emissions varies between landholders then the marginal cost of abatement will vary from landholder to landholder. Consequently, for a given level of abatement, the least cost combination of abatement measures will involve differing levels of abatement across landholders.

This suggests that the abatement programs instituted by landholders under the 'best management' approach should be the product of a deliberate effort to equalise the cost per unit of abatement across landholders and not determined primarily by the apparent

capacity of landholders to fund abatement works or sustain revenue losses as a result of implementing abatement measures. In the absence of a deliberate effort to equalise marginal abatement costs across landholders, the 'best management' process represents a relatively costly and inefficient approach to the achievement of environmental standards.

The 'best' management practice approach has been proposed for diffuse source discharges. For example, this approach was used for non point-source pollution in the USA for the ten years following the 1977 amendments to the *Clean Water Act.* Over this time it became clear that the approach, while conceptually attractive, was largely ineffective (Wilkinson 1987). Wilkinson identified three reasons for this failure which were:

- Implementation was voluntary,
- Very few site specific practices were developed so that agencies and the private sector relied upon very broad generic guidelines, and
- The best management practices that were developed tended to become ends in themselves encouraging a rote compliance attitude rather than a determination to find the most effective solutions.

2.2 Load-based taxes

In principle, the target inflow of nitrogen into Lake Taupo could be achieved through the levying of a tax on nitrogen emissions. The employment of taxes or effluent fees as instruments for achieving environmental goals has long been advocated by economists, primarily on the basis that a system of taxes or charges has the capacity to enable environmental standards to be achieved at the least cost to society (Baumol and Oates 1975; Parminter 2003; Scrimgeour and Piddington 2002).

The essential economic principles involved in the operation of a system of emission taxes are similar to those involved in the operation of a system of tradable emission permits. Assume a tax is levied at a fixed rate per kilogram of nitrogen discharged. A cost minimising discharger will undertake abatement works to the point where the marginal cost of abatement equates with the rate of tax. In other words, dischargers will reduce their nitrogen emissions to the level at which the cost of reducing emissions by one more unit equals the tax rate, more or less. By setting the tax at an appropriate rate the target aggregate level of emissions can be obtained.

The attractiveness of a system of emission fees or taxes as instruments for achieving environmental standards lies in their apparent simplicity. Unfortunately, such systems are often impractical in reality (Baumol and Oates 1975; Nicolaisen, Dean and Hoeller 1991; Glasbergen 1992; Scrimgeour and Piddington 2002; Stavins and Whitehead 1992). To begin with there is the problem of determining the appropriate rate of tax on emissions. In the absence of detailed information on abatement costs there will be uncertainty as to the response of dischargers to the imposition of a tax. If the rate of tax is set too low then environmental standards will not be met. Consequently, the rate would need to be raised and then subsequently lowered as the administrative authority pursues an iterative path converging on the target level of emissions. This may entail costly adjustments and readjustments on the part of dischargers in terms of production, input mix and investments in abatement technology.

The tax rate will also require periodic revision to accommodate changes in the volume of emissions due to the expansion or contraction of production by existing dischargers in response to changing economic conditions, and to allow for changes in the prices of inputs for abatement measures. The tax rate would also need to be revised to accommodate changes in the volume of discharges due to the entry of new enterprises into the catchment or the exit of existing enterprises from the catchment. This need for repeated changes in the tax levied on emissions in order to maintain the target level of emissions is an unattractive prospect from an administrative and political perspective.

Baumol and Oates (1975) note that another difficulty with the introduction of a tax on emissions is that such a tax may involve substantial increases in costs to dischargers relative to other regulatory policies. While a system of emission taxes or charges might ensure that the target level of abatement is achieved at least cost from a social perspective, the taxes or charges themselves impose a new financial burden on dischargers. The evidence suggests that this tax burden can be extreme (see Jacobs and Casler 1979). The political acceptability of the tax might be improved somewhat if the tax revenue was employed to fund public abatement programs — an approach that has been used successfully in France (Hahn 1989).

2.3 Input-based taxes and input imputing

Another approach that could be taken to achieve the water quality objective in Lake Taupo is to implement a tax on inputs to land based production systems. In this approach the tax is based on an ‘imputed’ rate of nitrogen emissions rather than the actual rate of emissions. The imputed rate of emissions is estimated given data on the

farm inputs and land management practices that are thought to influence emissions. For example, a tax could be levied on livestock given an imputed rate of nitrogen emissions per livestock unit.

An approach involving a system of input taxes is subject to the same criticisms that can be levelled at load based taxes. In addition, there is the problem of determining an appropriate regime of taxes across the spectrum of all relevant inputs, and the fact that repeated corrections to the regime may entail costly adjustments and readjustments on the part of landholders. Again there is the problem that the tax regime would require continual revision as input and product prices vary over time and innovative abatement technologies emerge. Also, the political feasibility of introducing such a regime of taxes needs careful consideration as they involve the imposition of an additional and unpredictable financial burden on landholders.

The feasibility of employing a regime of input taxes as a means of achieving environmental objectives also depends on the ease with which a rate of emissions can be imputed across the range of relevant inputs. If the rate of emissions for a given input varies widely depending on the conditions of use as is the case with nitrogen emissions from livestock (Environment Waikato 2003), then the tax regime that may be required may need to be quite elaborate. And the more elaborate the regime the more likely that it will be impractical from an administrative perspective.

Consider, for example, the imposition of a tax on livestock. The rate at which nitrogen is discharged from a property enters groundwater and ultimately the Lake depends on a number of factors. These include the soils and topography of the property and the extent and type of vegetation on it. The type of livestock enterprise conducted on the property, the stocking rate and grazing management practices followed by the landholder also have an impact on the rate at which nitrogen is discharged from the property (Environment Waikato 2003). Information about each of these factors for each property is required in order to derive an imputed rate of emissions for each property. Furthermore, as changes in any of these factors may alter nitrogen emissions then each of these factors must be monitored over time and imputations and tax rates adjusted accordingly.

In these circumstances the process of determining the tax liability of landholders and the costs of enforcing compliance could become quite demanding. Furthermore, the greater the range of factors that influence emission rates the greater the need for repeated adjustments to the tax regime. For example, not only will the tax liability of individual

landowners change as they change enterprises, alter their enterprise mix, adopt different grazing practices or invest in abatement works, the rate of tax consistent with the environmental standard will also change.

Another aspect of this approach which is of concern is the scope that a complex regime of input taxes creates for litigation. As the tax liability of a landholder is assessed on the basis of an imputed rate of discharge rather than actual discharges, then the imputation process itself may be subject to challenge. And there is no guarantee that the level of precision and uncertainty that may be required to provide an adequate legal defence of the imputation process will be consistent with that which is efficient from an economic perspective. Of course, the imputation process may also be subject to legal challenge if employed under a system of tradable emission permits.

The vulnerability of a policy instrument based on imputation is primarily due to the fact that the imputation is based on imperfect knowledge regarding the relationships between a range of production factors and the rate of emissions. Each of these relationships is a potential source of error and therefore could be challenged. This means that substantial public resources may need to be devoted to measuring these relationships in order to establish a defensible system.

3. Property rights

Establishing and enforcing water quality standards by constraining economic activities which generate nitrogen emissions necessarily involves restructuring the rights and obligations of individuals and organisations, and redistributing wealth. This restructuring and redistribution occurs irrespective of the approach that is used to achieve the standards - regulation, taxation, incentives or some kind of market though the approach that is used will influence the precise nature of the new structure of rights and the resulting distribution of wealth.

Consequently, the issues that are canvassed in the following discussion apply equally to any approach to establishing and enforcing water quality standards.

3.1 Discussion

The establishment of limits on nitrogen emissions in order to achieve a certain standard of water quality implies that the existing structure of rights and obligations has now become unacceptable in some way to one or more persons or institutions. The decision whether to proceed with the establishment of limits on dischargers requires a political judgement of the strength of public concern about the existing rights structure and of the public acceptability of any proposed changes to that structure.

Until recently, diffuse dischargers such as rural landholders exercised an unconstrained right to discharge nitrogen into ground and surface waters through their agricultural activities. The exercising of this right is now found to be in conflict with the rights that others wish to exercise with regard to the quality of water. As these others, under the existing rights structure, are unable to obtain compensation for any diminuition of the rights they wish to exercise, the unconstrained right to discharge amounts to a presumptive privilege (Bromley 1991).

In other words, the right to discharge nitrogen takes precedence over the rights of others where the two are in conflict without imposing any obligation on the part of dischargers to compensate others for their loss. This state of affairs may seem to have unfairly favoured dischargers but this is not the case (Weingarten 2001). It simply reflects the fact that, until relatively recently, the right to discharge nitrogen had not been perceived to infringe on the rights or activities of others. It is only now that nitrogen emissions are seen to infringe on the activities of others and it is not acceptable to the community and the government to allow unrestricted rights to discharge nitrogen.

The creation of a mechanism that establishes rights to water quality and enables those rights to prevail over previously unlimited emission rights involves a substantial restructuring of the rights of individuals and organisations. If this restructuring of rights is to entail the creation of a market of some kind in nitrogen emissions, a prerequisite to the creation of that market is that the right to discharge nitrogen be an exclusive and enforceable right, that is, a property right. To be exclusive, the existing privilege that *all* individuals in the catchment currently possess to discharge nitrogen must be surrendered.

Those individuals who are included in the subsequent allocation of rights to nitrogen emissions as a property right surrender their previously unrestricted privilege in exchange for an exclusive right to a restricted level of emissions. Those individuals who are not included in the subsequent allocation of emission rights as a property right forfeit their right to discharge. Consequently, the creation of a market in rights to emissions involves the restructuring, in some form, of the emission rights of all individuals. This is the case with any procedure designed to limit the activities of dischargers to achieve a particular standard of water quality.

The removal of the privileged status of the right to discharge nitrogen has further ramifications. The unrestricted right to discharge nitrogen was part of the parcel of rights attendant upon land ownership. To create rights to water quality that prevail over the unlimited right to discharge nitrogen implies that the attenuation of the parcel of rights attendant upon land ownership in this manner is justifiable.

Consider, for example, that nitrogen emissions occur as the result of an agricultural enterprise operated by a landholder. In operating the enterprise the landholder exercises claims over a bundle of rights of which the right to discharge nitrogen is one (Coase 1960). The ownership of land is the source of the legitimacy of the landholder's claims over this bundle rights. To the degree that the landholder is prevented from exercising claims over other rights in the bundle by the restriction of the right to discharge then the other rights that constitute the bundle are attenuated. This attenuation may take the form, for example, that livestock cannot be allowed to graze along the banks of streams or that the number of livestock is restricted.

The attenuation of rights could, of course, involve much more substantive forms. The restriction on emissions may, given the techniques currently available, prevent a landholder from undertaking pasture improvement. As a consequence the landholder may be unable to profitably undertake a livestock enterprise such as dairy production. Hence, the proscription of unconstrained rights to discharge nitrogen involves an attenuation of property rights in rural land by restricting the uses which may be made of the land by the landowner.

We have argued that the adoption of a water quality standard that constrains the discharging of nitrogen amounts to a redistribution of rights and wealth to the benefit of non-dischargers at the expense of dischargers. This means the selection of a standard for water quality is not simply a technical matter. It is also a matter for informed social choice.

The selection of a standard for water quality determines the extent to which rights to discharge will be constrained and the degree to which the bundle of property rights entailed in owning land will be attenuated. The stricter the standard, the greater the reduction of presumptive privilege in favour of rights with regard to water quality, and the greater the attenuation or abrogation of other rights in the bundle of rights attendant on land ownership. While the adoption of a quality standard may be thought to be a technical matter, it does in fact involve choices regarding the redistribution of rights and wealth between individuals.

To summarise, the existing unrestricted right of landholders to discharge nitrogen must be bounded in order that rights of others to water quality may be created and exercised. Importantly, the private and social costs imposed by declining water quality must be seen by the community to be an adequate justification for infringing the historical privilege of rural landholders to discharge nitrogen and for invoking changes in the management and use of rural land. The adoption of a water quality standard amounts to a redistribution of rights and wealth to the benefit of non-dischargers at the expense of dischargers. Therefore, the selection of a quality standard is a matter for informed social choice as well as a technical matter.

4. Allocation of emission rights

In this section some general issues relating to the allocation of tradable emission rights are discussed. We commence with a discussion of the allocation of emission rights by sale and then examine 'gifting' procedures. The points that are made regarding each of these approaches to allocating rights do not apply in all circumstances. Their validity depends on the precise nature of the rights which are to be allocated and on the bio-physical, social and economic context in which the market is intended to function. We should also bear in mind that in this context we are actually discussing the replacement of a previously unchallenged and unrestricted right to discharge nitrogen with a restricted right to discharge and a diminished bundle of rights entailed in owning rural land.

4.1 Allocation by sale

The distribution of losses and gains among landholders will depend heavily on the procedure employed to determine the initial endowment of emission rights received by each discharger. Allocation procedures which involve the sale of rights, usually in the form of permits of some kind, can appear attractive as the discharger is explicitly forced

to ‘pay’ for the privilege of ‘polluting’. However, the application of the ‘polluter pays’ principle by employing auction or tendering procedures to distribute rights may be unwarranted in this context for three of reasons.

First, dischargers will have experienced a diminution in rights and loss of wealth simply because of the loss of the presumptive privilege to discharge. This is because the resulting diminution of property rights in land translates into either restrictions on land use and income, increased costs entailed in abatement measures, or both. In other words, dischargers must bear the costs of either implementing abatement measures or else bear losses associated with scaling down, changing or closing enterprises. To require that as well as bearing these costs and losses, they purchase their initial endowment of discharge rights maybe judged unfair by many in the community.

Second, the adoption of a water quality standard places a constraint on nitrogen emissions in aggregate. The creation of a legally enforceable right to discharge within that constraint means that terms such as ‘polluter’ do not accurately describe the status of dischargers. The act of discharging is not an activity which may be undertaken without regard for others. The adoption of a water quality standard is intended, at least in principle, to restrict nitrogen emissions to a point at which the risk of any undesirable affects on other parties are deemed acceptable by the community. This means activities involving nitrogen emissions are an authorised activity provided the aggregate level of emissions remains within the bounds defined by the standard. Consequently, dischargers cannot be regarded as engaging in activities that are ‘punishable’ because they are ignoring the rights of others or the environment.

Third, at a more practical level, for the sale of emission rights to be meaningful the volume of emissions available for purchase must be less than the current or anticipated rate of emissions in aggregate. If this is the case, then at least some dischargers must be in a position to immediately reduce emissions. With respect to agricultural discharges of nitrogen this may not be the case. The accumulated reserve of nutrients in soils means that any change in farming activity is unlikely to lead to an immediately matching reduction in the rate of emissions. In addition, the current level of information concerning the effectiveness of abatement measures may be inadequate. Therefore, an extensive research effort spanning a number of years may be required before landholders will be in a position to invest in abatement measures with confidence. Consequently, allocating emission rights by sale may well be impractical.

The sale of rights may, however, be an appropriate method of allocation when rights are being expanded rather than restricted. This might be the case if it was found that the initial allocation of rights was too limited and that emissions could be increased without breaching the water quality standard.

4.2 Allocation by gifting

We have argued that changing an unlimited privilege to discharge nitrogen into a restricted emission right elevates the rights of other parties over the rights of landholders and constitutes a redistribution of wealth in favour of the other parties. In other words, the restructuring of rights in favour of others diminishes the wealth of landholders. Hence, even with 'gifting' of emission rights dischargers will experience losses except where a discharger is allocated rights in perpetuity equal to their current rate of emissions.

Gifting emission rights based on historical emissions, otherwise known as 'grandparenting', is open to the criticism that those dischargers that may have acted in the past to reduce their emissions receive a smaller endowment of rights and, therefore, are 'penalised' for having implemented abatement programs. Conversely, those that have not taken such socially desirable actions are 'rewarded' with a greater endowment of permits. The extent to which this criticism is valid depends on the extent to which emissions were known to cause problems in the past and on the extent to which dischargers had been called upon to reduce their emissions. In the case of Lake Taupo emissions of nitrogen have only recently been shown to pose a long term threat to the quality of water in the Lake.

The validity of this criticism also depends on the extent to which abatement measures are uniform in their effectiveness and efficiency. Some individuals may have been able to implement abatement measures at relatively low costs while the costs to others of similar measures may be relatively high. Alternatively, the measures that might have been implemented by some dischargers may be ineffective in the circumstances faced by others. In some cases relatively inexpensive measures may be effective whereas in other cases only relatively expensive abatement measures would be effective. Finally, in some cases abatement may be prohibitively expensive and the only means of reducing emissions is the closure of the business.

The fact that a particular discharger has not adopted abatement measures that may have been adopted by others is not, of itself, sufficient grounds for concluding that the discharger has, in some sense, disregarded the rights of others or failed to follow accepted industry practice and should necessarily be allocated a smaller endowment of emission rights.

In the case of Lake Taupo the mechanisms by which nitrogen are accumulated in the soil and then leached into ground and surface waters are still the subject of investigation. Consequently, our current understanding of abatement measures even now is limited. Hence, landholders cannot be justifiably criticised for not having acted in the past to reduce their emissions. Consequently, allocating emission rights on the basis of historical emissions cannot be readily dismissed on the grounds that it unjustly penalises those who have implemented abatement measure while unfairly rewarding those who have not.

A second possibility when gifting or grandparenting emission rights is to allocate the same volume of emission rights among all landholders irrespective of their historical emissions (that is, averaging). This method of allocating rights has two potential disadvantages. The first relates to the distribution of adjustment costs among landholders. Those landholders who face the greatest adjustment costs in terms of reductions in income and increases in costs as a result of the imposition of limits on emissions will be required to obtain emission rights from landholders who face the lowest adjustment costs if all landholders are allocated the same emission rights. Consequently, this type of allocation merely serves to redistribute adjustment costs among landholders in such way as to transfer wealth from those experiencing the greatest economic disruption to those experiencing the least economic disruption. This outcome is likely to be both inequitable and economically inefficient.

The second shortcoming of this allocation procedure relates to the practical issues of implementing abatement measures. Logically, if all landholders receive an equal share of emission rights then a proportion of landholders will be required to immediately implement abatement measures or obtain additional emission rights from another party. The point was made earlier that, with respect to agricultural discharges of nitrogen it is unlikely to be the case that abatement measures can be implemented immediately. Consequently, allocating emission rights equally among landholders appears impractical.

Allocating emission rights equally among landholders may appear attractive because it provides a mechanism for compensating, to some degree, those landholders that suffer losses in terms of foregone opportunities. These losses may take the form of foregone

future income or a decline in asset values. Our concern here is not with the question as to whether compensation should be paid in these circumstances. Our concern, as already indicated, is that this approach to compensation is inequitable, inefficient and possibly impractical. If compensation is to be awarded to those that suffer losses in terms of foregone opportunities then perhaps that compensation should be made in the more orthodox and explicit form of transfer payments.

Assuming there are significant differences between dischargers in terms of the costs and effectiveness of abatement measures then, from an economic efficiency perspective, the most appropriate means of distributing of emission rights is on the basis of historical emissions. Provided emission rights are tradable, the distributing of rights on this basis ensures that those that are likely to face the highest abatement costs are the most likely to retain emission rights and the least likely to implement abatement measures. Those that are likely to face the lowest abatement costs are the most likely to dispose of emission rights and implement abatement measures. Hence, in our opinion, the water quality standard is more likely to be attained with the least cost combination of abatement projects than is the case with other allocation procedures.

4.3 Allocation by regulation

The creation of regulations, industry standards or codes of conduct governing activities that result in emissions are all procedures for implicitly allocating emission rights among landholders. The effects of these allocation procedures on the adjustment costs of landholders and other stakeholders in the catchment depend heavily on the actual design of the regulation, standard or code. This makes generalisations about the effects of these allocation procedures difficult to make.

For example, a set of regulations might be formulated defining the types of activities that may be undertaken by landholders and specifying the manner in which those activities may be conducted. If this set of regulations were applied uniformly to landholders then, to the degree that the rate of emissions varies depending on location specific factors such as livestock type, soil type and topography, it implicitly assigns emission rights differentially among landholders. In these circumstances, adjustment costs will be distributed among landholders in line with the degree to which they are required to invest in abatement measures or scale down, change or close enterprises in order to conform to the regulations. The resulting distribution of adjustment costs may bear little relationship to the historical distribution of emissions among landholders. Consequently, this approach is likely to be relatively ineffective in achieving the water quality standard.

This approach is also likely to be inefficient to the degree it prescribes the type of adjustments landholders must make to reduce nitrogen emissions. The fact that adjustment costs are not distributed among landholders in line with their emissions may mean some in the community might regard this approach as inequitable.

Alternatively, a code of conduct might be formulated defining the types of activities that may be undertaken by landholders and specifying the manner in which those activities may be conducted taking into account site specific factors that influence emissions such as livestock type, soil type and topography. Such a code would implicitly assign emission rights among landholders roughly in line with their historical level of emissions. Adjustment costs are likely to be distributed among landholders in line with the degree to which they are required to invest in abatement measures or scale down, change or close enterprises in order to conform to the code. To the degree this approach prescribes the types adjustments landholders must make to reduce their emissions this approach is likely to be inefficient in achieving the water quality standard. The fact that adjustment costs are more likely to be distributed among landholders in line with their emissions may mean many in the community might find this approach acceptable.

5. Transferable emission permits

In this section we describe a hypothetical market in transferable nitrogen emission permits for the Taupo catchment. The market has been constructed on the basis that a gifting or grandparenting procedure would be employed to allocate permits among dischargers and that the twenty percent reduction in emissions required to achieve the water quality standards set for Lake Taupo has been achieved by other means (see Environment Waikato 2003)

5.1 Market fundamentals

A fundamental difficulty in designing tradable permit programs that incorporate emissions from diffuse source is the inability to measure emissions (see Higson and Kaine 2004). An additional difficulty in regard to nitrogen emissions into Lake Taupo is the fact that most may take twenty years or more to enter Lake. This is because most emissions must first pass through the groundwater system before entering surface streams that empty into the Lake. Hence, measurement of emissions of nitrogen from land in Taupo is not feasible and there is little relationship between current emissions and nitrogen concentrations in the Lake.

These two difficulties have three consequences. First, emissions from diffuse sources can only be estimated on the basis of land use factors that are associated with emission such as livestock numbers and types, land area soil type and so on. Second, a market in tradable emission permits, like any other policy intervention, would have to operate independently of observed changes in nitrogen concentrations in the Lake. In other words, contemporary changes in nitrogen concentrations in the Lake should have no direct influence on the currently permitted volume of emissions because such changes reflect emissions in the relatively distant past. Third, the volume of emissions that are permitted should change over time in response to new information on the relationships between the factors and emissions, and new information on the relationship between emissions and water quality in the Lake.

The need to estimate emissions means that the cornerstone of the market would be a nutrient budgeting model for agricultural land uses such as OVERSEER® (Wheeler et al 2003).

These types of models provide estimates of nitrogen leaching from land based on a number of contextual characteristics such as stock type and number, net supplementary feed inputs, and fertiliser loading based on soil type and rainfall as well as topography.

In using a model to estimate nitrogen emissions the model acts as a mechanism for imputing emissions. To achieve the target rate of nitrogen emissions into the Lake the aggregate volume of imputed emissions should be less than or equal to the target rate and each landholder should possess sufficient emission permits to match their imputed emissions. Hence, emission permits should be described in terms of the imputation process embodied in the nutrient budget model. As such the permits are defined, and the market operates independently of, actual emissions and observed changes in nitrogen concentrations in the Lake.

By defining emission permits in terms of processes embodied in the nutrient model the difficulty of measuring actual nitrogen discharges is avoided. Hence, the measurement difficulties typically associated with diffuse source pollution are circumvented. Furthermore, as emissions are determined in the model on the basis of estimated relationships between emissions and contextual characteristics, the model can be used to infer constraints on those characteristics which are inputs to agricultural production processes such as livestock numbers and types. This means that the nitrogen permits for a particular landholding can also be expressed in terms of the contextual characteristics of that holding as well as units of nitrogen emissions. This has four advantages.

First, because permits can be expressed in terms of contextual characteristics then, in principle, permits can define constraints on the permissible combinations and levels of inputs into agricultural production processes such as livestock numbers and types. This allows the landholder to evaluate nitrogen permits in terms of the economic value of the combinations of production inputs and management practices possession of the permit allows. The inputs would be translated to give the emission equivalent of kilograms of N per hectare per annum to facilitate trading. In other words, the permits can be expressed in terms of the opportunities for using land they make available, which is precisely the basis on which landholders will value the permits.

This perspective is consistent with Lancaster's product attribute theory (Antonelli 2004; Lancaster 1966; Lancaster 1971; Hoehn et al 2003). In this fundamental work Lancaster argued that demand for a product is not for the product as such. Rather, the value of a product lies in the characteristics of the product attributes and potential those attributes provide to purchasers to meet their needs. In the context of nitrogen emission permits this means that the value landholders derive from permits resides in the additional economic opportunities the permits allow them in terms of land use.

Second, expressing emission permits in terms of permissible combinations and levels of inputs into agricultural production processes such as livestock numbers and types facilitates monitoring of landholders' compliance. In principle, inputs to agricultural production processes are measurable whereas emissions are not.

Third, expressing emission permits in terms of contextual characteristics such as soil type and topography and permissible combinations and levels of inputs into agricultural production processes such as livestock numbers and types means that trades can be visualised in terms of familiar farm management measures which facilitates the exchange of emissions between landholders.

A unit of nitrogen emissions creates different economic opportunities in different locations depending on such characteristics as soil type. Conversely, a particular combination of livestock types and numbers will require a different volume of emissions permits in different contexts. Hence, any trade in emission permits between landholders requires determining the relevant changes in the production possibilities for each of the landholders. In other words, an exchange rate must be calculated to determine the rate at which emissions in one context convert to a unit of emission in another context. This exchange rate is embodied in the model and is given by the relationship between estimated emissions and the contextual characteristics, including inputs to agricultural production processes.

Fourth, our understanding is that, under the Resource Management Act, consents for emission permits that are expressed in terms of permissible combinations of inputs into agricultural production processes given contextual characteristics such as soil type and topography may be traded whereas consents for emissions permits defined as a polluting discharge into water cannot be traded (Milne and Mooar 2002; Sharp 2002).

Earlier we stated that by describing emission permits in terms of the imputation process embodied in the nutrient budget model then the permits are defined, and the market operates largely independently of, actual emissions and observed changes in nitrogen concentrations in the Lake. However, we do not mean to imply there is no relationship at all between the market, actual emissions and nitrogen concentrations in the Lake. These are linked in two ways.

First, as new knowledge is acquired about the relationship between contextual characteristics and emissions then the nutrient model may be updated. As a consequence we would expect that over time estimated and actual emission will become more closely correlated.

Second, this new knowledge may result in revisions in the aggregate volume of nitrogen emissions that may be permitted to enter the Lake. Such revisions necessarily require a corresponding adjustment in the aggregate volume of estimated emissions that is allowed under the permits that have been issued. We would expect that over time the aggregate volume of estimated and actual emission would become more closely correlated as information on the relationship between emissions and contextual characteristics accumulated.

The period of time in which emission permits are denominated is a critical issue in designing a market in tradable permits. On the one hand permits should not be denominated in periods of time that are shorter than the rate at which at least some landholders can implement abatement actions. Otherwise landholders may find themselves contravening the conditions of their permits and lacking the capacity to comply with those conditions. On the other hand, permits should not be denominated in periods of time that are substantially longer than the rate at which most landholders can implement abatement actions. To do so would unnecessarily reduce the rate of transfer of emission permits between alternative land uses thereby reducing the efficiency of the market and imposing economic losses on the community.

The dynamic behaviour of the supply of the natural resource can be another important consideration in determining the time denomination of permits. Generally speaking the aggregate volume of potential emissions authorised through the permits should change over time in accord with changes in the capacity of the environment to assimilate emissions.

In the Taupo context variations in the capacity of the Lake to assimilate nitrogen occur over long time horizons even though there is substantial variability in the volume of emissions from land in the short term (see Power et al 2002). Once nitrogen emissions from agricultural land pass through the soil and reach the water table they diffuse through the ground water at a rate that is proportional to the difference in the nitrogen concentration of the emission and the ground water. The volume of emissions is small relative to the volume of ground water. Hence, seasonal and annual variations in the volume and concentration of emissions are dampened by the ground water system. Consequently changes in the concentration of nitrogen in the groundwater, and ultimately in the Lake, occur gradually over periods of years.

This means that the time dynamics of the assimilative capacity of the Lake is not a major constraint in determining the time denomination of permits in the Taupo context. In fact the dynamics of nitrogen cycling in the Taupo context suggests that seasonal variations in emissions may be safely ignored and emission permits can be specified in terms of an average annual rate of emissions.

Note that, administrative efficiency is also a consideration in determining the time denomination of permits and, in this case, there are substantial costs associated with revising resource consents.

We conclude that emission permits should be denominated in terms of an average annual rate of emissions and the corresponding combinations of inputs into agricultural production processes such as livestock numbers and grazing management practices that are permissible given the contextual characteristics of the land such as soil type and topography. This timeframe is sufficient for nitrogen emissions to change to some degree in response to adjustments made by landholders in their farming operations. This timeframe would also be consistent with the specification of nutrient budget models such as OVERSEER® (Wheeler et al 1999).

Emission permits are issued to existing landholders as part of a resource consent process. The permits are issued for the life of the consent and are allocated in accord with landholders' historical emissions as estimated by the nutrient budget model. The permits contain a statement setting out the combinations and levels of inputs into agricultural production processes that are authorised given the contextual characteristics

of the landholder's property. The permit also contains a statement indicating the corresponding average annual volume of emissions that is authorised, until such time as the resource consent expires or the permit is traded. The permits are temporarily or permanently transferable either in whole or in part.

As emission permits are established within a resource consent process the trading of permits amounts to a formal market in resource consents.

A disadvantage of using nutrient budget models to estimate nitrogen emissions is the scope that it creates for litigation. As emission estimates are based on an imputed rate of discharge rather than actual discharges the imputation process as embodied in the model may be subject to legal challenge.

5.2 Trading rules

Trading can occur between market participants at anytime. Trading would be facilitated by establishing a central register recording the details of permits and reporting those interested in buying or selling permits. This would help to coordinate market transactions and reduce search costs for landholders. Trading rules should be simple and enhance the managerial flexibility of permit holders (Stavins 2000)

Ideally, trades would be authorised through a streamlined resource consent process. When a trade is negotiated the responsible regional council would be required to calculate for each landholder the required change in emissions and the corresponding change in the combinations and levels of inputs into agricultural production processes that are permissible given the contextual characteristics of their land. Our understanding is that, in principle, this procedure should be consistent with the consenting process. Permit holders negotiating a trade hold resource consents for a controlled activity, such as farming, which specifies limits on the inputs into agricultural production. Any increase beyond these limits would require a change in the consent. Hence, resource consents and the conditions attached to them are tradable.

In principle, trading of permits may be initiated in two ways. Typically, trading will be initiated by landholders in response to changes in their circumstances. Our view is that trading of this type should be possible at any time. However, trading may also be initiated in response to changes in the specification of, or the parameters in, the relationships between contextual characteristics and emissions of nitrogen as embodied in the nutrient budget model. Such changes in the model may be prompted either by the acquisition of new information, or by the development of agricultural technologies and practices.

A change in the relationships between contextual characteristics and emissions of nitrogen as embodied in the nutrient budget model has two implications. One implication is that the estimated rate of emissions will change for at least some landholders and the changes in the estimates may differ across landholders. This means that at least some landholders are likely to have too few emission permits to cover their newly estimated rate of emissions. Other landholders are likely to have emission permits in excess of their newly estimated rate of emissions. Landholders may reconcile the differences between their permit holdings and their revised emission estimates by either buying, selling or leasing permits, or by changing land use.

The second implication of a change in the relationships between contextual characteristics and emissions of nitrogen as embodied in the nutrient budget model is that the aggregate of the revised estimates is likely to differ from the target rate of emissions. This raises issue as to how the emission permits that have been issued are to be adjusted to match the target.

A variable cap and trade program is one approach to resolving this issue (Higson and Kaine 2004). Generally, the variable cap and trade program suits situations where the supply of the natural resource is highly variable and market participants can adjust their use of the resource rapidly. This approach involves expressing emission permits as a fixed proportion of the target aggregate rate of emissions. The actual volume of emissions authorised under a permit would then be revised each time the nutrient budget model is revised. This concept is similar in principle to the annual revisions made to the total allowable catch in fisheries (Guerin 2003). We believe this method is impractical in the Taupo context because such revisions would require landholders to alter their land use in order to accommodate the corresponding changes in the combinations and levels of inputs into agricultural production processes they are permitted given the contextual characteristics of their land.

Alternatively, the difference between the aggregate of the revised estimates of emissions and the target could be resolved by implementing a fixed cap and trade program and authorising the regional council to trade permits. The council purchases emission permits from landholders when the estimated emissions exceed the target and sells emission permits when the target exceeds the estimated emissions. This approach seems feasible provided changes in the emission estimates are small relative to the total volume of emissions permits issued.

Revisions to the nutrient budget model would need to be made periodically to incorporate new information and the introduction of new technologies, practices and land uses. We envisage the model being updated at a fixed date possibly every

alternate year. At that date the emissions for each landholder would be revised in accordance with the schedule setting out the combinations and levels of inputs into agricultural production processes that they are permitted under their resource consents. Landholders with revised emissions in excess of their consents would be allowed a three month period of grace to reconcile the difference. They may either purchase or lease additional emission permits in order to continue their farming operations as specified in the original consent or they may adjust their farming operations appropriately and continue to generate emissions at the rate specified in the original consent.

Modifications to the nutrient budget model introduce an element of uncertainty into the market for both landholders and the council. This uncertainty, in principle, is a corollary of the uncertainty that would normally attend the variable and unpredictable supply of a natural resource flows. A number of possibilities may be considered that would allow market participants to manage their exposure to this uncertainty.

One possibility is to allow landholders to carry forward or bank any unused fraction of an emission permit. Landholders could draw down such reserves by producing emissions in excess of their permits in the future. Conversely, provision could be made to allow landholders to produce emissions in excess of their permits and to write of the excess against their permits in the future. Such a provision would require limits on the magnitude of the excess relative to the permit.

Other possibilities are to facilitate the development of insurance, futures and options. Insurance might be most appropriate in situations where the nutrient budget model is updated relatively infrequently and the likelihood of a substantial change in emission estimates is small. In this situation, landholders may use insurance to protect their income in the event of a dramatic change in their estimated emissions.

Futures and options may be most appropriate in situations where there are relatively frequent modifications to the nutrient budget model and these modifications usually result in relatively small changes in emission estimates. Futures allow landholders to limit their financial risk should they have to purchase additional permits following a change in emission estimates. Options allow landholders the right but not the obligation to purchase or sell emission permits for an agreed price should certain conditions be fulfilled at some point in the future. In principle, options can be designed to automatically activate the purchase or sale of a specific quantity of emission permits depending on the outcome of the modification of the nutrient budget model. Note that options may also apply to the leasing of permits.

The buyer of the option has the right or choice to exercise the agreement while the seller of an option must fulfil the conditions of the trade should the buyer choose to exercise the option. For example, in buying a call option the purchaser secures the right to purchase a nominated quantity of emission permits should the modification of the model lead to a change in their estimated emissions. The purchase of a call option also enables the purchaser to 'lock-in' the price at which the emission permits will be purchased. This may be an advantage if the market price of permits is highly volatile.

Finally the effect of time lags in the nutrient cycling in Taupo is important in considering the potential impact of any abatement activity on the operation of the market in emission permits. In principle, abatement activities could be undertaken that would intercept nitrogen in the groundwater before entering the Lake. Removing nitrogen from groundwater would, in principle, contribute to improving the quality of water in the Lake quality in the future though the quality of water in the Lake currently would be unaffected. Consequently, consideration should be given to compensating abatement activities that remove nitrogen from groundwater by means of issuing the equivalent in emission permits.

Nitrogen may also be extracted directly from Lake. This would improve the water quality in the Lake currently but we suspect that it may have a very limited impact on the quality of water in the Lake in the distant future. Consequently, abatement activities that remove nitrogen directly from the Lake would be unlikely to warrant compensation by means of issuing emission permits.

5.3 Monitoring compliance

Consistency requires that compliance with permit conditions would be evaluated in terms of the way in which inputs into agricultural production processes are specified in the nutrient budget model.

Monitoring may entail monthly, quarterly or annual self-reporting of inputs and management practices by landholders with random checks conducted under the auspices of the regional council. Minor infractions of consent conditions might be offset against the permitted rate of emissions in the following year. Note that the averaging of emissions over time and banking and write off provisions allow landholders some discretionary flexibility in the management of their properties in the short term.

5.4 Funding

The use of a nutrient budgeting model may mean that the establishment of a market may require some initial investment as well as recurrent expenditure on research and modelling. The initial investment costs may be met with public funds set aside for the protection of water quality in the Lake Taupo catchment. Recurring monitoring, research and modelling costs may be funded through a levy on permits.

Such a levy may be justified by appealing to the 'user pays' principle. However, if program costs were to be met in this fashion then the tax revenue should be clearly seen to be employed for this purpose. Otherwise, there is the risk that the levy will simply appear to be a mechanism for augmenting government revenues.

5.5 Penalties for illegal emissions

Landholders should be liable to pay a penalty for emissions in excess of their entitlement and the penalty for unauthorised emissions must be sufficient to deter landholders from illegally discharging on a systematic basis. We suggest that penalties are applied per unit of discharge in excess of the permitted level and that the excess be deducted from their emission allowance for the following year.

The determination of the penalty to be exacted per unit of emissions needs careful consideration since the penalty will effectively place a ceiling on the prices at which permits are traded. If the penalty is set too low then the price of permits may rise above the penalty. This means the penalty for discharging illegally will be lower than the cost of purchasing emission permits in order to discharge legally. Consequently, the demand for permits could fall to zero. Hence, if penalties are not set at sufficiently onerous levels the price of permits could be bid down to the level of the penalty. Since the price of permits cannot rise above the ceiling imposed by the penalty the market for emission permits will fail to function effectively.

Clearly, for the market to function without being inappropriately influenced by the penalty, the penalty must be set at a rate higher than highest marginal net benefit a landholder will gain from nitrogen emissions. In terms of conventional economic analysis this rate given by the intersection of the demand curve for emission permits with the price axis (Kaine and Reeve 1993).

The likelihood of prosecution for unauthorised emissions must also be sufficiently high to deter landholders from illegally discharging on a systematic basis. This means legislative support must be available which allows landholders can be successfully

prosecuted for breaching the terms of their discharge permits. If prosecutions for recovery of penalties and forfeiture of permits do not have a high probability of success then penalties can be effectively evaded. Establishing a market in emission permits is problematic in such circumstances as the right to discharge is no longer exclusive to those in possession of emission permits.

As a final point, penalties for illegal emissions could be employed as an additional, supplementary source of funding. However, revenue from this source may be better employed on public abatement projects or directed to general revenue. If revenues from fines and penalties were to be employed to fund the administration of the market some may take the view that these penalties are exacted for the purposes of raising revenue.

6. Issues in market establishment

In the discussion of allocation issues we noted that the distribution of gains and losses between stakeholders is determined by the initial allocation of permits among landholders in conjunction with the standard set for water quality. In that discussion the point was made that the relevance of many of the issues concerning the allocation of rights depends on the exact nature of the right under consideration, and the context within which rights are to be allocated. Having now described a market in emission permits in detail the issues entailed in allocating these permits will now be addressed specifically for the market we have described.

6.1 Distribution of adjustment costs

The setting of a standard for water quality is probably the most important determinant of the distribution of adjustment costs between landholders and other stakeholders and the distribution through time of those costs. As noted earlier, the limited control landholders can exert over emissions effectively prevents them from substantially adjusting their emission rate in the short term without making highly disruptive changes in their land management. Consequently, in practical terms the more rapidly the rate of emissions in aggregate is to be reduced in the short to medium term to achieve the desired standard of water quality, the greater the burden of adjustment will fall on landholders.

A case might be made that other stakeholders should at least partially compensate landholders for the adjustment costs incurred by the latter if a relatively rapid reduction in emissions is desired. Such compensation could be made in a variety of ways such as using public money to purchase and retire land from agricultural production as suggested by Environment Waikato (2003).

The initial allocation of permits is probably the most important determinant of the distribution of adjustment costs among landholders themselves. As noted earlier, the limited control landholders can exert over emissions effectively prevents them from substantially adjusting their emission rate in the short term without making highly disruptive changes in their land management. Consequently, the adjustment costs faced by landholders increases the smaller their allocation of emission permits relative to their historical rate of emissions. Hence, an allocation rule that gifts all landholders an equal allocation of emissions effectively transfers wealth from landholders with relatively high emissions to landholders with relatively low emissions. Landholders with high emissions are forced to cease operations, make disruptive changes in land use or purchase permits from landholders with low emissions in order to continue their activities.

In the context of Lake Taupo such a rule effectively transfers income from agricultural landholders to other landholders such as foresters. Such a rule also redistributes wealth among agricultural landholders. In our view the use of this type of allocation rule to compensate non-agricultural landowners for loss of future income seems inappropriate. If there is need for such compensation then presumably the community as a whole should bear the cost of that compensation.

6.2 Price discovery

In principle, a high volume of trading on the emissions market in the short term would promote price discovery which would facilitate decisions by all parties regarding the return to investing in abatement measures. In principle, the prices at which permits are exchanged in the market are a function of the differences in the marginal abatement costs faced by landholders and differences among landholders in the economic return to emissions. Provided there is some variation in economic returns to emissions to land holders and some variety among them in the cost of implementing abatement measures then trading should occur.

The greater the degree of variability among landholders in economic returns to emissions and the more greater the extent to which those returns change over time, the more likely a high volume of trading will occur in the short term. For example, the price

of agricultural products such as milk, lamb and beef are key determinants of the profitability of farming operations. Consequently, these prices are also key determinants of the economic value of emission permits. To the degree that these prices vary over time, both in absolute and relative terms, the economic value of an emission permit will differ across these enterprises. In theory this means the price that landholders are willing to pay in order to lease or purchase an additional permit will change over time in accord with changes in the economic returns to different land uses. Those landholders experiencing an increase in returns may seek to acquire additional permits to take advantage of the increase. Those landholders experiencing a decrease in returns may choose to lower production and sell or lease permits to offset their loss.

Similar arguments can be advanced to suggest the greater the degree of variability among landholders in terms of abatement costs the greater will be the opportunities for the exchange of permits among landholders. Much will depend on the flexibility with which landholders are able to modify their emissions by taking measures such as changing livestock numbers and types, altering grazing management practices and so on.

Relatedly, changes in the relationships between contextual characteristics and estimated emissions as embodied in the nutrient budget model will prompt the exchange of permits. As discussed earlier modifications to the nutrient budget model are likely to result in changes to estimates of landholders' emissions given the combination of agricultural inputs they are permitted through their resource consent. Differences among landholders in the opportunity costs of taking measures to reduce emissions and the price of emission permits are likely to encourage some landholders to dispose of a portion of their permits and to encourage other landholders to acquire permits. The greater the sensitivity of estimates of emissions to changes in the specification of the nutrient budget model the greater the volume of trading in permits changes to the model are likely to provoke.

The functioning of the price mechanism in the short term may also be influenced by the manner in which emission permits are allocated among landholders. In circumstances where landholders are allocated permits using a rule that results in their initial allocation of permits differing from estimates of their historical emissions, a high volume of trading may be expected in the short term. Allocating all landholders an equal volume of permits is an example of such a rule. While this encourages price discovery there are, as described earlier, significant equity implications arising from the wealth transfers entailed in such allocation rules.

Finally, price discovery will be influenced by the rules governing eligibility to enter the market and trade in emission permits. The more restricted the number of participants that may trade in the market the greater the degree of distortion in prices for emission permits. Ideally, market participation should be unrestricted if the market is to be contestable (Baumol 1982; Langridge and Sealey 2000; Paech 1998). This would mean individuals may purchase emission permits without being landholders but individuals who are landholders and are undertaking activities that discharge nitrogen must possess emission permits. Some stakeholders may not be comfortable with tradable emission permits unless the right to possess emission permits is restricted. See Schilizzi (2003) for a further discussion of the equity issues associated with market mechanisms.

6.3 Gifting of emission permits and windfall gains

The nature of the emission permits that have been proposed in this report is such that, whatever procedures are followed in allocating permits there will be an element of 'gifting' in the allocation procedure. Landholders that are allocated permits will, in effect, be freely granted the right to discharge nitrogen within the limits defined by those permits. We expressed the view earlier that this 'gifting' may be defended as a form of compensation for the curtailment of previously unlimited emission rights in exchange for the enforcement of the rights of others in water quality. We also expressed the view that it may be impractical as well as impolitic to compel landholders to purchase permits in this context.

Given then, that the allocation of permits would necessarily involve an element of 'gifting', at least some transactions in permits will involve an element of 'windfall' gains. In most cases, however, this windfall will represent only a fraction of the price at which permits are exchanged. For example, where a landholder sells permits the sale must be associated with a measurable reduction in emissions in order to proceed. Hence, for the landholder to realise a profit on the transaction, the revenue obtained from the sale of permits must exceed the cost of the measures undertaken to lower emissions. While such profits may be regarded as a windfall they also reflect the fact that relatively low cost measures are being employed to reduce emissions.

The size of such windfall profits will be limited by the operation of the market mechanism. Through trading the price of permits is equated with unit abatement costs thereby eliminating supernormal profits. If abatement is inexpensive then the demand for permits will be relatively low since emissions can be reduced at low cost. At the same time, the potential supply of permits for sale will be relatively high since landholders can

reduce emissions at relatively low cost thereby releasing surplus permits for sale. Thus, if there is ample scope for lowering emissions by implementing abatement measures that are relatively inexpensive, then the price at which permits will be exchanged in the market will be correspondingly low. Hence, the opportunities for landholders to secure windfall gains will be limited.

7. Prospects for market implementation

In assessing the feasibility for the market in nitrogen emission permits described in the previous chapters, it is important to distinguish between the prospects for establishing the market and its performance once established. In this chapter we review studies of the factors that influence the degree of success experienced in the establishment and operation of markets in resource rights. The implications of the findings of these studies for establishing a market in nitrogen emission permits in the Lake Taupo catchment are discussed.

7.1 The performance of markets in resource rights

Since the first suggestions of tradable permit systems for dealing with environmental problems by Crocker (1966) and Dales (1968) the concept has been proposed for a range of situations. Examples include control of air pollution (Montgomery 1972; Baumol and Oates 1975; Krupnick, Oates and Van de Berg 1983; McGartland and Oates 1985; McGartland 1988), global carbon dioxide levels (Dasgupta and Maler 1991; Hinchey, Thorpe and Fisher 1992), salinity (Hodge 1982; Quiggin 1986; Kaine, Musgrave, Burton and Bryant 1991); phosphorus discharges to waterways (David, Eheart, Joeres and David 1980), and other water pollutants (Eheart 1980; Brill, Eheart, Kshirsagar and Lence 1984; Lence, Eheart & Brill 1988). Much of the research effort in this area has gone into assessing, on theoretical grounds, the relative advantages and disadvantages of tradable permits compared to other policy instruments. Most studies have, as this report has done, pointed to the potential merits of markets in tradable permits in obtaining the most efficient allocation of scarce assimilative capacity for the least expenditure on abatement. These studies have also suggested that transaction and enforcement costs can be a serious impediment to the success of market-based instruments.

Rather fewer studies have reported the success or otherwise of tradable permit systems in practice. Hahn (1989) reviewed experience to that time with tradable permit systems for air pollution, lead in fuel, and for biological oxygen demand and phosphorus in discharges into waterways. While trading and the achievement of abatement costs savings had been substantial in the case of air pollution and lead in fuel, there had been little or no trading in the markets in biological oxygen demand and phosphorus permits. Hahn attributed this to the regulatory involvement in individual trades.

Atkinson and Tietenberg (1991), in reviewing a number of evaluations of the Emissions Trading Program in the USA found there was a general consensus that, while the Program had achieved significant abatement cost savings, it had not brought about a cost-effective allocation of abatement responsibility. Generally the cost savings and the amount of trading had been less than expected from theoretical analyses. Atkinson and Tietenberg (1991) attributed the inadequate performance to the constrained trading process which consisted of a series of bilateral sequential trades each of which was checked by the Environmental Protection Agency to make sure the trade did not result in ambient air quality levels being exceeded.

Higson and Kaine (2004) reviewed a number of examples of market based mechanisms in relation to water allocation, water quality, salinity, fisheries, biodiversity, air pollution and weather. In their view the key factor constraining the use of tradable permit schemes in natural resource management was the challenge of measuring the use of resources by diffuse sources. They found that the inability to measure the use of a resource means that the establishment of a direct link between economic activities and resource use is problematic. They believed possible solutions lay in the development of common property rights (Kaine and Reeve 1993) and the use of quantitative models to predict resource use.

The lessons here appear to be that markets in tradable emissions that involve diffuse sources are particularly difficult to create in the absence of an imputation mechanism such as a quantitative model and the operation of markets can be seriously hampered by regulatory constraints on trading.

7.2 Establishing markets in resource rights

In addition to the question of whether markets in tradable permits will function as theory suggests a very real issue is the question of the political feasibility of establishing a market. Riker and Sened (1991) argued that, while neo-classical economic theory provides an excellent framework for understanding and predicting the behaviour of

markets given particular property right structures, the theory does not predict when new property rights structures such as tradable emission permits can emerge. Blomquist (1992) in a study of institutions for governing the allocation of groundwater resources in southern California found that the arrangement of property rights and rules arrived at is more a consequence of what can be accommodated by stakeholders than of considerations of economic efficiency.

Riker and Sened (1991) suggest that four conditions are necessary to bring about the desire for political negotiation and cooperation among stakeholders that is essential if a new property rights structure is to be established. These conditions were that:

- The content of the right has to be sufficiently scarce that the value of the rights exceeds enforcement costs.
- Right holders desire the right.
- Government agencies recognise that the rights will be advantageous to them in pursuing their organisational goals despite the additional administrative costs imposed on them in overseeing and enforcing the new rights structure.
- Those upon whom the right imposes a duty acquiesce and adhere to their duties without undue coercion.

Libecap (1989) studied the history of the emergence of property rights associated with natural resources in the United States and identified a number of factors that favoured speedy development of new institutions for allocating resources. These were:

- There is widespread agreement among stakeholders that there is a need for new institutions to allocate the resources.
- There are a small number of stakeholders.
- All stakeholders expect to gain from the establishment of new institutions.
- Stakeholders are socially and culturally homogeneous.
- An institution can be devised that is consistent with existing institutional norms for allocating resources.
- There are no acute information uncertainties that prevent negotiating parties from reaching agreement in valuing shares of the resource.

Stavins and Whitehead (1992) reviewed experience in the United States with a range of environmental policy instruments. They concluded that four political forces have worked to inhibit the implementation of market based instruments. The forces they identified were:

- The adversarial nature of the early phases of environmentalism in the 1960s and 1970s led many to view tradable permits as licences to pollute or to see permits as legitimising pollution.
- Some environmental agencies have seen tradable permits as a threat to them because of perceptions that many of the monitoring and evaluation activities performed by the agencies might no longer be required.
- The regulatory approach has often been characterised by protracted negotiation between industry, environmental groups and legislators. The lobbyists involved have made considerable investments in gaining the expertise and contacts needed to obtain favourable outcomes for their clients. Consequently, lobbyists generally see market based instruments as a threat to their vocation.
- Some environmentalists fear that market based instruments will make the costs of environmental protection too salient thereby dampening demand for such protection.

Ostrom (1990) investigated nineteen case studies of common property resource management. They identified a number of factors associated with the successful establishment of common property regimes of which two also apply to the development of markets in tradable resource permits in our opinion. These were that the rules for use of the resource were appropriate to local conditions and that there was provision for stakeholders to participate in any process for modifying those rules.

In summary, these studies indicate that five fundamental preconditions will determine whether a market in tradable emission permits might be implemented successfully. These are that the emission rights are widely recognised among stakeholders as a scarce and valuable resource, there is widespread agreement among stakeholders that there is a need for new institutions to allocate emission rights, that a market mechanism fits with existing community and institutional norms for allocating resources, that the rules governing the operation of the market are appropriate for local conditions, and that stakeholders participate in the processes for changing those rules. In our view most, though perhaps not all of these preconditions, have been met with respect of Lake Taupo.

There appears to be widespread awareness and acceptance amongst all stakeholders including landholders that nitrogen emissions need to be limited to protect water quality in Lake Taupo. There also appears to be widespread awareness and acceptance amongst stakeholders that restricting nitrogen emissions will impose significant economic costs on landholders and, as a consequence, nitrogen emissions are scarce and valuable resource which should be used to the maximum benefit of the community.

Generally speaking, there appears to be agreement among stakeholders that there is a need for a new institutional mechanism to allocate emission rights but there is disagreement among stakeholders as to the type of mechanism that should be implemented. Some stakeholders favour a market mechanism while others such as landholders appear to favour more of a regulatory approach involving voluntary industry codes practice. Although these differences probably reflect sectional interests to some degree we believe that a lack of familiarity among some stakeholders with the use of markets to manage rights to natural resources may also be a contributing factor.

The regional council and other institutional stakeholders are well aware that adversarial conflicts over resource allocation can be costly and damaging for those involved and there is some merit in using a market for allocation. The role of the regional council in ensuring the preservation of water quality in Lake Taupo remains essentially the same under a tradable permit system compared to a more regulatory approach based on standards or an industry code of conduct. Furthermore, the council already undertakes many of the activities that are involved in the operation of a market in emission permits such as monitoring compliance behaviour, estimation of emissions, and the granting of consents. The council would be required to undertake these activities whatever approach was taken to managing emissions. Consequently, we believe that a system of tradable emission permits is reasonably consistent with institutional behaviours and that the regional council and relevant government departments believe such a system could assist them to achieve their goals.

We believe the principles underlying the rules governing the market in emission permits that we have described suit the conditions in Taupo. Given that measurement of nitrogen emissions is problematic and that nutrient cycling in Lake Taupo occurs over decades the use of a nutrient budget model to estimate emissions seems the only practical way of assessing nitrogen emissions from rural land, both on an individual basis and in aggregate. This means that irrespective of the approach taken to distributing emission rights among landholders (market, regulatory or voluntary codes of conduct) key issues for stakeholders to consider are the conditions that initiate a change in the knowledge embodied in the nutrient budget model, the processes for initiating change in the model, and the nature of stakeholders' participation in those processes.

In conclusion, we believe there is a distinct possibility that a market in nitrogen emissions could be established for Lake Taupo. Much depends on the degree to which sectional groups among landholders can be reassured that a market in emissions offers them adequate protection of their interests.

8. Conclusion

Our aim in preparing this report was to constructively contribute to the discussions and debate concerning policies for managing nitrogen emissions from agricultural land in the Lake Taupo catchment.

In the report we discussed some of the policy options that have been proposed for managing nitrogen emissions from agricultural land in the Lake Taupo district. Our main focus was on developing a proposal for a market in nitrogen emission permits.

We believe that the type of market in emission permits that we have described in this report has the potential to enable community aspirations with respect to the quality of water in Lake Taupo to be achieved while avoiding disruptive change and preserving the managerial flexibility and autonomy of landholders.

9. References

Abler, D.C. and Shortle, J.S. 1991. 'The political economy of water quality protection from agricultural chemicals'. *Northeastern Journal of Agricultural and Resource Economics,* 20(1): 53-60.

Antonelli, C., 2004. 'Localized product innovation: the role of proximity in the Lancastrian product space.' *Information Economics and Policy*, 16 (2), June: 255-274.

Atkinson, S. and Tietenberg, T. 1991. 'Market failure in incentive-based regulation: The case of emissions trading'. *Journal of Environmental Economics and Management,* 21: 17-31.

Baumol, W. J. 1982. 'An uprising in the theory of industry structure'. *American Economic Review,* 72: 1-15.

Baumol, W.J. and Oates, W.E. 1975. *The Theory of Environmental Policy,* Cambridge University Press, Cambridge.

Blomquist, W. 1992. *They Prefer Chaos: Institutions for Governing Groundwater Systems in Southern California.* Institute for Contemporary Studies Press, San Francisco.

Bohm, P. and Russell, C.S. 1985. 'Comparative analysis of alternative policy instruments'. In: Kneese, A.V. and Sweeney, J.L. (eds) *Handbook of Natural Resource and Energy Economics.* Elsevier Science Publishers, 395-460.

Bradsen, J.R. 1988. *Soil Conservation Legislation in Australia.* Report for the National Soil Conservation Program, University of Adelaide, Adelaide.

Brill, E.D., Eheart, J.W., Kshirsagar, S.R. and Lence, B.J. 1984. 'Water quality impacts of biochemical oxygen demand under transferable discharge permit programs'. *Water Resources Research,* 20(4): 445-455

Bromley, D.W. 1991. *Environment and economy: property rights and public policy.* Basil Blackwell Ltd, Oxford.

Coase, R. 1960. 'The problem of social cost'. *Journal of Law and Economics,* Oct. 1960. 1-44.

Crocker, T.D. 1966. 'The structuring of atmospheric pollution control systems'. In: Wolozin, H. (ed) *The Economics of Air Pollution.* W.W. Norton, New York.

Dales, J.H. 1968. *Pollution, Property and Prices.* University of Toronto Press, Toronto.

Dasgupta, P. and Maler, K-C. 1991. 'The environment and emerging development issues'. *Proceedings of the World Bank Annual Conference on Development Economics,* 1990: 101-131.

David, M., Eheart, W., Joeres, E. and David, E. 1980. 'Marketable permits for the control of phosphorus effluent into Lake Michigan'. *Water Resources Research,* 16(2): 263-270.

Eheart, J.W. 1980. 'Cost-efficiency of transferable discharge permits for the control of BOD discharges. *Water Resources Research,* 16(6): 980-986

Environment Waikato, 2003. *Protecting Lake Taupo: A Long Term Strategic Partnership,* Environment Waikato, Hamilton, New Zealand.

Falconer, K. 1998. 'Managing diffuse environmental contamination from agricultural pesticides: An economic perspective on issues and policy options, with particular reference to Europe.' *Agriculture, Ecosystems and Environment*, *69*: (37-54).

Glasbergen, P. 1992. 'Seven steps toward an instrumentation theory for environmental policy'. *Policy and Politics,* 20(3): 191-200.

Guerin, K. 2003. 'Property rights and environmental policy: A New Zealand perspective.' *New Zealand Treasury Working Paper: 03/02*, Wellington.

Hahn, R.W. 1989. *A Primer on Environmental Policy Design.* Harwood Academic Publishers, Chur, Switzerland.

Higson, M. and Kaine, G. 2004. Tradable Permit Systems for Natural Resource and Environmental Management, *Social Research Working Paper 03/04*, AgResearch, Hamilton, New Zealand.

Hinchey, M.D., Thorpe, S.J. and Fisher, B.S. 1992. 'Quantifying a tradable emissions scheme'. *International Association of Energy Economists Conference,* Tours, France, 18-20 May. ABARE Conference Paper 92.13.

Hodge, I. 1982. 'Rights to cleared land and the control of dryland seepage salinity'. *Australian Journal of Agricultural Economics,* 26: 185-201.

Hoehn, J. P., Frank L., and Kaplowitz, M. D., 2003. 'Untying a Lancastrian bundle: valuing ecosystems and ecosystem services for wetland mitigation**.'** *Journal of Environmental Management,* 68 (3), July: 263-272.

Jacobs, J.J. and Casler, G.L. 1979. Internalising externalities of phosphorus discharges from crop production to surface water: Effluent taxes versus uniform reductions. *American Journal of Agricultural Economics,* 61(2): 309-312.

Kaine, C., Musgrave, W., Burton, J. and Bryant, M. 1991. 'Towards introducing markets for riverine resources'. Paper presented at the *3rd Annual Ministerial Water Forum,* New South Wales Department of Water Resources, Sydney, July, 1991.

Kaine, G. and Reeve, I. 1993. *A Proposal for a Market in Phosphorus Discharge Permits.* Report to the NSW Department of Water Resources, The Rural Development Centre, UNE, Armidale.

Krupnick, A., Oates, W. and Van de Berg, E. 1983. 'On marketable air-pollution permits: The case for a system of pollution offsets'. *Journal of Environmental Economics and Management,* 10: 233-247.

Lancaster, K.,1966. 'New approach to consumer theory'. *Journal of Political Economy*, 74 (2), April 1966: 132-157.

Lancaster, K., 1971. *Consumer Demand: A New Approach*. Columbia University Press, New York.

Langridge, R. and Sealey, R. 2000 'Contestability in the UK bus industry? The National Bus Company and the `Tilling Mark II' effect'. *Transport Policy,* 7: 105-115.

Lence, B.J., Eheart, J.W. and Brill, E.D. 1988. 'Cost efficiency of transferable discharge permit markets for control of multiple pollutants'. *Water Resources Research,* 24(7): 897-905.

Libecap, G.D. 1989. *Contracting for Property Rights.* Cambridge University Press, Cambridge.

McGartland, A. 1988. 'A comparison of two marketable discharge permit systems'. *Journal of Environmental Economics and Management,* 15: 35-44

McGartland, A.M. and Oates, W.E. 1985. 'Marketable permits for the prevention of environmental deterioration'. *Journal of Environmental Economics and Management,*12: 207-208

Milne, P., and Mooar, J., 2002. *Water Allocation and Sustainability*. Report prepared for MAF Policy, Simpson Grierson, Wellington.

Montgomery, W.D. 1972. 'Markets in licenses and efficient pollution control programs'. *Journal of Economic Theory,* 5: 395-418.

Nicolaisen, J., Dean, A. and Hoeller, P. 1991. 'Economics and the environment: a survey of issues and policy options'. *OECD Economic Studies,* Spring: 8-38.

Ostrom, E. 1990. *Governing the Commons. The Evolution of Institutions for Collective Action.* Cambridge University Press, Cambridge.

Paech, N.P. 1998. 'Contestability reconsidered: the meaning of market exit costs'. *Journal of Economic Behaviour and Organization,* 34: 435-443.

Parminter, T. 2003. 'Policy strategies for natural resource management.' *MAF Technical Paper No: 2003/1*.

Power, I., Ledgard, S., and Monaghan, R., 2002. 'Nutrient budgets for three mixed farming catchments in New Zealand'. MAF Technical Paper No: 2002/17, Wellington.

Quiggin, J. 1986. 'Common property, private property and regulation: The case of dryland salinity'. *Australian Journal of Agricultural Economics,* 30: 103-117.

Riker, W.H. and Sened, I. 1991. 'A political theory of the origin of property rights'. *American Journal of Political Science,* 35(4): 951-969.

Rae, J. 1997. 'Economic instruments for environmental protection.' *Environmental Economics Research Paper No. 6*, Environment Australia,Canberra.

Randall, A. 2003. 'Market based instruments: International patterns of adoption, remaining challenges and emerging approaches.' *6th AARES Annual National Symposium on Market Based Tools for Environmental Management*. Canberra, September.

Schilizzi, S. 2003. 'Should equity concerns impose limits on the use of market based instruments?' *6th AARES Annual National Symposium on Market Based Tools for Environmental Management*. Canberra, September.

Scrimgeour, F., and Piddington, K. 2002. 'Environmental taxation in New Zealand: What place does it have?', *IPS Policy Paper: 12/2002.*

Sharp, B. 2002. 'Institutions and decision making for sustainable development.' *New Zealand Treasury Working Paper: 02/20*, Wellington.

Stavins, R. 2000. 'Experience with market-based environmental policy instruments'. *Resources for the Future: Discussion Paper 00-09.*

Stavins, R.N. and Whitehead, B.W. 1992. 'Dealing with pollution'. *Environment,* 34(7): 7-11 & 29-42.

Weingarten, P. 2001. 'Tradable pollution permits: A useful instrument for agri-environmental policy?' ACE seminar on Environmental Effects of Transition and Needs for Change, September 10-16, Slovakia.

Wheeler, D. M., Ledgard, S. F., deKlein, C. A. M., Monaghan, R. M., Carey, P. L., McDowell, R. W., and Johns, K. L. 2003 'OVERSEER® nutrient budgets – moving towards on-farm resource accounting'. Proceedings of the New Zealand Grassland Association 65: 191-194.

Wilkinson, C.F. 1987. 'Soil conservationists and the uses of the law'. *Journal of Soil and Water Conservation,* 42(5): 304-311.