CSIRO

CSIRO LAND and WATER

Market-Based Instruments for Managing Water Quality in New Zealand

Darla Hatton MacDonald[1], Jeff Connor[1] and Mark Morrison[2]

[1]Policy and Economic Research Unit, CSIRO Land and Water

[2]Charles Sturt University

FINAL REPORT for NZ Ministry for the Environment

January 2004

Folio No: S/03/1393

Cover Photo: Kayaking - Lake Taupo, Maori Rock Carvings
Credit: Destination Lake Taupo www.laketauponz.com

EXECUTIVE SUMMARY

Market-based instruments (MBIs) have been described as promising tools for advancing sustainable development in New Zealand (Sinner and Salmon, 2003). Sometimes described as "economic instruments", these instruments seek to bring market opportunities and processes into areas that have been traditionally controlled by direct regulation, information and motivational processes.

In the area of water quality, MBIs hold potential to speed adjustment towards optimal outcomes and stretch available resources. But there is devil in the detail. To be effective they must be well designed and, in particular, there must be adequate investment in science and in the administrative arrangements necessary to make them work.

Often these instruments need also to be supported by the development of standards, regulation and other instruments. MBIs are rarely a substitute for regulatory and other approaches. They just make it possible to achieve desired environmental outcomes at less cost. They do this by creating incentives for firms and households to act in the interests of society. For those who have an impact on water resources - whether industries, farmers or households - MBIs allow choice and may result in cost savings when compared with strict prescriptive solutions.

<u>Types of instrument</u>

This report focuses on opportunities. At the end, it recommends the introduction of a range of pilot projects which, by trialling the use of MBIs, would develop expertise and experience in NZ. This approach would also reduce the risk of failure.

The report offers a framework for the assessment of alternative instruments that will assist decision makers to identify which instruments can most appropriately be applied to a particular water quality problem. Sometimes, instruments are most appropriately used in concert with one another. Movement from theory to practice requires being able to identify the prerequisite conditions necessary for their application.

MBIs work either by changing prices or by limiting the quantity of an environmental resource that may be used. Thus, the suite of instruments available can be classified easily as:

- Price-based instruments and
- Quantity-based instruments.

To facilitate the work undertaken in this report and ensure that the report focused on opportunities relevant to NZ, a particular set of instruments were selected by the Steering Committee for consideration in this report:

- Price-based
    - Environmental charges,
    - Tendering,
    - Compensated covenants,
    - Land leasing,
- Quantity-based
    - Transferable permits, and
    - Environmental offsets.

A short description of each of these instruments is provided in the second section of the report. The emphasis is on instruments that are currently not widely used in NZ. NZ has developed some innovative approaches with respect to covenants and has important experience that could be extended to other countries.

<u>Instrument choice</u>

Choosing the most appropriate instrument (or mix of instruments) is difficult and often context specific. The third section of the report provides a framework designed to simplify this process. It focuses on the attributes and conditions needed for an instrument to work. An efficient screening process is offered. Environmental processes of concern, monitoring technology, market conditions and administrative arrangements all have to be considered. The range of values held by different cultural groups is also an important factor to consider. When, for example, these groups have different risk profiles, they can be expected to respond to the introduction of the same instrument in a different manner.

The screening process offered provides a sufficient base of information to answer these key questions:

- Which of the variants of MBIs are feasible in the existing institutional settings?
- What is the potential of different MBIs to ensure environmental quality goals are met?

- What is the potential of different MBIs to reduce the cost of meeting environmental policy objectives?

The result of applying this screening process may not be a definitive identification of "the" MBI for a given problem. Rather, for any particular water quality problem, use of this screening process should result in:

- Identification of the types of instruments that are feasible given the current institutional setting, and the range of institutional changes that might be required;
- Identification of the environmental process, institutional setting, market characteristics, technology and policy design attributes that favour or impede reliable and cost-effective realisation of natural resource management (NRM) policy goals.

Sometimes, new administrative processes or new regulations must be introduced before a market-based instrument can be introduced.

Non-market valuation

The Terms of Reference for this report also required a review of the role of non-market valuation in the use of MBIs. Essentially, non-market valuation offers a way to understand the degree to which market instruments should be used. It can also assist with the preparation of cost-benefit analyses designed to advise whether or not the benefits from using an instrument are likely to be greater than the costs of doing so.

Application to New Zealand

Finally to place these instruments in a NZ context, two case studies are presented. The first focuses on diffuse source nutrient pollution associated with agricultural drainage into groundwater which then discharges into freshwater lakes. The second focuses on stream and estuary sedimentation as a result of surface runoff from urban development and forestry.

The case studies provide a focus for the development of recommendations for implementation. It is inevitable that there will be some political aversion to new policies like higher environmental standards, differentiated charge rates based on outcome or input use levels, and additional development restrictions. Such aversion does not have to stifle change. Standards that limit how property can be used have been introduced in the past and are now widely accepted. For example, the right to sell farm products are typically conditional on meeting certain food safety and trade related restrictions on production.

The case studies lead to the conclusion that :

- Tendering could be used to deliver more efficient incentive payments. This would involve soliciting bids to take actions to reduce nitrate or sediment loading and choosing among bids based on levels of measurable factors correlated with the outcome of interest (e.g. prioritisation of bids to reforest based on slope and proximity to vulnerable water bodies).
- Offset approaches could be implemented by setting limits that preclude future conversion of land to pasture to control nitrate loading or setting limits on logging or urban development to control sediment loading. Then some level of actions such as livestock farming intensification, urban development or forestry could be allowed where compensating mitigation is provided based on formulas that would guarantee a "net" decrease in nitrate or sediment load.
- Tradeable permits could be applied to either nitrate or sediment loading by placing a cap on the level of nitrate or sediment load allowed from each source. Alternatively, a cap could be placed on levels of allowable input use for some input correlated with nitrate or sediment load. Those able to reduce loading or input use below their cap level would be able to sell permits.

Successful introduction of change can be facilitated with several strategies. One useful approach involves a transition to higher standards with an initial period of reduced compliance burden, followed by a period of gradually increasing standards and penalties for non-compliance. The approach can reduce the perceived threat of changes by allowing individuals to gain some degree of comfort with the production modification required before there are serious sanctions for non-compliance.

Issues around the development of institutional arrangements, such as standards, were identified as a factor that limits the use of MBIs. Several mechanisms could be used to create more effective standards that could then be used to limit activities that are the source of diffuse source water quality externalities. For example:

- Discharges from diffuse sources could in principle be regulated by local councils under the resource consents process of the Resource Management Act (RMA) through requirements for mitigation under defined conditions.
- Development of catchment-based water plans could be required that provide standards for environmental outcomes and best-practice performance. This approach was intended in the original RMA enabling legislation but has only been pursued by three Councils.

- Either of these above listed approaches could in turn be augmented by offset or tradeable permit approaches for agricultural and forestry diffuse source water quality issues.
- More fundamentally, changes in standards relating to activities influencing water quality are possible at the national government level under section 43 of the RMA. NZ is currently in the process of developing standards for air quality and landfill gas under these provisions and it would be possible to develop standards for water quality in a similar way.
- Another fundamental change in the RMA worth considering would involve clarifying the ambiguity between provisions for continued land use and intensity at current levels (provisions 10 and 20a) and provisions prohibiting discharges with demonstrably significant adverse effects. This could enable development of more effective standards and support MBI development.
- In peri-urban settings where sediment loading from development is an issue, existing development laws generally allow for the development of controls that limit activities impacting sediment loading. In such settings, further development of such controls could represent an effective standard on which quantity-based MBIs of the offset type can be built.

This report outlined how a number of attributes of biophysical processes and markets can limit potential for cost saving and/or environmental reliability of MBIs. One key finding was that the difficulty in measuring actual outcomes of interest for diffuse source water quality issues has made application of MBIs particularly challenging. There are essentially three ways to overcome the "monitoring problem":

- Focus on practices that are correlated with the outcome of interest;
- Focus on an input which is correlated with the outcome of interest – rather than the outcome of interest *per se*; or
- Setting standards or charges, or incentive payments on some easily measurable input or practice, and then using zones and other locational arrangements to correlate them with the outcome of interest.

Focussing on practices is not recommended for NZ. While the approach has often been chosen elsewhere for ease of implementation and environmental reliability, it has often failed to produce the anticipated cost savings because the flexibility of the landowners or developers is often curtailed.

An additional strategy worth considering in NZ is implementation of a series of trial or pilot projects designed to test feasibility and build administrative experience. Typically, such trials would be of limited duration and limited geographic extent.

Experience in Australia suggests that the approach could represent an effective, and politically feasible way to gain an understanding of MBI design and how implementation influences environmental and cost effectiveness. An advantage is that best practice for NZ can be developed using pilots before new policies are implemented on a larger scale.

ACKNOWLEDGEMENTS

The authors would like to acknowledge the assistance provided by Matthew Hickman (NZ Ministry for the Environment), Chris Richmond (NZ Department of Conservation) and Gerard Horgan (NZ Ministry of Agriculture and Forestry). Thank you for patience and guidance in finishing this report.

The evaluative framework developed in this paper is based in part on earlier work completed by Melissa Bright, Doug Young, Amanda Hamilton, Jeff Connor and Mike Young. We have extended this original work in a number of ways but the genesis of some of these ideas can be traced back to the work of Bright *et al*, 2002.

TABLE OF CONTENTS

1 INTRODUCTION

Market-based instruments (MBIs) have been receiving a lot of attention as the way forward for environmental management and have been described as promising tools for advancing sustainable development in New Zealand (Sinner and Salmon, 2003). Sometimes described as "economic instruments", these instruments seek to bring market opportunities and processes into areas that have been traditionally controlled by direct regulation, information and motivational processes. In the area of water quality, MBIs hold potential but the tools are not without cost to government. MBIs can be described as levers for natural resource managers to use to change behaviour of polluters (point source and non-point source) so that they are aligned with the government's goals and objectives for resource management. MBIs are basically market mechanisms that are designed to address context specific problems of environmental quality.

This report was commissioned by the NZ government to summarise the current state of knowledge concerning the use of MBIs for managing diffuse source contamination of water quality. The terms of reference specified that the report should cover:

1. Identification of MBIs applicable to non-point sources of water pollution. These will include fees/charges/resource rentals, taxes and differential rating, subsidies, tradable permits, land leasing and compensated covenants to restrict land use activities.
2. Pros and cons of the identified MBIs with regard to the NZ context.
3. Description of national (if available) and international examples of the uses of MBIs specifically to address diffuse sources of water pollution.
4. How the use of non-market valuation can inform the use of MBIs in addressing diffuse sources of water pollution.
5. Using examples of a range of pollutants (including nitrate, phosphate, pesticides and other toxic contaminants, BoD and suspendable solids) identify the most appropriate MBI in each case.
6. Recommendations regarding the feasibility of the use of MBIs in relation to diffuse sources of water pollution in NZ.

The project team and the steering committee worked to address these points through the use of examples and specific case studies. The report outlines the ways that markets and the overall institutional relationships between producers, consumers and governments can be aligned to address water quality issues in NZ including:

- The opportunities that exist immediately and the opportunities that may be possible over the longer term with MBIs and markets more generally in NZ;
- What has come to be understood from international experience about how instrument performance is determined by biophysical processes, institutions and the nature of the economic agents; and
- The characteristics of biophysical and institutional characteristics that are likely to be critical for the design of MBIs for diffuse source water quality issues in NZ.

There is no single instrument for solving water quality issues for NZ or any other jurisdiction. The effectiveness of any given instrument will depend on the problem faced, the context of the problem, the details of instrument design and existence of complementary or competing policies and incentives. The appeal of well-designed MBIs is the potential for providing flexibility for individuals and firms as well as enhancing the ability of institutions to contend with change.

The innovative contribution of this report is the development of a practical framework for evaluating potential instruments that takes account of the instruments, the biophysical processes, people and communities and the characteristics of the institutional setting. Selecting among potential MBIs requires natural resource managers to draw on their knowledge and this report offers a framework for sorting through the choices.

1.1 Background

NZ has an extensive network of high-quality fresh water systems. Water quality in NZ, although generally high by world standards, is variable. Land-use intensification poses a threat for the multiple uses of freshwater supplies including water for the environment, drinking water supplies, recreational users and stock water drinking supplies.

The NZ government is embarking on a Programme of Action to ensure that principles of sustainable development underpin all government activity and that government decisions take account of the well-being of current and future generations. The Programme of Action establishes a set of operating principles for policy development that require government to take account of the economic, social, environmental, and cultural consequences of its decisions. The focus is on the practical application of the sustainable development approach to certain key issues including the quality and allocation of freshwater.

The Ministry for the Environment and the Ministry of Agriculture and Forestry are leading work on water quality, water allocation and waterbody protection in association with other government departments. The focus for the water quality project is rural land use impacts on waterways and water quality; the economics component of this project focuses on the use of MBIs to manage the cumulative effects of diffuse sources of pollution from intensified rural land use. This report was commissioned as part of the water quality project.

1.2 The Role of MBIs

Traditionally, the main tools of natural resource and environmental managers were thought to be regulation, monitoring and enforcement of penalties. This approach has been called command and control and the terminology suggests an approach that asserts more control than often occurs in reality. For instance, international experience suggests that agencies are often under-resourced, and hence do not have the capacity to undertake the extensive monitoring required to implement command and control environmental regulation (Randall, 2003). Considerable resources are required to prove non-compliance that will stand up to court challenges. There is also the general problem of obtaining information from polluters (known as the asymmetric information problem) or there may be a lack of political will to impose penalties at levels that will induce firms to alter their behaviour.

Regulatory agencies may take the more pragmatic approach of setting out regulations and negotiating with firms to find suitable paths to compliance. This more pragmatic approach has been dubbed responsive regulation (Ayres and Braithwaite, 1992) and by its nature is an adaptive form of governance. It is worthwhile highlighting that this form of environmental governance is gaining credence and offers another tool for environmental management.

The development of many MBIs and the more adaptive forms of governance, has resulted from the recognition that strict prescriptive regulation may impose too many costs on both government agencies and the firms being regulated. Theoretically, MBIs potentially offer a way to introduce more flexibility and thereby reduce the costs associated with achieving environmental outcomes.

However, MBIs are not without their own hazards and are not substitutes for regulation and standards. In fact, MBIs often rely on regulation and standards. Certainly, there are no one-size-fits-all solutions to ensure environmental quality, as markets, on their own, are unlikely to produce uniformly socially optimal outcomes. Moreover, many environmental problems may not be suited to the use of MBIs.

Often a mixture of regulation and MBIs is required to achieve appropriate levels of environmental quality at minimum cost.

1.3 Outline of the Report

One of the main purposes of this report is to describe the potential role for MBIs and provide decision makers with a framework that enables them to select the most appropriate instruments to use in a specific circumstance. Moving from theory to practice requires being able to describe the prerequisites for implementing a particular MBI. To facilitate this process, a particular set of instruments was selected by the Steering Committee for consideration in this report. A short description of each of these instruments is provided in the second section of the report. The emphasis is on instruments which are currently not widely used in NZ as this is where the additional gains in social welfare are likely to occur. NZ has developed some innovative approaches with compensated covenants and has a few lessons to extend to other countries.

The report then describes an evaluative framework for sorting through the potential options in terms of instruments that are available in the short term and which instruments will require institutional change and therefore could only be implemented over time. Within this framework, the relative cost-effectiveness of the instruments can be determined. This is done by considering the attributes of the environmental process, the technology for monitoring and the nature of the market. In addition, the nature of the groups that are potentially being targeted is also an important factor to consider. These groups can potentially represent different segments of the market with different risk profiles, for instance, that need to be taken into account.

The terms of reference for this report also required a review of the role of non-market valuation. Non-market valuation is an important issue in the determination of optimal strategies for environmental management, including the setting of standards and estimating damage costs for environmental charges.

Finally to place these instruments in a NZ context, two case studies are presented where:

- Diffuse source agricultural drainage rich in nutrients enters groundwater and then discharges to freshwater lakes; and

- Sedimentation occurs in streams and estuaries as a result of surface runoff from urban development and forestry.

2 OVERVIEW

MBIs are a broad class of mechanisms that seek to align private incentives with public natural resource management (NRM) goals. Expectations regarding what can be achieved using MBIs should be dependent on how well the instrument has been tailored for the circumstances in which it is to be applied. Ideally,MBIs will:

- Achieve a given level of environmental improvement compared with a prescriptive regulatory approach for a lower cost; or
- Create markets for environmental outcomes.

Inclusion of MBIs in the policies used for NRM is underway or being considered in many OECD countries including NZ, Australia, Canada and the United States (Stavins, 2000). The approaches being used or proposed fall into several distinct categories. Most instruments involve modifications to existing "non market-based" policy to include market mechanisms. One example is using a tendering approach or an auction process to distribute public funds. Another example is tradeable permits which involve modifying regulation to allow for buying and selling permits.

2.1 What are MBIs?

A series of names are currently being used in the economic and policy literature to describe MBIs (Hahn 1989; Sterner 2003; Randall 2003) and these include:

- Economic instruments,
- Market based instruments (MBIs),
- Incentive based policy,
- Policy instruments, and
- Flexible instruments.

The term, market-based instruments (MBIs), has become the standard terminology in countries such as Australia. Changing the language and abandoning the term "economic instruments" was important in shifting the policy debate from an approach advocated by economists and business people to an approach that included a wider circle such as natural resource managers, planners and regulators.

A typology for categorising MBIs is presented in Text Box 1. MBIs rely on a well-defined regulatory framework, and in some cases property rights and entitlement systems, in order to send market based signals to change behaviour for the benefit of the environment.

At the outset it is important to emphasise that MBIs are not a “free market” substitute for all NRM policy approaches such as command and control, but rather MBIs involve modifying traditional policy approaches to include market mechanisms. The need to investigate and set goals and targets for environmental quality either remains the same or increases. Similarly, the need for research to understand the biophysical processes remains the same or may increase for NRM agencies that wish to utilise MBIs.

An attractive feature of MBIs in NRM policy is that the approach can align private incentives for profit with public goals of improving environmental quality. Often individuals and firms have detailed knowledge of the opportunities that exist for them to improve their interaction with their environment compared with a regulator. Policies with incentives related to NRM outcomes reward individuals who use this detailed and specialised knowledge to reduce the environmental impacts of their activities in innovative, low cost ways. For example, an irrigator facing a charge related to water use may be motivated to implement low cost ways of reducing water use. It is important to acknowledge that there are some significant constraints limiting when MBIs can be effectively applied. For example, some MBIs require the development or refinement of property rights, monitoring capacity, reporting and accounting systems. If the initial investment required to develop such systems is large, or the administrative cost in running them is large, the benefits of these MBIs may not be sufficient to offset their costs.

Some MBIs require changes to property right regimes that may not be politically feasible, implying that it is not possible to utilise certain MBIs. The attitudes of individuals and businesses toward some of the MBIs available may preclude their use in some catchments. In any of these situations, it may be necessary to choose from a reduced set of MBIs and/or rely on traditional command and control approaches to environmental management. It should be noted that what is perceived as politically acceptable evolves over time and can be influenced through policy designed to inform and influence.

The remainder of this section outlines the major types of MBIs that will be considered for use in managing diffuse source environmental water quality issues.

Text Box 1: Environmental policy instrument typology

Non-market based instruments:

Output or performance based standards – this type of instrument involves setting limits on performance or output (e.g. limits on effluent load or concentration).

Input, practice or process based standards – these instruments can involve setting limits on input level, specifying that a particular technology be used in production (technology or best management practice requirements) or development and zoning regulations.

Education, moral suasion – these instruments seek to influence behaviour in ways that improve environmental outcomes of interest by educating those who create externalities about public or private benefits of reducing externalities.

MBIs:

Price-based instruments – are instruments that attempt to influence environmental performance by pricing negative externalities or subsidising mitigation actions. There are several variants including:

Environmental charges – charges with the rate related to the level of an environmental externality (e.g. discharge fees for effluent). Alternative implementations can involve charges on inputs related to an externality (a charge for vehicle registration with rate based on engine displacement as a proxy for a discharge fee).

Incentive payments – involve subsidising the cost of actions to mitigate an externality. Often, incentive payment levels are set at fixed rates.

Tendering – is an alternative approach to distributing incentive payments that involves distributing funds by tender or auction. This involves those seeking incentive payments making offers describing mitigation action and cost sharing payment terms. The Government selects among offers based on value of mitigation per cost sharing dollar.

Quantity-based instruments – involve setting standards for mitigation effort (e.g. emissions standards) and allowing trade among those providing mitigation (allowing individual underperformance if it is compensated by over performance elsewhere). There are two major variants:

Tradeable permits – involve setting individual rights to input levels, output levels or performance standards (e.g. individuals are granted an allowable level of emissions as a number of emissions permits). Individuals are then only allowed to exceed the standard if they purchase additional permits from someone who is under their allowable emissions and therefore has excess permits.

Environmental Offsets - environmental offsets are actions taken to meet a standard (reducing pollution or environmental impacts) at a site away from where the action causing an environmental externality occurs. The party causing the externality can either take the action themselves or pay for others to do it on their behalf.

Market barrier elimination instruments – focus on improving environmental outcomes by increasing consumer awareness of environmental attributes of products they may value, or removing barriers to market activity. Product labeling schemes are perhaps the most widely applied market creation MBI approach. They involve providing information about the environmental outcomes of production so that those who value associated improved environmental outcomes can express their preferences through markets. Source: Connor and Bright (2003).

A basic description of each type of instrument is provided and examples of applications that are particularly relevant are outlined. The discussion focuses on:

- Environmental charges,
- Tendering,
- Compensated covenants,
- Land leasing,
- Transferable (or tradeable) permits, and
- Environmental offsets.

Environmental charges, tendering, compensated covenants, land leasing are all incentives that affect the cost of using or conserving a resource and are price-based incentives. Transferable permits and environmental offsets affect the overall quantity of the resource and are referred to as quantity-based incentives.

2.2 Environmental Charges

Environmental charges aim to reduce the level of a discharge to the environment by charging a fee per unit emitted. A closely related approach involves reducing the amount of some tax or fee that an individual is required to pay in exchange for undertaking some activity that benefits the environment. An example is making expenditures on solar hot water heater tax deductible to address green house gas emissions. Both environmental charges and the related tax or fee reduction approach create private incentives to reduce environmental impact by offering opportunities to minimise environmental charge payments (or maximise tax rate reductions). In some settings, charges can induce innovation and result in declining discharges over time.

The choice of whether to use charges or tax rate reductions depends on social and political values about whether the polluter should pay or the beneficiary (society) should pay for the environmental improvements. Charges are clearly an application of the polluter pays principle, while tax rate reductions are an application of the beneficiary pays principle. The person that takes that action with positive environmental impact receives a public payment in the form of a reduced tax burden.

However, as environmental quality has deteriorated or come under threat in NZ, there has been a trend towards introducing fees charged to those whose activities adversely impact the environment. This is broadly reflective of trends elsewhere. In the United States, Australia and the EU, there has been a distinct move towards setting charges in proportion to the quantity of emissions. Some European countries such as Denmark and the Czech Republic, have been implementing charges targeted to municipal wastewater dischargers and industry. In the Netherlands, there are significant charges for nitrogen and phosphorous use on farms in excess of set limits. Australia as part of its National Competition Policy is reviewing how environmental externalities are incorporated in urban and rural water charges (National Competition Council, 2003).

Charges can be designed in any number of ways and for any number of reasons including raising revenue. If the purpose is to send a clear signal to the polluter, then the charge would require each polluter to pay a charge equal to the cost of the individual incremental damage associated with their discharge. Economists refer to this ideal charge as a Pigovian tax. Pigovian taxes result in each polluter having the incentive to reduce discharge until the cost of avoiding the last increment is equal to the value of the last increment of damage avoided. Actually implementing this type of ideal charge in the real world is simply so difficult that it is rarely, if ever, done. The actual physical damage per unit discharge is often not understood with any certainty and the value of the damage done is equally difficult to determine. Even attaining approximate values often involves considerable and expensive research. The "Pigovian prescription" also involves charging each polluter differently. Attaining such fine differentiation involves a level of administrative effort that is typically not feasible or cost effective. In addition, such differentiation is likely to be seen as inequitable by many and can be subject to legal challenge.

The result is that most charges are "second best" approximations to Pigovian charges given real world information and administrative cost limits. For instance, a charge that retains some properties of a Pigovian tax is a per unit charge that increases in "tiers or blocks" reflecting the nature of the damages. The NSW EPA Load-based Licensing program for regulating industrial facility pollution, described in Text Box 2 (from Bright *et al,* 2002), is a good example of a charges approach. The program involves a "tiered pricing scheme". Charge rates increase when discharges exceed prescribed limits. This reflects the increasing cost of higher rates of discharge to the environment and other users. An alternative second best charging mechanism, where the impact of discharging pollutants is not uniform across location, is zone-based charges.

Charges may not necessarily represent the best policy choice in situations when discharges over a certain threshold level cause significantly more damage than discharges below the threshold level. This is because the level of discharge that can be expected is determined by responsiveness to price signals that may not be well understood before charges are implemented. Responsiveness to price will also change over time with factors like technology. If the response to charges have been underestimated[1], much greater than anticipated environmental damage can result from charges. Quantity-based approaches, in contrast, involve a discharge rate and thus ensure a particular outcome.

A charging approach is most easily applied to situations where charges can be set in proportion to the environmental outcomes of interest. The prerequisite is that outcomes can be monitored in a relatively straightforward manner. When monitoring is difficult and expensive, implementing charges will typically require significant scientific effort to relate outcomes to levels of some easily measured "proxy". A proxy is an input or outcome which is correlated with the outcome of interest. For example, the level of manure application to fields might be a proxy for nutrient loading from manure application. The benefits of implementing charges can become more difficult to justify when charges are not based on direct outcomes. This is because the costs of establishing a relationship between the proxy and outcomes can be expensive. Typically, outcomes are less than perfectly correlated with proxy input use. In practice, the use of such charges is much easier if it is made clear that the aim is to signal the general nature of the cost rather than to set the charge with precision.

[1] That is, a much larger increase in price would be required to reduce the demand for the input associated with the pollution.

Text Box 2: The NSW Load-based Licensing Program

The NSW EPA Load-based Licensing (LBL) program initiated in 1999 is a new system for regulating air, water and noise pollution and waste management from industry. Under the program, industries are charged an annual licence fee based on the total amount of pollution emitted each year. The annual licence fee consists of a basic administrative fee and a pollutant load fee. The load fee varies both by pollutant and by location in recognition of the fact that environmental damage depends on the pollutant released and where it is released. To provide industries using particularly antiquated "dirty technology" a strong incentive to update, fee rates double if the load goes over an emissions threshold that is defined for each industry type and pollutant.

A shortcoming of the charges approach can be that the level of pollution reduction that will occur in response to charges is unknown. The LBL program provides an upper limit on this type of uncertainty by describing an effective upper limit on each pollutant for each industry type. Violations of this upper limit can result in fines of up to $250,000 for corporations and $120,000 for individuals.

The LBL fees that are charged to NSW industries can represent a significant expense and a significant incentive to reduce emissions. Large coal-fired power plants such as those in the Hunter Valley and on the central coast near Lithgow, can expect LBL fees of up to $1.3 million per year, primarily as a result of the large amounts of nitrogen oxides (NO_x) emitted from such plants. LBL incentives have created a market for software that can reduce NO_x emission from coal-fired power plants by up to 20%.

"By comparing emissions of NO_x to other measurable variables, such as the temperature of steam and flue gas, the position of air dampeners and the rate of fuel supply, the software can deduce which configuration of boiler operation leads to the cleanest production. The software learns quickly and can optimise performance after only two weeks of operational monitoring.

A number of NSW power station operators are currently investigating the installation of the software at their sites. With annual load-based fee savings on NO_x emissions for a typical four-boiler power station of $250,000, and with increased operational efficiency, the software will usually pay for itself in just two years."

Sources: Bright *et al*, (2002), NSW EPA, 2001 and quoted material from NSW EPA cleaner production case studies web-site
http://www.epa.nsw.gov.au/cleaner_production/cases-05.htm

Text Box 3: Zone Based Salinity Charge

Environmental charges have been applied to finance the mitigation of salinity impacts from diffuse irrigation sources in a part of the River Murray in Victoria since 1993 under the *Nyah to the South Australian Border Salinity Strategy*. This charges approach applies only to new irrigation development and is a good example of a feasible "second best" environmental charges approach.

Ideally, it would be desirable to charge irrigators per tonne of salt that resulted from their irrigation. However, this is not possible because salinity impacts of irrigation, like many diffuse source emissions, cannot be monitored directly. Salinity results from processes that take place below the soil surface and involve significant time delays (sometimes several decades) between actions and impacts. Furthermore, the salinity impacts of similar irrigation practices are "non-standard" - they vary considerably across locations.

To overcome the monitoring problem, charges are levied on "a proxy" input, the volume of irrigation water applied. At any location, irrigation application can be considered a proxy for salinity loading because the level of use of this input is correlated (albeit imperfectly) with the environmental performance variable of interest, salt loading. To address the non-standard impact issue, "impact zones" have been identified and different charges set for each zone. A standard "average" salinity resulting from the same level of irrigation has been defined for each of four zones. As summarised below the approach involves higher charge rates per ML of irrigation water applied in zones where the modelled salinity impact of irrigation is greater and lower charge rates in zones where irrigation salinity impact is less (Sunraysia Rural Water Authority, 2002).

Zone	**Estimated Salinity (EC/1000 ML)**	**Charge per ML (Paid Once Off)**	**Annual Charges per ML if paid over 10 years**
L1 – low impact zone 1	0.02	$26	$3.21
L2 – low impact zone 2	0.05	$65	$8.01
L3 – low impact zone 3	0.1	$130	$16.03
L4 – low impact zone 4	0.2	$260	$32.06

* There is an additional $3.20/ML/year charge for operations and maintenance in all zones.

2.3 Tendering, Compensated Covenants and Land Leasing

2.3.1 Tendering

Tendering is being used by NRM agencies to distribute public funds across private firms and individuals who engage in activities to improve the environment. Tenders work by having those interested in the public funds compete by submitting an offer to undertake work for a price. The tenders describe the on-ground works that will be provided, an assessment of the likely benefit and the payment that would be required to undertake the works. The NRM agency then ranks all tenders received on the basis of cost per unit of environmental benefit offered. Tenders are accepted in order of cost per unit of environmental benefit offered until the budget is exhausted.

With this tendering system, each individual, rather than the government agency, sets the cost share rate they would be willing to accept. The tendering approach allows landholders who are willing to undertake environmental improvements at low cost-sharing rates to do so. The result is that a higher level of environmental improvement effort can be attained for a given expenditure than is achievable when a single cost sharing rate is offered to all. Text Box 4 below from Connor and Bright (2003) describes the Victorian DNRE BushTender approach to allocation of biodiversity conservation cost sharing. Text Box 5 presents an example of tendering already in place in NZ.

Text Box 4: The Victorian BushTender Program

BushTender is an incentive based approach to encouraging private land management practices that will protect and enhance remnant native vegetation. The program was first trialled at two pilot sites in Victoria in 2001. Participants, working with program officers, prepare plans describing actions they are willing to take to enhance biodiversity on their property. After preparing plans, potential participants submit sealed bids stating the cost-share payment they would be willing to accept to carry out the plans. The Department of Natural Resources and the Environment (DNRE) then sort the bids on the basis of cost per unit of ecological value. The result is a ranked list of offers ordered by the incentive payment requested per unit of ecological value. The DNRE then accepts cost sharing offers in order of value of ecological benefits per cost-sharing dollar until the program budget is exhausted.

Evaluation of the first year of program experience led to the conclusion that significant numbers of program participants offered to undertake high levels of on ground works for small incentive payments. It was estimated that about 25% more environmental benefit was achieved as the result of giving out $400,000 of incentive money through tendering than would have been achieved had cost sharing been offered at a set cost sharing rate (Stoneham *et al*, 2002).

Text Box 5: The East Coast Forestry Program

> The aim of East Coast Forestry Project in NZ is to provide a financial incentive for land users to change the management practices in an area of 60 000 hectares in the Gisborne District by 2020 where erosion is worst. Landholders with land in the Gisborne district submit an offer to change management practice in return for a payment.
>
> Land in the project area has been mapped based on the potential threat of stream sedimentation. Tenders are accepted from landholders willing to undertake treatment options in two rounds - a round for small blocks and a round for larger blocks. Tenders are prioritised based on a ranking that utilises a land classification weighting system that seeks to identify areas where erosion is worst. A tender application with a low percentage of erodible land is more likely to be rejected, all other things equal.

Recent work by Hailu and Schilizzi (2003), using an agent based modeling approach, suggests that the efficiency gains from BushTender may be limited to the first few years. After the first few years, the least cost bidders have completed their on-ground works and the "expensive" bidders have dropped out. The remaining bidders tend to provide bids that converge towards a band of acceptable bids. Applying this concept to water quality issues, the early rounds of tendering allows the NRM agencies to identify the landowners or firms that are the most enthusiastic and tendering allows NRM agencies "to pick the low hanging apples". Sustained cost savings therefore may not occur over successive rounds as landholders "learn".

2.3.2 Compensated Covenants

As the use of MBIs has increased, an issue has been emerging with respect to how governments can protect publicly funded on-ground works on private land. More generally, the question is one of how can governments ensure the on-going provision of environmental benefits/ecosystem services on private land after a payment has been made. These two inter-related questions can be addressed through different types of agreements, easements and covenants that essentially contract a land holder (and sometimes their successors) to continue providing the benefits in exchange for tax concessions or nominal payments.

Governments have been experimenting with ways to protect the public interest on private land. The approach can be as general as a management agreement where the landholder and government enter a contract regarding how the land is to be managed for a specified time period and payment schedule. Examples of this include the Land Retirement Programs, Farm and Ranch Lands Protection Programs, and the Debt for Nature program where development rights are transferred in exchange for the cancellation of loans with the Farm Service Agency in the USA. Examples from the

UK include the Countryside Stewardside Scheme where landholders enter into 10 year agreements to manage land for habitat in return for annual payments. In South Australia, once-off payments have been used to negotiate covenants, known as Heritage Agreements, in perpetuity.

The basis for most programs has been a contractual arrangement where the landholder agrees to undertake some activity or refrain from some activity in return for some financial incentive such as a payment, tax concession or cancellation of loans, etc. Contractual arrangements provide a high degree of variability and can be tailored to the situation such that the agreement provides the legal protection that parallels the conservation value (Binning and Young, 1997).

In NZ, covenants are used to protect important ecological assets. Examples include the blocks of forest in the Eastern Bay of Plenty and the Aorangi-Awarua project. Maori landholders can protect indigenous ecosystems under a Nga Whenua Rahui kawenta. These covenants provide long-term protection with provisions for inter-generational reviews of the agreements. Funding for fencing, pest control and other management costs may be provided through the Nga Whenua Rahui Fund.

2.3.3 Protective Land leasing

In NZ, NRM agencies will purchase a leasehold interest in privately owned land in order to directly undertake land use practices that are necessary to achieve particular protection or conservation outcomes. An example is Kapenga Wetland which is being leased from Maori Farm Trustees.

Opportunities also exist for using other forms of land leasing where public land is leased for agriculture on the condition that best management practices for water quality are implemented. One condition might be to require that environmental management systems that focus on aspects of erosion control features be developed and implemented.

2.4 Transferable Permits

Transferable permits involve setting an initial overall limit (cap) on the emissions or pollutants. A system is then set up for allocating the amount that a group or individuals are allowed to emit or to pollute in the form of a permit. Individuals are only allowed to exceed their initial allocation if they purchase additional permits from someone else. Those who can achieve large reductions in pollutants at low cost are motivated to sell part of their allocation of permits at a profit to others. The approach encourages producers to think about the external impacts of their production activities and to search for innovative, low cost ways to reduce them. As a result, transferable permits can allow achievement of a desired level of emission at considerably less cost than uniform environmental standards can.

Transferable permit approaches tend to be preferred to environmental charges by producers who anticipate that they may be able to profit by producing significant reductions in emissions. In contrast, with a discharge fee policy, all producers pay the environmental agency, unless they produce absolutely no emissions. In effect, most permit systems use a market mechanism to set the lowest charge needed to achieve an outcome and then redistribute the resultant revenue amongst producers. In contrast, under a charging system, the resultant revenue is usually kept by the agency.

Experience with transferable permits suggests that establishing rules for trading is not sufficient to guarantee trades will occur. The Fox River in Wisconsin is an example of a program where very few trades have occurred. At best only a thin market was created due to the restrictions on trades designed to protect third parties from local adverse water quality impacts of trade and the uncertainty created for the polluters concerning renewals and how trade might affect renewal rights. An important implication is that when significant restrictions on trade are necessary to protect third parties, a tradeable permit approach might not be a good instrument choice.

Ex-post evaluations of programs in the USA, suggest that for point source pollution problems, transferable permits have had some success. The Acid Rain programme was successful in demonstrating that large scale trading can occur and well-designed programs can be resilient to unexpected changes or surprises. When compared with command and control regulation, it is likely that transferable permit systems were better at adjusting to sudden changes such as the deregulation of the railway systems which allowed new sources of low sulphur coal and significant fuel-switching to occur. Smith (2002) suggested that design mistakes to be avoided include allowing too much flexibility or bonus allowances in the early phases to ease the adjustment process.

Transferable permit programmes are being considered and in some cases implemented in many forms across the industrialised world in the area of green house gas emissions, renewable energy, solid waste and other non-point source pollutants. While Raux (2002) suggested that while transferable permits are particularly challenging for diffuse source pollutants, the approach has been used for such problems with some success. An interesting example is presented in Text Box 6 for on-farm nutrients.

Text Box 6: The Dutch excess nutrient trading program

The Netherlands have experienced significant non-point source pollution problems. One important cause is the very intensive agricultural production (particularly livestock production) in the country. In response they have developed a rather extensive policy to deal with the issue. One of the approaches being used in the Netherlands is the MINAS program. MINAS is essentially a tradeable permit approach for nitrogen and phosphorous applied as fertilizer on farms.

The system applies to pig, poultry, mixed livestock and cattle farms with stock rates above a set density (in total about 50% of Dutch livestock farms) and arable farms. Farmers in the MINAS program are required to prepare and submit a farm level mineral account. The sum of phosphorous and nitrogen "surplus" from artificial and manure sources is calculated in these accounts that are audited once a year. The allowable surplus has been revised downward over time. For instance, P limits have declined from 40 kg/ha in 1998 to 30 kg/ha in 2000 to 25kg/ha in 2002. Farmers exceeding their surplus can "trade" by giving excess manure to farms that are under their surplus. Those exceeding their quota are charged. The charge is 2.5 guilder/kg/ha. As many as 90% of farms pay no charges because they supply manure to arable crop farms with unused manure capacity.

While there have been no empirical evaluations of the program to date, there have been several modelling studies. One study found that the policy should decrease N use by 20% and P use by 30%.

Source: Dwyer *et al*, 2002

Finally, water quality is often related to the quantity of water being pumped whether it is a groundwater or surface water system. Young and Hatton MacDonald (2003) have suggested that water quality issues can be accounted for in a groundwater trading system by finely differentiating rights to water. Specifically, the right to water is separated into a right to receive a periodic allocation of water and a right to use the water. This allows for trading possibilities with respect to the water, salinity and other water quality issues.

2.5 Environmental Offsets

Environmental offsets are most often used where environmental quality goals are only just being achieved, or where there is non-attainment, and there is pressure to allow further development. The idea behind offsets is to allow further development to occur, but in a way that either maintains or improves the stock of environmental capital. Thus it is possible to have further development that damages the environment, provided that works are undertaken to either offset or more than offset this damage. This is consistent with the concept of weak sustainability, whereby sustainability is achieved by maintaining the stock of environmental capital at some specified minimum level, possibly by "projects/policies designed to produce environmental benefits" (Hanley, Shogren and White 1997, p.429).

In a catchment, for instance, developers may be required to implement best available technology (BAT) to control water pollution. However, development with BAT may still lead to a net increase in discharges. Under an offset program, this increase would be offset by a reduction of an equal or greater amount somewhere else in the catchment. Developers either undertake the offsetting action themselves or pay for others to do it on their behalf.

Environmental offsets have been used in a range of circumstances. Broadly these can be grouped as involving either pollution or biodiversity/vegetation offsets. Examples of pollution offsets include air quality in the USA (as prescribed by the Clean Air Act), water quality offsets in the USA (see Text Box 7), and drinking water and air quality offsets in NSW, Australia[2]. For biodiversity/vegetation offsets the best known examples are wetland mitigation and streambank mitigation banking. This sort of environmental offset has grown rapidly in the USA. In 2001, there were estimated to be over 200 operational wetland mitigation banks in the USA, and over 100 awaiting regulatory approval (ELI, 2002). However, wetland mitigation banking is not without its critics. While it is apparent that wetland mitigation banking has led to much larger areas of wetland being preserved, there is evidence that the quality of the wetlands produced through offsets is inferior to that lost through development (Salzman and Ruhl, 2000). However, this is less likely to be an issue with pollution offsets.

[2] http://www.epa.nsw.gov.au/greenoffsets/index.htm

Text Box 7: Rahr Malting Company Offset

The Rahr Malting Company is located on the Minnesota River. Along the lower 25 miles of the river the total daily maximum load for biological oxygen demand (BOD) is fully allocated. The Rahr Malting Plant was treated as a new point source when it redirected its discharges into its own wastewater treatment plant instead of the local municipal water treatment plant. A stringent discharge limit plus an offset clause was written into the facilities discharge permit..

In exchange for increasing BOD discharges, the plant has financed upstream reductions in phosphorus non-point source discharges. Rahr has established a trust fund to oversee the offsets. The trust fund was initially established at $200,000 and will be augmented by $5,000 per year over the life of the offset. A board that includes citizens, state officials and company representatives oversees the trust.

An offset ratio of 2:1 was used to allow for differences between point and non-point source discharges, plus an additional 8:1 ratio to allow for control of phosphorus rather than BOD. This latter ratio reflects a scientific assessment of the relative impacts on chlorophyll from phosphorus runoff and BOD discharge. The Rahr Plant has now fully offset 150 pounds of BOD per day, and has exceeded the required offset by 62 pounds per day.

Source: Environomics (1999), Klang (2000).

When the development and offset actions have similar impacts – as is more often the case with pollution offsets - ensuring net environmental benefit from an offset policy is more straightforward. This is achieved by developing “trading ratios”, which indicate the amount of offsetting required for a given development impact. As an example, a trading ratio of 2:3 implies that a development that introduces 2 tonnes of phosphorous into a waterway would require 3 tonnes of phosphorus elsewhere in the catchment to be offset. Trading ratios are usually based on several factors, including the desired environmental improvement, the level of uncertainty about environmental impacts, and the distance between the development and offset sites (Morrison, 2003).

Environmental offsets are usually undertaken prior to the development occurring so that there is no temporal loss in environmental quality, and to provide additional certainty that the offsets will be effective. In the case of water-based offsets (such as non-point source pollution), offset trades are also usually restricted to occur within a “geographical service area” so that no localised problems or hotspots occur.

Environmental offsets can be implemented via bilateral negotiations between stakeholders (i.e. where developers directly contract with owners of potential offset sites), or through privately or publicly owned offset banks. An offset bank is not a bank in the usual sense. Rather an offset bank involves the completion of one or more projects in which environmental remediation works are undertaken. By completing these works, offset banks earn “credits” which can then be sold to developers who are creating net-impacts on environmental quality (Morrison, 2003). In the case of wetland mitigation and streambank mitigation banking, offset banks are run by either private business, non-profit organisations or, in some states, by government organisations. The majority of wetland mitigation banks are privately operated (ELI, 2002).

Compared to other MBIs, environmental offset schemes have a number of unique requirements. These include:

- Development of enabling legislation (this is also required for trading schemes, but not for other instruments such as tendering where participation is voluntary),
- Changes to licenses of new developments so that no net discharges are allowed,
- The capacity to estimate changes in loads at development and offset sites (several software packages have been developed to do this), and
- Development of offset banking instruments.

As a general rule, one of the main advantages of an offset program is that they do not require all land users to be monitored. As a result they tend to be less expensive to implement than a tradeable permit system. Moreover, with an offset program, there is no need to find a way to "allocate" permits in a manner that is politically acceptable. Other implementation issues for environmental offset schemes are discussed in Morrison (2003).

The use of environmental offsets is likely to be supported by developers because it provides a way of achieving further development in areas where there is no "unused" environmental capacity. Politically this may be a useful option where the benefits associated with development (e.g. jobs, regional income) may be high. In contrast, environmental offsets have received less support from green groups because of concerns that offsets have been of inferior quality to what is lost through development, and because of concerns that it will increase the rate of damage to the environment.

The economic theory behind the use of offsets has not been well developed (Baumol and Oates, 1989; Vernon and Goddard, 2003; Morrison, 2003). From an environmental management perspective, and from the perspective of the community, offsets are an appealing instrument because development can proceed provided that environmental damage is offset.

Efficient regulatory instruments should move an economy from a private market optimum to an optimum that maximises social welfare. An effluent charge (or Pigovian tax) does this by setting a tax equal to the damage cost of pollution. With any permit, offset or charging scheme, it is possible that the cost of the offsetting action (plus the cost of administering the scheme), may exceed the damage cost of the externality. This is likely to be the case in highly urbanised areas where the cost of offsets can be particularly high and require specialised engineering solutions and costly retrofits. In peri-urban or rural areas where there are more offsetting alternatives, this is likely to be less of an issue. As well, from a policy perspective the concern is not only with costs, but also with benefits and equity. An offset program may ensure that the environmental capital stock is maintained, but a change in the composition of this capital stock may alter the benefits of the capital stock to the community, as well as affect the distribution of benefits. For instance, developing a wetland within an urban area and replacing it with a wetland some distance from the urban area could substantially reduce the local recreational value of the wetland area at one location but increase recreational opportunities at the location of the offsetting action. Overall, however, the net-recreational benefits to the community may fall. Further, a new wetland may not provide the same quality of ecosystem services that an existing wetland provides.

These warnings should not be seen as suggesting that offsets are likely to be an unworkable MBI. Cost-benefit analysis with careful statement of the basis for comparison will indicate the practicality of implementing an environmental offset scheme.

One important limitation of offsets is that they do not directly deal with existing pollution, except to the extent that developers more than offset their impacts. In this sense, offsets are not fully consistent with the polluter pays principle. New polluters pay to reduce their pollution, but existing polluters are not penalised. This characteristic of offset arrangement can sometimes be useful because it is harder to "claw back" existing entitlements than establish new standards, but it can also limit the effectiveness of the instrument when current entitlement are causing significant environmental harm.

Finally, it should be noted that there are opportunities to use environmental offsets together with other instruments. Offsets can be a good complement to other MBIs such as trading schemes, traditional regulation, and environmental charges.

3 DEVELOPING A FRAMEWORK FOR EVALUATING MBIs

A goal of this report is to provide a framework for identifying which MBIs might be most attractive to deal with the particular water quality issues faced by NZ. The framework proposed is a qualitative approach based on identifying key advantages and constraints from Connor and Bright (2003) and Lockwood, Morrison and MacKay (2003). The core of the framework is a stepwise screening approach.

Each step gathers information about the environmental processes, institutional settings, potential adopters and technologies that may influence the performance of an MBI. Ultimately information is obtained to answer these key questions:

- Which of the variants of MBIs are feasible in the existing institutional settings?
- What is the potential of different MBIs to ensure environmental quality goals are met?
- What is the potential of different MBIs to reduce the cost of meeting environmental policy objectives?
- What is the risk profile of the groups being targeted by the policies and instruments? How will these groups react to the policies and instruments?

The result of applying this screening process may not be a definitive identification of "the" MBI for a given problem. Rather, applying the framework should result in:

- Identification of the types of instruments that are feasible given the current institutional setting, and the range of institutional changes that might be required;
- Identification of the environmental process, institutional setting, market characteristics, technology and policy design attributes that favour or impede reliable and cost-effective realisation of NRM policy goals;
- Identification of the likely mix of instruments both market and regulatory approaches required to achieve NRM goals.

It is likely that even after applying the screening process some questions will remain unanswered. For example, the framework identifies the mitigation response as a key determinant of the effectiveness of an environmental charge. This type of information may not exist. The framework may well provide an indication of future research priorities - research that may be required before good judgements can be made about the potential for some MBIs.

3.1 Screening institutional determinants of feasibility

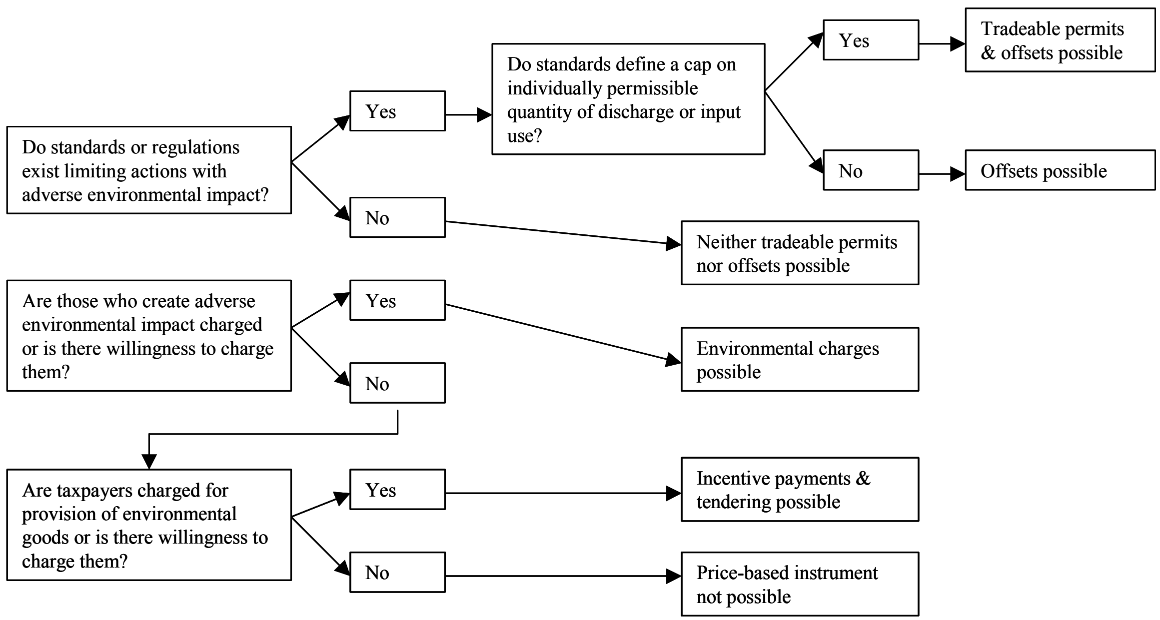
Once a decision is made to explore the potential of market-based tools, decision makers are confronted with numerous examples of instruments that can be used and the practical question becomes "which instrument"? Figure 1 shows the first step or the initial screening process where the class of MBIs (refer to the categorisation of Text Box 1) is identified given the institutional rules that are already in place.

The feasibility of price-based instruments is influenced by a key set of institutional rules - rules about who is held responsible to pay the costs of mitigating adverse environmental impacts. These rules are often less formally institutionalised than property rights and often set on a once-off basis in specific legislation. Nonetheless, precedent is often an important basis for institutional rules. So environmental charges are much more likely to be politically acceptable where there is some tradition of making the polluter pay. Likewise, incentive payments and tendering approaches are more likely to be politically acceptable where there has been a tradition of charging the general public for provision of public environmental goods.

Quantity-based MBIs are only feasible if some type of overall cap or standard that limits emissions is in place. Tradeable permits require a very specific type of property right that is usually associated with water use or discharges to air or water - a performance standard that assigns a specific amount that is allowable for use or discharge for each individual and allows trading in the right to use or discharge. Offsets, in contrast, can be implemented with a range of standards, development restrictions or other rules limiting activities with adverse environmental consequences. In essence, once a standard has been established, offsets can be used to allow the standard to be relaxed at one site, if this is compensated for by providing environmental improvement elsewhere.

Ultimately, institutions change over time. The absence of an institutional prerequisite, identified through the screening process, does not mean that the MBI remains forever infeasible. Rather, it indicates where effort in institutional development would be required before the MBIs could be implemented. In addition, there are sometimes opportunities to overcome institutional impediments with clever policy design. Such possibilities will be discussed in the application of the framework to analysis of MBIs for water quality that follows.

Figure 1: Institutional determinants of feasibility of a particular MBI



3.2 Screening determinants of potential to reduce cost of achieving environmental outcomes

Table 1 is the second step in the screening process from Connor and Bright (2003) for ascertaining the extent to which attributes of the environmental process, available monitoring technology, markets and the details of instrument design favour or impede cost effective realisation of NRM goals.

Table 1: Determinants of environmental policy instrument cost effectiveness

Type of attribute	Are the following attributes favouring cost savings from instrument implementation present?	Type of Instrument		
		Tradeable permits and offsets	Tendering	Environmental Charges
Environmental process /market	Large variation in cost of supplying environmental impact mitigation	x	x	x
Technological /market	Monitoring, modelling of environmental performance feasible at reasonable cost	x	x	x
Environmental process	Absence of highly localised environmental impact "hot spots"	x		
Market	Private willingness to pay for environmental performance (or product attributes associated with environmental performance)		x	

The screening process identifies large variations in the cost of mitigation as a factor influencing the potential cost effectiveness for both quantity-based and price-based MBIs. Tradeable permits, offsets, environmental charges and tendering all give those with the lowest cost of reducing impact the incentive to provide higher levels of mitigation than those with higher cost. Market-based approaches set the right incentives for private parties acting in their own interest either to internalise the cost of pollution or provide more pollution reduction from least cost mitigation sources and less from higher cost sources. The result can be significantly lower cost of achieving an emissions goal where there is significant variation in cost of mitigation.

An additional determinant of when efficiency gains to quantity-based MBIs are possible is the technical feasibility and the cost of monitoring environmental performance. MBIs have generally been more effective where emissions are from a point source such as a pipe, or smoke stack discharging from an industrial facility than when emissions are from diffuse sources such as farms. One of the main reasons is the difference in the feasibility and cost of monitoring in point source versus diffuse source emissions settings. In point source settings, direct monitoring of emissions performance (at the end of a pipe or smokestack) is generally possible at a cost that is reasonable compared to the benefit. Thus in point source settings, it is usually possible to base permit trading or offsetting on actual performance (levels of discharge).

In diffuse source emissions situations, in contrast, it is often necessary to specify best available technology (BAT) or best management practice standards (BMPs) as a proxy to a performance standard. Text Box 7 presents an example of scheme that allows point sources to trade with diffuse sources by allowing point sources to buy permits from diffuse sources (farms) for implementing best management practices. It also highlights the potential for associations representing a small group of polluters to take more responsibility for developing and implementing best management approaches. Small numbers need not present an insurmountable problem.

Text Box 8 - Trading Point Source and Non-point Source

The Tar-Pamlico Basin was designated as "nutrient sensitive" and a cap or basin-wide bubble was established for the 16 municipal and industrial point source dischargers. The Association agreed to reduce its nutrient discharges into the basin and to share a single nitrogen discharge limit in lieu of individual nitrogen limits. The Association enforces the limit and internally allocates discharge limits among its members. As part of a move to change collective behaviour, the association has undertaken initiatives specifically targeted to addressing the best management practices of its members.

If the Association exceeds the annual limit, it purchases offsetting credits by contributing $56 per kilogram needed to the fund, which supports installation of BMPs (best management practices) on agricultural land. Credits are good for three or ten years, depending on whether they are for non-structural or structural BMPs. The State Division of Soil and Water Conservation regularly inspects cost share-funded BMPs and monitors BMP performance.

The Association believes that the reductions in nutrients would have cost the point source pointers in the order of $7 Million US. Trading allowed the same level of nutrients to reduced at a cost of $1 Million US through trades with non-point source emitters.
http://www.fwwa.org/PDF/NewToolsMarketIncentives.pdf
http://www.epa.gov/owow/watershed/trading/tarpam.htm

Other determinants of cost effective implementation of MBIs are listed in Table 1. In the absence of highly localised environmental impact “hot spots”, both theory and experience suggests that the way discharges disperse in the environment influences the capacity of permit trading schemes to achieve environmental goals. Trading schemes are easier to implement when emissions are “global” in that they disperse uniformly across large areas (i.e. SO^2 and global green house gases) and there is little difference in damage per unit emission across locations. For instance, when impacts are global, trades with simple one-for-one exchange rules are possible over large areas with many market participants.

When the diffusion is localised, the impact of emissions can be quite different from location to location. So for example, less dilution flow at some points in a river can result in more damage per unit emissions. One-for-one permit trade in such a setting can result in net damages rather than improvements. This would be the case in instances where permits were traded from a high dilution to a low dilution river reach. Consequently, if localised adverse environmental impacts are to be avoided, the approaches typically have to be implemented with restrictions on locations where emissions can be traded.

Overall, MBIs are likely to be more successful for emissions with global impacts compared with localised impacts. Simple one-for-one exchange rules are possible over larger areas with more market participants and less restrictive trade rules. Less complex trade rules generally increase willingness to supply permits because the cost of seeking out trading partners, evaluating trades and other transactions costs are lower.

3.3 Screening the segments of society that will be targeted

The range of attitudes and behaviours within the community towards water quality and the adoption of best practices suggest that there are different 'market segments' in relation to sustainable land management. For instance, there may be five segments within the 'sustainable land management policy market':

Segment 1: Potential adopters, financial capacity to invest in best practice

Segment 2: Potential adopters, no financial capacity to invest in best practice

Segment 3: Non-adopters, risk averse

Segment 4: Non-adopters, respond to peer group norms

Segment 5: Non-adopters, exclusive focus on farm products.

In marketing strategy, the existence of different segments typically induces firms to develop a range of products. For example, in the automotive market, there are many different kinds of vehicles to cater for the wealth and tastes of different consumers. People in different segments would also be expected to respond differently to different policy instruments and this is a facet of policy development that can easily be overlooked. Ideally, a mix of instruments could be devised that will result in behavioural changes in each market segment. Policy makers need to select instruments that are likely to be most effective for each segment.

If Segment 1 is the majority of the population, there is a case for using MBIs. If most people come from Segment 2, MBIs may work, but the incentives will need to be sizeable. Segment 3 may respond to incentives that are designed to deal with risk, and Segment 4 may respond to promotional campaigns that seek to encourage a general change in attitude in the agricultural sector towards water quality. Segments 3 to 5, however, may require strategies that do not involve financial incentives, because of their general unwillingness to undertake conservation management.

The appropriateness of different instruments to different market segments is considered in Table 2. Most instruments are suitable for creating behavioural changes amongst people in Segment 1. There are also many instruments suited to people in Segment 2, provided that the issue of limited finances is dealt with. For the remaining segments, there are fewer suitable instruments and regulation is most likely to be one of the few tools for addressing water quality with segment 5.

The relative size of these different market segments may be influenced by the presence of different ethnic groups within a catchment. Especially if these groups hold values or have attitudes to risk that are different to other groups. From a NZ perspective, recognition of these considerations could result in the selection of different instruments in catchments where a significant proportion of a catchment is managed or used by Maori people.

We have put forward a hypothetical example of market segments in Table 2 to illustrate the importance of targeting policies to particular groups within the community. The existence of market segments may partly explain why some MBIs have not worked in some cases, despite their apparent suitability for a given context. Although work has been done on understanding some of the social, psychological and economic attributes of rural communities, these data do not yet enable the derivation of policy-relevant market segments.

Table 2: Hypothetical classification of the suitability of policy instruments for different market segments

Instrument	**Segment 1: Wealthy Adopters**	**Segment 2: Low income Adopters**	**Segment 3: Risk averse non-adopters**	**Segment 4: Peer group Non-adopters**	**Segment 5: Farm product Non-adopters**
Environmental Charges	***				
Incentive Payments	***	***			
Transferable Permits	***	***	*	*	
Tendering	***	*			
Environmental Offsets	***	***	*	*	
Land Leasing and Compensated Covenants	***	*			

*** will achieve a significant behavioural response
** will achieve a moderate behavioural response
* will achieve a minor behavioural response

3.4 Determinants of potential to ensure environmental quality goals

Existing institutional settings, market conditions, size of market segments and details of policy design can all influence the likelihood of actually achieving desired environmental outcomes with an MBI as outlined in Table 3. Existing institutional settings are a significant determinant of the environmental effectiveness of any MBI. In the absence of well-defined standards, tradeable permit or offset approaches simply cannot be expected to meet environmental goals. Likewise, incentive payment approaches will not achieve desired environmental goals if they are not adequately funded. Environmental charges will not achieve desired goals if institutional constraints limit charges to rates that motivate little environmental mitigation.

One of the major challenges of instrument design is the need to reduce uncertainty for all parties, including uncertainty about the capacity of MBIs to ensure desired environmental outcomes. Essentially there are three sources of this uncertainty concerning:

- Responses to the incentives that the instrument approaches offer,
- How mitigation actions relate to environmental outcomes,
- How the institutional environment will evolve.

As a general rule, when supply response to price is poorly understood, and damages are a steeply increasing function of price, quantity-based instruments are more desirable. This is because if price responsiveness is underestimated, the price-based instrument will result in significantly more expensive damage than anticipated. Even when, supply response is reasonably well understood and is found to be quite elastic (ie very responsive to price changes) the argument may still be applicable, because supply response can change relatively quickly with factors such as technology.

In many instances, there is considerable uncertainty about the damage function because the biophysical processes that relate actions to environmental impact are incompletely understood. In such cases, ensuring that desired environmental outcomes are met through the use of quantity-based instruments requires definition of property rights in a way that allows updating as improved information about environmental processes becomes available (Smith, 2002). One approach can be to define the standards that underpin the quantity-based instrument for limited temporal duration. It is important if this approach is taken to define the uncertain conditions that will lead to revisions of standards, and to allow standards once set to remain in place long enough to allow some certainty about costs of compliance over that period (i.e. updating on 5 year or longer intervals).

Young and Hatton MacDonald (2003) identified that transparent processes about how rules will change over time in water rights will reduce the uncertainty for firms and individuals contemplating investments. If there is uncertainty about how transferable permits might be renewed, it is easy to see how risk adverse firms might not adopt new technology.

Table 3: Determinants of potential to ensure environmental quality goals

Type of attribute	**Attributes favouring cost savings from instrument implementation**	**Instrument type**		
		Tradeable permits and offsets	**Tendering**	**Environmental Charges**
Institutional	Sufficiently strict standards underpinning tradeable permit / offset scheme	x		
Institutional	Sufficient cost sharing budget		x	
Institutional / Policy design	Charge rate level sufficiently high and differentiated based on impact differences			x
Market	Environmental mitigation supply responsiveness to price		x	x
Market	Private willingness to pay for environmental outcome		x	
Policy design	Standards updateable Charge rates updateable Based on improved information about sustainable level of environmental impact, mitigation supply response to price	x		x

4 THE ROLE FOR NON-MARKET VALUATION

Non-market values are regularly included in cost-benefit analyses to indicate the value of changes in environmental quality. Sometimes the value of environmental quality is evident in market data. For example, the cost of reduced water quality might in part be indicated by the value of declining commercial fish production. However, this usually comprises only a part of the total economic value of a change in environmental quality. Non-market valuation is regularly used by government agencies to estimate the community wide value of changes in environmental quality for cost-benefit analyses. It can also be used in any other context where there is a need to identify the marginal value of a change in environmental quality – such as in the application of MBIs.

4.1 Non-market Values

When applying non-market valuation techniques, several different kinds of non-market values can be estimated (Pearce and Markandya, 1989).

- *Use value* refers to the value that individuals receive from the actual use of a resource. This includes any kind of consumption, benefits from amenity or aesthetics, vicarious use and recreational use.
- *Option value* refers to the value of maintaining opportunities into the future and reflects the value of uncertainty. In theory it can have either a positive or a negative value. Uncertainty can arise about the provision of a resource (supply-side option value) and if individuals are willing to pay a premium to remove this uncertainty, then the option value is positive. However, if the uncertainty arises about whether individuals will have a demand for a resource (demand-side option value), then the value is negative.
- *Existence value* refers to the value individuals have for a resource apart from any in situ use of a resource. It arises for various reasons, including altruism, bequest motives and stewardship. This value is sometimes described as non-use or passive-use value.

Total economic value is the sum of use value, option value and existence value. It is this value that one seeks to estimate using non-market valuation techniques.[3]

[3] The two main databases of non-market values in NZ and Australia are http://learn.lincoln.ac.nz/markval/ and http://www.epa.nsw.gov.au/envalue/.

4.2 Techniques for Valuing Changes in Environmental Quality

As mentioned above, part or all of total economic value can sometimes be derived from existing market data. This typically involves looking at changes in damage costs or preventative expenditures, or estimating changes in producer and consumer surplus[4]. For example, information about the reduced cost of treating potable water provides an indication of the benefits of reduced turbidity (Moore and McCarl, 1986). Or it may be possible to estimate the costs to tourism from an outbreak of blue-green algae (Walker and Greer, 1992). However, many of the benefits of improved water quality can only be estimated by using related or hypothetical market data. In these cases it may be possible to estimate non-market values using either revealed or stated preference non-market valuation techniques.

4.2.1 Revealed preference techniques

Revealed preference techniques use information from related markets to impute a value for non-market goods (Mäler, 1974). A related market is one that indirectly reveals values for environmental goods. The most commonly used revealed preference techniques are the hedonic price method and the travel cost method. The hedonic price method, a variant is the hedonic wage model, uses differences in property prices or wages to impute a value for changes in environmental quality such as noise, air quality, water quality, etc. In most hedonic price studies a regression equation is estimated where the price or wage is a function of all of the attributes, including environmental quality. The effect of changes in environmental quality on property prices or wages can then be quantified.

A second revealed preference technique is the travel cost method. The travel cost method is used to value recreational quality or changes in recreational quality. It uses information on travel costs and reductions in the number of site visits at greater distances from a site to estimate recreational use value. This technique is regularly used by State government agencies such as the National Parks and Wildlife Service to estimate recreational values (Gillespie, 1997). Where changes in environmental quality affect visitation rates, the travel cost method can also be used to value changes in recreational value. For example, Bockstael, Hannemann and Kling (1987) estimated that people were willing to pay $27 per household per season for a 30%

---

[4] Changes in consumer and producer surplus indicate the value to consumers and producers of changes in production. Consumer surplus is the difference between the price paid for a good and the amount that people would be willing to pay, while producer surplus refers to the difference between what producers are paid for a good and their costs.

reduction in oil, chemical oxygen demand, turbidity and faecal coliform pollution at downtown Boston beach sites.

However, revealed preference techniques such as the hedonic price and travel cost methods can only be used in limited situations where there are existing related market data. They can only be used to estimate use values and they are retrospective which limits their usefulness for cost-benefit analyses which typically require valuation of options which don't yet exist and of non-use values.

4.2.2 Stated preference techniques

A second class of techniques that can be used to estimate non-market values are those based on the stated preferences of individuals. Stated preference techniques involve the use of surveys from which estimates are derived of the non-market benefits of different resource use alternatives. Because they rely on the use of surveys, stated preference techniques can be used in more applications than revealed preference techniques. They can be prospective and used where no related market data are available for estimating use values. They can also be used to estimate non-use values.

The most widely used stated preference technique for estimating non-market values is the contingent valuation method (CVM). CVM questionnaires contain several well defined elements including a description of the study site, details of the proposed changes (including a method of payment), an elicitation question and a series of socioeconomic and attitudinal debrief questions. State-of-the-art applications of the CVM generally utilise the 'referenda' format for the elicitation question, an example of which is shown in Text Box 9.

Text Box 9: The referenda CVM format

> Would you vote to support the proposal to improve water quality at a cost of $50 per household, or do you oppose the proposal? (Tick one box)
>
> I would vote to support the proposal at a cost of $50 ☐
>
> I would vote against the proposal at a cost of $50 ☐

Under this format, respondents are asked whether they support a project given that they are required to pay a certain amount towards it, with the payment amounts being varied between respondents. The responses to the elicitation question are then regressed against several variables including the payment amount, respondents' attitudes, and socio-economic characteristics such as income, age, education etc. This equation is then used to estimate mean and median willingness to pay.

The CVM has the advantage of being recognised by respondents as a standard public choice instrument (as it is similar to a referendum). However, despite its wide usage, the CVM has several limitations. It is relatively costly to use, provides limited information about people's preferences and is arguably prone to various biases (Kahneman and Knetsch, 1992; Diamond and Hausman, 1994).

A second stated preference technique that has been increasingly used to estimate the value of improved environmental quality and could be applied to valuing improved water quality is choice modelling (CM). CM has been used in a small, but growing, number of studies to estimate non-market values. It has grown in popularity because of its informational efficiency, ability to generate values for resource attributes (see below) and because of concerns regarding the validity of contingent valuation. Given its increasing usage in non-market valuation application, we have provided additional detail in this report about this technique, including about the value estimates that can be generated using it (see Bennett and Blamey, 2001).

CM questionnaires are similar to CVM questionnaires in that they contain background information about the non-market good, an elicitation question, and debrief questions. The main difference between the two methods is in the form of the elicitation question. In CM questionnaires, respondents are presented with a series of choice sets, each containing usually three or more resource use alternatives. An example of a choice set is shown in Table 3. From each choice set, respondents are asked to choose their preferred alternative. The alternatives in the choice sets are defined using a common set of attributes (eg characteristics such as water quality, % native fish, health of aquatic plants etc.), the levels of which vary from one alternative to another.

Text Box 10: Example of one choice set in a choice modelling questionnaire

Please indicate the alternatives you prefer most by ticking one of the boxes below:

	Alternative 1	Alternative 2	Alternative 3 (the status quo)
Water quality	Good	Fair	Poor
% native fish	50%	80%	5%
Health of aquatic plants	Fair	Good	Poor
Increased water rates	$10	$40	$0
	☐	☐	☐

In CM applications goods are decomposed into a set of 'attributes' or characteristics. For example, a car could be considered to be simply the sum of its component parts i.e. 4 wheels, a chassis, an engine etc. Or in the case of valuing river quality, the characteristics might be water quality, the percentage of native fish and the health of aquatic plants. The trade-offs respondents make when choosing between alternatives are quantified using statistical techniques. Where one of the attributes involves a monetary payment, the resulting trade-offs can be used to estimate the value of each of the environmental quality attributes. This can be conceptualised in the case of purchasing a car. Existing market data might show that on average people may be willing to pay $1000 extra for air conditioning — this implies that air conditioning is worth this amount of money. Similarly, CM survey data may indicate that respondents are, on average, willing to pay $5 extra for an additional fish species to be present in a certain river.

Two different types of value estimates are derived in choice modelling applications. The first is known as implicit prices. These are point estimates of the value of a unit change in an attribute. They are useful for management decisions where information is required about the value of marginal changes in environmental quality, such as the value of an extra 100 km^2 of wetland preserved. They are also useful for identifying the relative importance people place on different attributes. Implicit prices are calculated as follows, if utility is a linear function of all attributes:

$$IP = \beta_A / \beta_M$$

where IP is the implicit price, β_A represents the coefficient of the Ath environmental attribute, and β_M represents a monetary attribute.

The second type of value estimate derived using choice modelling is compensating surplus (or variation). This corresponds to what is estimated using contingent valuation. Compensating surplus is the value of a discrete change in environmental quality. This can be different from the sum of the changes in the implicit prices if the value that respondents have for an environmental improvement is not totally explained by the changes in the attributes. These value estimates are appropriate for use in cost-benefit analysis. Assuming that the only personal characteristic is income (m), and prices and other environmental goods are constant, compensating surplus is defined as follows:

$$V(\mathbf{x^0}, m) = V(\mathbf{x^1}, m\text{-}cs)$$

where V is an indirect utility function, $\mathbf{x}^0$ is the original vector of attributes, $\mathbf{x}^1$ is the vector of attributes after the change in environmental quality and cs is compensating surplus.

In practice, for a standard conditional logit model, this is estimated as follows:

$$CS = -\frac{1}{\beta_M}\left(V_{0-}V_1\right)$$

where β_M is the coefficient for the monetary attribute and is interpreted as the marginal utility of income, V_0 represents the utility of the initial state, and V_1 represents the utility of the subsequent state.

4.3 The Use of Non-market Valuation with the Application of MBIs

Non-market valuation techniques are frequently used in cost-benefit analysis where projects have an environmental dimension. They have been used to a much lesser extent with MBIs, but theoretically at least non-market valuation has an integral role in the application of many MBIs. The use of MBIs does not preclude the use of non-market valuation. Rather non-market valuation can be used to improve the use of MBIs.

Many economic and traditional regulatory instruments require that optimal levels of environmental quality be established independently. For example, traditional *command and control* approaches, *trading schemes* and *offset schemes* all require the setting of environmental standards. Whenever standards are set there is a potential need for non-market valuation. Theoretically, approaches are optimal when the marginal benefit from increasing the standard equals the marginal cost of doing so.

Non-market valuation is typically needed to measure non-market benefits. For example, consider a proposed policy to restrict nutrient loadings in a catchment – deciding whether or not this policy is appropriate would require estimates of both marginal benefits and marginal costs of implementation. If the marginal benefits are less than the marginal costs, then this new policy would decrease social welfare. If the opposite is true, the policy will be welfare enhancing. Similar to cost-benefit analysis, the determination of marginal benefits in this context would involve estimating Total Economic Value. In some cases it may be possible to use only revealed preference techniques. However, where non-use values need to be estimated, or where the change in environmental quality cannot be related in any way to existing market data, this would require the use of stated preference techniques.

Other MBIs can also benefit from the use of non-market valuation techniques. For example:

- For an optimally set e*nvironmental charge* it is necessary that the charges be set equal to the damage cost of the polluting activity. Estimation of these damage costs will normally require the use of non-market valuation. For some damages (e.g. due to effects on drinking water costs, or where any preventive actions are required), market data could be used to estimate this value. However, where recreational values are affected, techniques such as the travel cost method (a revealed preference technique that is commonly used to estimate recreation values) would be needed. And if existence values are affected – which is often the case when environmental quality changes – stated preference techniques would be used.
- For *tendering*, non-market valuation can be used to reveal whether or not the marginal benefits of any environmental improvements achieved exceed the costs of implementing the scheme and hence, if the budget allocated to this instrument should be increased.
- *Offsets* involve reductions in environmental quality in one area, but improvements in other area(s). As discussed previously, the community may have different value for these changes in environmental quality, even if the changes in environmental quality are ecologically similar. Non-market valuation provides one method of determining whether marginal benefits to the community from these multiple changes are equivalent, and whether an offset program would indeed enhance community welfare. Choice modelling would be a particularly useful non-market valuation technique in this context as it provides estimates of the value of the attributes of different resources. It can therefore readily be used to indicate whether an offset proposal would increase, decrease or have no effect on community welfare. Studies of this kind are already being undertaken in NZ (Kerr, 2003).

Thus non-market valuation should be seen as a means to increase the effectiveness of MBIs. It is not the case that non-market valuation is not needed if MBIs are being used; rather non-market valuation is needed for the optimal operation of MBIs.

5 MBIs FOR DIFFUSE SOURCE WATER QUALITY ISSUES IN NZ

Rich soils and rainfall make agriculture and forestry in the country very productive and an important sector of the economy. Agricultural products excluding forestry accounted for 6.7% of GDP in 2002 and agricultural exports are some 47% of all exports with forestry accounting for 11.7% of exports (MAF, 2003). Still in comparison to many other parts of the world, agriculture is relatively extensive in NZ and has to date, not had as large an adverse environmental impact as is the case elsewhere. There is, however, concern that if current practices are not controlled, further degradation of ecologically important assets, including some iconic and relatively pristine natural features, will result. In addition increasing peri-urban development fuelled by an increasingly urban and wealthy population is augmenting pressure on NZ's natural resources, particularly estuaries in the vicinity of urban centres. As in other countries, these three processes - agricultural, forestry development and urban development can come into conflict with ecosystem protection objectives and water quality objectives.

While the institutional arrangements and policies currently in place may have been adequate to deal with these issues in the past, additional measures may be required if increasingly severe adverse environmental impacts of agriculture, forestry and development are to be avoided in the future. MBIs could be an attractive approach for dealing with some of these looming environmental issues in NZ. The remainder of this report is an application of the framework for evaluating the potential use of MBIs to address diffuse source water quality issues in NZ.

5.1 The diffuse source water quality issues considered in this assessment

There is a range of diffuse source water quality issues in NZ. To demonstrate how the framework proposed in earlier sections would work in practice, the focus here is on two particular water quality concerns in NZ:

- Diffuse source leaching and runoff of agricultural nutrients to groundwater then to freshwater streams and lakes, and
- Turbidity and sedimentation of streams and estuaries as a result of surface runoff from urban development and forestry.

5.1.1 Discharge of nitrate from groundwater to freshwater lakes

One important and high profile water quality issue in NZ is the growth of phytoplanktonic algae and cyanobacteria (blue-green algae) leading to loss of clarity in many, still relatively pristine lakes (particularly Lake Taupo and the Rotorua Lakes). Algae have an impact on water quality for drinking and recreational purposes. For instance, the potential loss of clarity poses a real threat to a lake such as Lake Taupo. Algal growth is largely influenced by sunlight penetration, water depth, temperature, nutrients (mainly nitrogen and phosphorus) and grazing by invertebrates. Healthy streams and lakes will have little obvious alga. Algal blooms are usually a symptom of a system stressed by excess nutrients and/or elevated temperatures (Quinn and Meleason, 2003).

Nutrients reach rivers and lakes through surface run-off and through leaching and discharge of nutrient contaminated groundwater. Phosphorus from anthropogenic sources mainly enters water bodies in a particulate form with surface run-off because when phosphorus is applied to soils as artificial or manure fertiliser, it forms a strong chemical bond with erodible soil particles. In contrast, nitrogen is usually leached in a urea or ammonium form then transformed into its nitrate form where it is usually transported into surface water bodies through groundwater.

With NZ freshwater lakes in the central North Island volcanic plateau, nitrate is of particular concern as it is now considered to be the limiting nutrient and therefore the main cause of increasing chlorophyll and decreasing water clarity (Environment Waikato, 2003). A certain amount of nitrogen has always entered NZ lakes from rainwater and undeveloped land. It is the increased loading that has resulted primarily from development of grazing based farming systems after European settlement that is leading to declines in water quality (Environment Waikato, 2003). Nitrogen in grazing stock excreta, particularly urine, leaches readily to groundwater (Munday, 1999). While some nitrogen may be lost to the atmosphere through nitrification then denitrification, especially where soils have high organic matter content, once it has entered groundwater nitrate is typically conservative and does not transform into another compound (Cox, 2003).

Continued reliance on septic tank systems, which may fail over time, is also another important source of water quality problems. While the nitrogen loads from this source are often quite small in comparison to loads from pasture, (e.g. only 3% of the total nitrate load to Lake Taupo) they can have a noticeable impact on littoral zone slimes and filamentous algae that significantly detract from the attractiveness of lake edges for recreation (Environment Waikato, 2003).

Hydrology research confirms that there can be considerable delays between the time that nitrogen from a pasture begins to leach to groundwater and when resultant nitrate discharges to surface waters. These delays are the result of the relatively slow flow of groundwater and have the following important implications:

- Deterioration of water quality may continue even if no further expansion of agriculture in the vicinity of ecologically sensitive freshwater lakes occurs. This potential for continuing deterioration is a "legacy" of the expansion of agriculture through the 1960s and 1970s that is thought to not yet have fully manifested itself in nitrate discharges.
- Actions taken in the near future to reduce groundwater nitrate loading will have little immediate impact on water quality. For example, in the case of Lake Taupo, reductions in nitrate loading are only expected to change rates of groundwater nitrogen discharge to the lake twenty years or more in the future (Environment Waikato, 2003).
- Any intensification of agriculture that takes place will not immediately lead to water quality deterioration but is likely to cause time delayed increase in groundwater nitrate loading further down the track.
- The distance between upper catchment agricultural development and lake symptoms 30-40 km downstream creates challenges in demonstrating cause and effect to landowners.

In general, the capacity of lakes and wetlands to assimilate various nutrients without experiencing adverse impacts depends on a number of factors such as flushing time, depth, temperature, etc. Long term monitoring and significant scientific investigation are often required to explain and predict threshold levels beyond which additional nutrient loading is likely to have significant adverse impacts on water quality. Some ecologically or culturally important water bodies have been extensively studied. As a result, the level of nutrient loading reduction that would be required to reverse or avoid adverse impacts is now better understood. For example, the science is suggesting that further algal growth and water clarity declines in Lake Taupo could be avoided in the long run if the rate of nitrogen loading were reduced by 20%. In other less studied systems, nitrate loading reductions required to meet water quality goals are not as well understood.

5.1.2 Sediment loading in rivers and estuaries

Land uses involving clearance of natural vegetative cover that holds soil in place can lead to higher than natural rates of sediment loading in freshwater streams and estuaries. Important activities leading to sedimentation include agricultural cropping (with periods of bare soil) removal of tree cover for forestry purposes and urban development activities that involve removal of the vegetative cover. The level of sediment loading to water bodies resulting from such activities will vary across locations with generally higher rates where vegetation clearance is closer to water bodies, on steeper slopes, and in areas with more intense rainfall events.

Adverse economic impacts of sedimentation can include higher costs associated with water filtration and the cost of dredging to maintain navigable waterways.

Adverse ecological impacts can include:

- *Reduced photosynthesis* – this results when large amounts of suspended fine sediments lead to turbid conditions and light penetration is reduced. Impacts can be reductions in the presence of certain light dependent alga that form a part of the food chain and consequent reduced growth of microbes and animals that feed on them.
- *Disturbance of breeding habitat* – deposition of fine sediments into streambed gravels reduces the availability of breeding habitat for fish (such as salmon and trout) and macro-invertebrates that rely on such habitat for breeding.
- *Oxygen depletion and toxic impacts* - sediments trap and sometimes become attached to pollutants such as fertiliser, pesticides or heavy metals. When fertiliser in sediment is decomposed by microbes the result can be depletion of oxygen levels in water to below what can be tolerated by many important aquatic organisms. Toxic compounds like ammonia and hydrogen sulphide accumulate in sediment and can be harmful to aquatic organisms.
- *Disruption of the food web* – the many processes described above lead to potential changes in plant communities that in turn disrupt the natural food web.

Many urban and peri-urban developments in NZ are in close proximity to important estuaries (i.e. Christchurch, Nelson, Tauranga, Wellington, and Auckland) and, as a result, sedimentation from development is a threat to several important estuaries in NZ. The Okura Estuary and its Marine Reserve, for example, is significantly threatened by proposed urban development. There is pressure for new subdivisions in the Okura catchment because of its attractive location close to Auckland. Both the initial earthworks and subsequent construction can result in erosion with each rainfall event. With the establishment of subdivisions, in-stream erosion and sediment transport can be expected to continue because development results in more water impermeable surfaces and therefore higher flood flows. Where stormwater would have soaked into the ground and been filtered before reaching the estuary, stormwater would be channelled away from houses and infrastructure, so entraining and carrying sediments and other pollutants into the estuary.

Research by NIWA for the Auckland Regional Council and the two territorial authorities controlling subdivision and land use in the Okura catchment determined the maximum extent of soil exposure and therefore intensity and rate of subdivision development that could proceed without causing sediment inflows to exceed the threshold rate at which the estuary could digest the inputs without degrading ecological and cultural values. Plan-making processes and Environment Court appeals have recently resulted in science-based decisions to retain rural scale subdivision densities, with a four hectare minimum lot size in the east and an average of two hectares in the western catchment.

Another significant threat to freshwater streams is the erosion, turbidity and sedimentation associated with some aspects of forestry. Extensive plantation forests in NZ are the source of most timber and pulp production. While conversion of pastoral land to forests provides erosion control and other environmental benefits, the requirement for regular clear-felling and replanting every 20-40 years does create soil exposure and erosion risks. Of special concern is logging in the often mountainous terrain of along the east coast where clearing of forest in steep gullies in close proximity to streams is particularly threatening. Deforestation in such setting typically causes greater sediment loads than deforestation elsewhere.

5.2 Options to control diffuse source water quality issues

From a technical perspective there are essentially two solutions to the problems of nitrogen discharging through groundwater to freshwater bodies, and sedimentation of streams and estuaries:

- Dealing with the symptoms - this involves mitigating impacts after they occur with engineering solutions. Examples could involve such actions as removing nutrient enriched water from the bottom of lakes that have already been subject to high nitrate groundwater discharge (which is currently being investigated for Lake Okareka near Rotorua) or dredging estuaries that are already clogged with sediments (recently investigated for Matata lagoon/estuary but not pursued because of contaminant disposal risks).
- Dealing with the causes – this involves reducing that rate of groundwater loading with nitrates or surface water loading with sediments. In the case of nitrate loading from a pasture source, this could involve reducing the area that is grazed or reducing the intensity of grazing on land already in pasture within the catchments of ecologically sensitive lakes. In the case of nitrate loading from septic tank systems, replacement with sewerage systems, sealed on-site treatment such as composting toilets or other structural or maintenance measures are possible. In the case of sedimentation, this could involve re-

establishing forest cover on pasture or bare ground, for example through re-afforestation or riparian retirement.

While remedial actions can be taken to mitigate some of the adverse impacts of diffuse source emissions to water, remediation is typically expensive and only alleviates some of the impacts. Some ecological impacts are not easily reversible. Thus measures that prevent the sediment or nutrient loading in the first place are generally more cost-effective, have less adverse effects and are therefore preferable.

From a policy perspective, at least conceptually, any of the range of traditional 'non-market' regulatory or more 'market-based' instruments outlined in Text Box 1 could be used to deal with either nitrate loading of groundwater or sedimentation of surface water.

At least in principle, the following approaches could be used:

- Standards could be implemented in several ways. Output-based standards could be applied as standards limiting nitrate discharge, or sediment load. Practice based standards could be applied as standards limiting practices known to influence nitrate loading. For example, limits could be placed on livestock stocking rates to control nitrate loading or sediment loading could be controlled by establishing allowable levels of earthwork.
- Charge schemes could be implemented that involve a charge on performance (e.g. a charge per kg of nitrate loading or tonne of sediment load). Charging could also be implemented on some proxy for performance that is easily measurable (e.g. per head of livestock). Alternatively, charges could be based on some easily measurable practice (e.g. hectares logged) and differentiated on some easily observed site characteristics that are correlated with the environmental outcome of interest (e.g. rates per hectare differentiated by slope and proximity to stream).
- In a similar vein, incentive payments could be offered for outcomes (e.g. nitrate reduction of sediment load reduction) or proxies for these outcomes or practices correlated with outcomes.
- Tendering is a possible approach to incentive payments that can improve cost effectiveness. The approach requires differentiating among bids to take actions that reduce nitrate or sediment loads based on measurable factors correlated with the outcome of interest (e.g. prioritisation based on slope and proximity to vulnerable water bodies).
- Offset approaches could be implemented by:

  - Setting limits that preclude future conversion of land to pasture to control nitrate loading or setting limits on logging or urban development to control sediment loading;
  - Then in the case of nitrate loading, allowing some livestock farming development or intensification where compensating reductions in stocking rate or area in pasture was provided as an offset. In the case of sediment loading allowing logging where compensating reforestation or other mitigation is provided.
  - Such offset approaches would require offsetting actions based on formulas that would guarantee a "net" decrease in nitrate or sediment load.
- In principle, a tradeable permit approach could be applied to either nitrate or sediment loading.
  - This would require placing a cap on the level of nitrate or sediment load allowed from each source.
  - Alternatively, as a means of dealing with the challenge of measuring actual nitrate or sediment load, caps could be set on levels of allowable input use for some input correlated with nitrate or sediment load.
  - Each individual would be assigned an entitlement to load or use input in the form of tradeable permits.
  - Individuals could meet their cap by reducing loading, using their permitted amount or buying permits. Those able to reduce loading or input use below their cap level would be able to sell permits.

5.2.1 Current diffuse source water quality policy

NZ's institutional setting for dealing with diffuse source environmental issues has undergone significant change over the last two decades. The main NZ government legislation relating to management of land, water, air, and coastal environments in NZ is the Resource Management Act, 1991 (RMA). The Act replaced all or part of 75 statutes dealing with related issues. Two important features of the RMA are that controls are intended to be effects-based rather than activity-based and that environmental management responsibilities have been largely devolved to two levels of local government. A result is that most of the policies and rules that influence diffuse source pollutants are managed at the regional and local level. This includes: water quality and quantity regulation, soil conservation, landuse and subdivision controls and pest management policies.

While the particulars of resource management relevant to diffuse source environmental impacts vary across regional councils, some generalisations apply. First, any significant point source discharge to water generally requires council assessment and permission, and effects-based conditions will usually apply. This is typically the case for effluent ponds from dairy-sheds but not necessarily for other diffuse source discharges to water. Second, regional councils charge user fees for permits and there is a trend toward increasing fees to recover more of the costs of administration and water treatment programs. Third, regional councils are able to manage water quantity by requiring permits for water used for irrigation and food processing.

The RMA enables Councils to impose few conditions on resource use activities that are the primary source of agricultural diffuse source water quality externalities. In fact, the RMA includes an entitlement for agriculture and other existing land uses to continue that land use at its current intensity and type (sections 10 and 20A). This means that Plan rules alone cannot require the land use type or intensity to change, even if the adverse effects of the land use itself are severe. These entitlements to continue existing lawfully established activities do not (appear to) extend to "discharges" for which a significant adverse effect can be demonstrated. The ambiguity has impeded effective and confident use of the potential of the legislation to address diffuse source discharges.

These institutional settings have resulted in reluctance by Councils to establish site specific and significant standards to limit activities that are the source of diffuse source water quality externalities, and consequently this significantly limits the feasibility of quantity-based MBIs (tradeable permits and offsets).

A logical consequence of this institutional constraint is that most of the NZ government programs to deal with non-point sources of environmental impacts are incentive programs. A fairly typical example is the incentive payment approach for practices correlated with sediment and nitrate load reduction in the Lake Taupo area. The Lake Taupo Catchment Control Scheme presently has assets valued at $16.1 million and has facilitated 5,145 hectares of land retirement for erosion control (Environment Waikato, 2003). The public funds that will go into the proposed new nitrate reduction scheme incentive will facilitate land use change through tendering and other incentive mechanisms.

A range of NZ government incentive programs to encourage sustainable land management exist with many of the programs focused on facilitating more ecologically sustainable forest management. This includes the Nga Whenua Rahui Fund, the Nature Heritage Fund, The Queen Elizabeth II National Trust, and the

Biodiversity Condition Fund Programs. All of which offer incentive payments for agreement guaranteeing long-term protection of ecologically important indigenous ecosystems.

The devolution of environmental management responsibility to regional and local governments has generally meant that most environmental incentive programs to address water quality and soil conservation impacts resulting from agriculture are locally funded. Local funds are most commonly raised from property taxes (rates) with part of the cost paid by farmers and rural properties and part paid by urban households. Some funds for administration of local programs are also raised through user fees. However there appears to be some political aversion, similar to other jurisdictions, to environmental charges that involve differentiated charge rates based on outcome or input use levels. However there is currently little if any use of environmental charges that involve differentiated charge rates based on outcome or input use levels.

In addition, there is enormous variability in the size and economic mass of the various Councils, and therefore their ability and willingness to fund incentive programmes. Those Regional Councils that retained large shareholdings in privatised regional Port Companies generate large dividends, some of which is deployed to environmental incentive programmes (e.g. Bay of Plenty Environmental Enhancement Fund averages $1M/annum).

5.3 Evaluating Opportunities and Challenges to using MBIs for Diffuse Source Water Quality Issues in NZ

Having described the biophysical nature of the two diffuse source water quality issues in NZ, the options for dealing with the issues, and the institutional context, we now have the background to evaluate opportunities and challenges to implementing economics instruments. This involves using the framework described in section 3 of this report as a basis for outlining:

- Opportunities to use or modify existing institutional mechanisms defining treatment of diffuse source environmental issues in NZ in ways that would facilitate development of more effective diffuse source water quality policy; and
- Opportunities to overcome key characteristics of biophysical processes and markets that limit potential for cost-effective, or environmentally reliable implementation of MBIs to diffuse source water quality issues.

5.3.1 Potential for institutional changes to support water quality objectives

The discussion of the current policy settings above suggests that the current institutional setting has resulted in relatively few effective standards related to diffuse source water quality in NZ. A result is that the predominant approach to the issue is incentive payments. This policy approach has significant limitations. Generally, the potential of incentive approaches is limited by the available budget and limited responsiveness to price signals. There is also a probability with incentive payments of creating perverse incentives against environmental improvements, whereby landholders delay undertaking environmental improvements until they receive payment to do so. As an approach to protecting ecologically important resources like Lake Taupo, incentive approaches are unlikely to be sufficient.

It follows from the arguments presented in section 3 that MBIs can represent a way of decreasing the cost of complying with effective environmental standards. However, guaranteeing that the valuable and relatively pristine ecological assets that characterise NZ remain in good condition will require some institutional change to enable or require establishment of more stringent standards or regulation.

The institutional setting in NZ includes several mechanisms that could be used to facilitate creation of more effective standards limiting activities that are the source of diffuse source water quality externalities. Discharges from diffuse sources can, in principle, be regulated by local councils under the resource consents process of the RMA. The provision can involve requirements for mitigation under defined conditions. For example, in the Lake Taupo catchment regulation prohibiting any further intensification of livestock stocking is being proposed under the RMA by the Regional Council (Hickman, 2004).

One approach is to require the mandatory development of catchment-based water plans that provide standards for environmental outcomes and/or best practice performance. Because the RMA is an enabling rather than a prescriptive statute, it was not expected to function effectively until the full suite of National Policy Statements and Regional Plans was in place. To date less than one third of the necessary water and catchment plans are complete. Such legislative amendments and institutional changes could be augmented with new provisions to enable offsets and tradeable permit schemes for diffuse source discharges of contaminants that would enhance compliance flexibility and decrease compliance costs. Recent legal advice has confirmed the constraints in the existing RMA which limit options for trading permits between properties (Richmond, 2004).

More fundamentally, changes in standards relating to activities influencing water quality are possible at the national government level. In particular, new standards can be introduced under section 43 of the RMA including standards for contaminants, water quality, level or flow, air quality and soil quality in relation to discharge of contaminants. NZ is currently in the process of developing standards for air quality and landfill gas under these provisions and it would be possible to develop standards for water quality in a similar way.

Another fundamental change in the RMA that could enable development of more effective standards would be modification of RMA entitlements for agriculture and other existing land uses to continue that land use at its current intensity and type (sections 10 and 20A). Modifying these provisions to clearly define a set of "discharges" to which the provisions do not apply because a significant adverse effect can be demonstrated. Clarifying the ambiguity between provisions for continued land use and intensity and provision prohibiting discharges with demonstrably significant adverse effects should allow Councils to more effectively and confidently develop local standards and plans to address diffuse source discharges.

In peri-urban settings where sediment loading from development is an issue, existing development laws generally allow for the development of controls that limit activities impacting sediment loading. In these settings, further development of such controls could represent an effective standard on which quantity-based MBIs of the offset type can be built.

5.4 Dealing with Barriers to Cost Effectiveness and Reliability in the introduction of MBIs

Section 3 of this report outlined how a number of attributes of biophysical processes and markets can limit potential for cost saving and/or environmental reliability of MBIs. One key finding was that neither price nor quantity-based instruments that focus on outcomes are typically applicable to diffuse sources of discharge to water. This is because measuring outcomes in most diffuse source settings is typically infeasible (at a reasonable cost compared to the benefit). This "monitoring problem" is certainly an issue with both nitrate and sediment loading from diffuse sources in NZ.

The difficulty in measuring actual outcomes of interest for diffuse source water quality issues has made application of MBIs to such issues particularly challenging. As discussed in section 3, there are essentially three ways to overcome the "monitoring problem":

- One approach is to focus on a "proxy" – such as an input where the use of the input is correlated with the outcome of interest – rather than the outcome of interest *per se*. The Dutch manure policy outlined in section 3 is an example of this approach. The policy involves standards and charges set on total (animal plus mineral) fertiliser input.
- Another approach involves focussing on practices that are correlated with the outcome of interest. This could involve things like limiting times and locations that grazing can take place in proximity to threatened lakes, requiring waterways to be fenced off, requiring establishment of riparian buffer strips, and/or limiting logging in riparian buffers of some prescribed width.
- A final approach involves: setting standards, charges, or incentive payments on some easily measurable proxy and where appropriate, these arrangements can be varied by zone or other locational attributes correlated with outcomes of interest. The zone based charge scheme for salinity in the River Murray outlined in section 3 is an example.

In very broad terms, world-wide experience suggests that the practice-focussed approach is often chosen for ease of implementation and environmental reliability. However, practice-focussed approaches have often failed to produce the anticipated cost savings. This is because the flexibility of the landowners or developers is often curtailed. In particular, a practice-focussed approach limits the opportunity for landowners and developers to seek out individualised solutions or take advantage of specialised information about lower cost solutions.

Using inputs as a proxy, in the form of stocking rates to address the nitrate issue in freshwater lakes, could be a feasible basis for implementing a charging approach. Alternatively, if changes in the institutional setting allowed the use of standards, stocking rates could be used as the basis for an offset approach.

In the case of sediment loading from urbanisation, forestry activities, the third approach outlined above, despite its limitations, holds some attractions. Much of the cost savings potential of outcome based charges, incentive payments, tendering and offset systems could be realised with a focus on hectares of deforestation or afforestation but differentiated based on location attributes that are correlated with sediment threat (e.g. slope). This approach is already in use in the East Coast Forestry Program in the context of a tendering program (see Text Box 5).

An important prerequisite to implementation of any of the three approaches outlined above is a good scientific understanding of relevant environmental processes. Depending on which of the approaches used, this can require capacity to model relationships between environmental outcomes, input use levels, practices and/or variations in outcomes across location with some confidence. An important implication is that implementing MBIs approaches often requires a significant investment in environmental modelling.

5.5 Transition approaches to implementation of MBIs

It is inevitable that there will be some political aversion to new policies like higher environmental standards, differentiated charge rates based on outcome or input use levels, and additional development restrictions. Such aversion does not have to stifle change. Standards that limit how resources can be used have been introduced in the past and are now widely accepted. For example, the right to sell farm products are typically conditional on meeting certain food safety and trade related restrictions on production.

Successful introduction of change can be facilitated with several strategies. One useful approach involves a transition to higher standards with an initial period of reduced compliance burden, followed by a period of gradually increasing standards and penalties for non-compliance. The approach can reduce the perceived threat of changes by allowing individuals to gain some degree of comfort with the production modification required before there are serious sanctions for non-compliance.

An additional strategy worth considering in NZ would be to implement a series of limited duration, limited geographic extent pilot projects that trial the use of MBIs. This is an approach that is currently being pursued in Australia through an initiative jointly sponsored by the Commonwealth ministers for the Environment and Agriculture. The initiative is currently sponsoring 10 pilot projects being lead by Commonwealth, State and Local government agencies that involve investigation of various tendering, offset, tradeable permit and other market based approaches to dealing with diffuse source salinity, water quality and biodiversity issues (http://napswq.gov.au/about/mbi_projects.html). Pilot projects were chosen on a competitive tender basis with the selection criteria oriented towards favouring projects that promised to offer insights into how the most important impediments to successful implementation of MBIs can be overcome. This same sort of approach could be used in NZ to build a base of experience and expertise.

6 SUMMARY AND CONCLUSIONS

This report has considered the many facets of MBIs and developed a framework for evaluating the potential of these instruments to manage diffuse source water quality issues in NZ. To lend concreteness to this analysis, the focus has been on two particular water quality concerns in NZ:

- Diffuse source agricultural drainage rich in nutrients that is entering groundwater and then discharging to freshwater lakes. With NZ freshwater lakes in the central North Island volcanic plateau, nitrate is of particular concern as it is now considered to be the limiting nutrient and therefore the main cause of increasing chlorophyll and decreasing water clarity. A certain amount of nitrogen has always entered NZ lakes from rainwater and undeveloped land. It is the increased loading that has resulted primarily from development of grazing based farming systems after European settlement that is leading to declines in water quality.
- Sedimentation of streams and estuaries as a result of surface runoff from urban development and forestry. Many urban and peri-urban developments in NZ are in close proximity to important estuaries (i.e. Christchurch, Nelson, Tauranga, Wellington, and Auckland) and as a result, sedimentation from development is a threat to several important estuaries in NZ. The Okura Estuary and its Marine Reserve, for example, is significantly threatened by proposed urban development. Another significant threat to freshwater streams is the erosion, turbidity and sedimentation associated with some aspects of forestry. Extensive plantation forests in NZ are the source of most timber and pulp production. While conversion of pastoral land to forests provides erosion control and other environmental benefits, the requirement for regular clear-felling and replanting every 20-40 years does create soil exposure and erosion risks.

A range of policy options could at least in principle be used to deal with these issues. More traditional 'non-market' regulatory approaches that could be used include:

- Standards limiting nitrate discharge, or sediment load, standards limiting practices known to influence nitrate or sediment loading, or standard limiting levels of inputs that relate to sediment or nitrate loading;
- Charges or incentive payments on performance (e.g. a charge per kg of nitrate loading or tonne of sediment load) or some proxy for performance that is easily measurable (e.g. per head of livestock); and

- Differentiating charges or incentive payments based on some easily observed characteristics that are correlated with the environmental outcome of interest (e.g. rates per hectare differentiated by slope and proximity to stream to differentiate among rates of payments for actions to control sediment).

Three MBIs are of particular interest:

- Tendering could be used to deliver incentive payments. This would involve soliciting bids to take actions to reduce nitrate or sediment loading and choosing among bids based on levels of measurable factors correlated with the outcome of interest (e.g. prioritisation of bids to reforest based on slope and proximity to vulnerable water bodies).
- Offset approaches could be implemented by setting limits that preclude future conversion of land to pasture to control nitrate loading or setting limits on logging or urban development to control sediment loading. Then some level of actions such as livestock farming intensification, urban development of timbering could be allowed where compensating mitigation is provided based on formulas that would guarantee a "net" decrease in nitrate or sediment load.
- A tradeable permit approach could be applied to either nitrate or sediment loading by placing a cap on the level of nitrate of sediment load allowed from each source or on levels of allowable input use for some input correlated with nitrate or sediment load. Those able to reduce loading or input use below their entitlement would be able to sell permits. Experience from the USA suggests that point to non-point trade could be introduced. Caution must be employed, but there are opportunities for associations to take more responsibility for water quality.

Approaches to overcome challenges in implementing MBIs were identified by evaluating:

- Opportunities to use or modify existing institutional mechanisms defining treatment of diffuse source environmental issues in NZ in ways that would facilitate development of more effective diffuse source water quality policy; and
- Opportunities to overcome key characteristics of biophysical processes and markets that limit potential for cost-effective, or environmentally reliable implementation of MBIs to diffuse source water quality issues.

An understanding of NZ's institutional setting for dealing with diffuse source environmental issues was reviewed as this is a prerequisite to understanding constraints to implementation of MBIs. The main NZ government legislation relating to management of land, water, air, and coastal environments in NZ is the Resource Management Act, 1991 (RMA). Important features of the RMA are that:

- Environmental management responsibilities have been largely devolved to two levels of local government.
- While significant point source discharge to water generally requires council assessment and permission, this is not typically the case for diffuse source discharges to water.
- The devolution of environmental management responsibility to regional and local governments has generally meant that most environmental incentive programs to address water quality and soil conservation impacts resulting from agriculture are locally funded. Local funds are most commonly raised from property taxes (rates) with part of the cost paid by farmers and rural properties and part paid by urban households.
- The RMA includes an entitlement for agriculture and other existing land uses to continue that land use at its current intensity and type (sections 10 and 20A). These entitlements to continue existing lawfully established activities do not (appear to) extend to "discharges" for which a significant adverse effect can be demonstrated. The ambiguity may be impeding effective and confident use of the potential of the legislation to address diffuse source discharges.
- A logical consequence of this institutional constraint is that, most of the NZ government programs to deal with non-point sources of environmental impacts are incentive programs.

The conclusion from review of potential institutional constraints was that guaranteeing that NZ's valuable ecological assets remain in good condition will require institutional change to enable more stringent regulation as part of the process of increasing the use of MBIs. Several mechanisms could be used to facilitate creation of more effective standards limiting activities that are the source of diffuse source water quality externalities. They include:

- Discharges from diffuse sources could in principle be regulated by local councils under the resource consents process of the RMA through requirements for mitigation under defined conditions. For example, in the Lake Taupo catchment regulation prohibiting any further intensification of livestock stocking is being proposed.
- Development of catchment-based water plans could be required that provide standards for environmental outcomes and best-practice performance. This

approach was intended in the original RMA enabling legislation but has only been pursued by three Councils.

- Either of these above listed approaches could in turn be augmented by offset or tradeable permit approaches for agricultural and forestry diffuse source water quality issues.
- More fundamentally, changes in standards relating to activities influencing water quality are possible at the national government level under section 43 of the RMA. NZ is currently in the process of developing standards for air quality and landfill gas under these provisions and it would be possible to develop standards for water quality in a similar way.
- Another fundamental change in the RMA worth considering would involve clarifying the ambiguity between provisions for continued land use and intensity at current levels (provisions 10 and 20a) and provision prohibiting discharges with demonstrably significant adverse effects. This could enable development of more effective standards and support MBI development.
- In peri-urban settings where sediment loading from development is an issue, existing development laws generally allow for the development of controls that limit activities impacting sediment loading. In this setting, further development of such controls could represent an effective standard on which quantity-based MBIs of the offset type can be built.

This report outlined how a number of attributes of biophysical processes and markets can limit potential for cost saving and/or environmental reliability of MBIs. One key finding was that the difficulty in measuring actual outcomes of interest for diffuse source water quality issues has made the application of MBIs to such issues particularly challenging. There are essentially three ways to overcome the "monitoring problem":

- Focus on practices that are correlated with the outcome of interest;
- Focus on a "proxy" – an input the use of which is correlated with the output of interest – rather than the outcome of interest *per se*; or
- Setting standards or charges, or incentive payments on some easily measurable proxy input or practice, and differentiating standard stringency, charge or incentive payment level across locations based on location attributes correlated with outcome of interest.

Focussing on practices is not recommended for NZ. While the approach has been used elsewhere for ease of implementation and environmental reliability, it has often failed to produce the anticipated cost savings because the flexibility of the landowners or developers is often curtailed.

Using inputs as a proxy, in the form of stocking rates, urban development limits, or logging limits could be a feasible basis for implementing a charging systems or standards. In particular, much of the cost savings potential of outcome based charges, incentive payments, tendering and offset systems could be realised with a focus on inputs such as hectares of deforestation or units of livestock per hectare but with differentiation based on location attributes that are correlated with sediment or nitrate threat level (e.g. slope).

An important prerequisite to the implementation of any of the three approaches outlined above is a good understanding of the market segments and a good understanding of the science underlying the relevant environmental processes. Depending on which of these approaches is used, this can require capacity to model relationships between environmental outcomes, input use levels, practices and/or variations in outcomes across location with some confidence. An important implication is that in many instances, implementing MBIs may require significant investments in environmental investigations, modelling, and monitoring.

Finally, it was noted that there will inevitably be some political aversion to new policies like higher environmental standards, differentiated charge rates based on outcome or input use levels, and additional development restrictions. Successful introduction of change can be facilitated with several strategies. One useful approach involves a transition to higher standards with an initial period of reduced compliance burden, followed by a period of gradually increasing standards and penalties for non-compliance.

An additional strategy worth considering in NZ is implementation of a series of pilot projects designed to test the approach and build experience in the use of MBIs. This approach is currently being pursued in Australia. The experience suggests that this approach could represents an effective, and politically feasible way to gaining understanding and acceptance of how MBI design and implementation influences environmental and cost effectiveness. An advantage is that best practice can be developed using pilots before new policies are developed on a larger scale.

REFERENCES

Ayres, I. and J. Braithewaite (1992). *Responsive Regulation: Transcending the Deregulation Debate*. Oxford University Press.

Baumol, W.J. and Oates, W.E. 1998. *The Theory of Environmental Policy*. Cambridge University Press, Cambridge.

Bennett, J. and Blamey, R. (2001). *The Choice Modelling Approach to Environmental Valuation*. Edward Elgar, Cheltenham.

Binning, C. and M. Young (1997). *Motivating People: Using Management Agreements to Conserve Remnant Vegetation*. Research Report 1, CSIRO Wildlife and Ecology.

Bockstael, N.E., Hannemann, W.M. and Kling, C.L. (1987). Estimating the Value of Water Quality Improvements in a Recreational Demand Framework. *Water Resources Research*, 23(5): 951-960.

Bright, M., D. Young, A. Hamilton, J. Connor and M. Young (2002). Application of Market Based Instruments in Natural Resource Management for South Australia. Rural Solutions SA.

Connor, Jeff and Melissa Bright (2003). "Which Market Based Instrument Approaches to Natural Resource Management?" Australian Agricultural Resource Economics Society 2003.

Connor, Jeff and Darla Hatton MacDonald (2002). "Incentives, Policy and Voluntary Approaches to Improve Natural Resource Management in the Onkaparinga Catchment". CSIRO Land and Water Report.

Cox, James (2003) Personal communications with Dr. James Cox, principle research scientist, CSIRO Land and Water, Adelaide, South Australia.

Diamond, P.A. and Hausman, J.A. (1994). Contingent Valuation: Is Some Number Better than No Number? *Journal of Economic Perspectives*, 8(4): 45-64.

Dwyer, Janet, Rosy Eaton, Andrew Farmer, David Baldock, Paul Withers, and Paul Silcock (2002) *Policy mechanisms for the control of diffuse agricultural pollution with particular reference to grant aid.* English nature research reports, number 455.

ELI (2002). Banks and Fees: The Status of Off-site Wetland Mitigation in the United States. Environmental Law Institute.

Environment Waikato (2003) Protecting Lake Taupo. http://www.ew.govt.nz/policyandplans/taupo/index.htm

Environmental Economics Series. NSW National Parks and Wildlife Service, Hurstville.

Environomics (1999). *A Summary of U.S. Effluent Trading and Offset Projects.* Prepared for: Dr. Mahesh Podar, U.S. Environmental Protection Agency, Office of Water.

Godden, D. and Vernon, D. (2003). *Theoretical Issues in Using Offsets for Managing Biodiversity*. Paper presented at the Annual Conference of the Australian Agricultural and Resource Economics Society, Fremantle, February.

Gillespie, R. (1997). *Economic Value and Regional Economic Impact Minnamurra Rainforest Centre, Budderoo National Park*. NPWS.

Hahn, R. (1989). Economic Prescriptions for Environmental Problems: How the Patient Followed the Doctor's Order. Journal of Economic Perspectives 3: 95-114.

Hailu, A. and S. Schilizzi (2003). Learning in a `Basket of Crabs' An Agent-Based Computational Model of Repeated Conservation Auctions, Discussion Paper, School of Agriculture and Resource Economics, The University of Western Australia, 2003.

Hanley, N. J.F. Shogren and B. White (1997). Environmental Economics in Theory and in Practice. Macmillan.

Hickman, M. (2004). Personal Communication with Mr. Matthew Hickman, Senior Adviser, NZ Ministry for the Environment - Manatü Mö Te Taiao.

James, D. (1997) *Environmental Incentives: Australian Experience with Economic Instruments for Environmental Management*. Consultancy report commissioned by Environment Australia © Commonwealth of Australia, 1997.

Lockwood, M., M. Morrison and Jacinta MacKay (2003). Policy Instruments for Sustainable Landscapes. Charles Sturt University, Working Paper.

Kahneman, D. and Knetsch, J.L. (1992). Valuing Public Goods: The Purchase of Moral Satisfaction. *Journal of Environmental Economics and Management*, 22: 57-70.

Kerr, G. (2003). Personal Communication with Dr. G. Kerr, Lincoln University.

Klang, J. (2000) . Trading in Watersheds. EPA's National Environmental Innovations Symposium. http://www.epa.gov/opei/symposium/m-water2.htm

Mäler, K.-G. (1974). Environmental Economics: A Theoretical Inquiry. The Johns Hopkins University Press, Baltimore.

Ministry of Agriculture and Forestry 2003: In Brief - Agriculture, Forestry and Horticulture. New Zealand Government IBBN 0 478 07669 X (June 2003).

Moore, W.B. and McCarl, B.A. (1986). Off-site Costs of Soil Erosion in the Willamette Valley. *Western Journal Of Agricultural Economics*.

Morrison, M. (2003). Using Offset Banking to Control Nonpoint Source Pollution in Urban Areas. Presented at Environmental Economics Network Meeting, Australian National University, May.

Mundy, G.N. (1999) A Review of Nitrogen Research with Irrigated Pastures in Northern Victoria. Kyabram Dairy Centre, Institute of Sustainable Irrigated Agriculture. Natural Resources and Environment, Victoria.

NAP Working Group on Market-based Instruments (2002) Investigating New Approaches: A review of Natural Resource Management Pilots and Programs in Australia that Use Market-based Instruments. National Action Plan for Salinity and Water Quality, 2002.

National Competition Council, (2003). *The 2003 National Competition Policy Assessment Framework for Water Reform*, National Competition Council, Canberra.

NSW EPA (2001) Load-based Licensing: *A Fairer System that Rewards Cleaner Industry*. NSW Environmental Protection Authority, Sydney.

Pearce, D and Markandya, A. (1989). The Measurement of Environmental Benefits. OECD, Paris.

Quinn, J. and M. A. Meleason (2003). Trends in nuisance periphyton at National Rivers Water Quality Network sites 1989–2001. http://www.niwa.cri.nz/ncwr/nrwqn

Randall, A. (2003). Market Based Instruments - International Patterns of Adoption, Remaining Challenges and Emerging Approaches" Market-Based Instrument Workshop, Canberra, October.

Raux, C. (2002). The Use of Transferable Permits in the Transport Sector. *Implementing Domestic Tradeable Permits: Recent Developments and Future Challenges*. OECD.

Richmond, C. (2004). Personal Communication with Mr. Chris Richmond, Senior Policy Analyst (Freshwater), Conservation Policy Division, Department of Conservation - Te Papa Atawbai.

Salzman, J. and Ruhl, J.B. (2000). Currencies and the Commodification of Environmental Law. *Stanford Law Review*, 53(3): 607-694.

Sinner, J. and G. Salmon (2003). *Creating Economic Incentives for Sustainable Development.* A report to the New Zealand Business Council for Sustainable Development. Ecologic Foundation.

Smith, S (2002). Ex Post Evaluations of Tradeable Permits Programmes. *Implementing Domestic Tradeable Permits: Recent Developments and Future Challenges*. OECD.

Stavins, Robert (2000) *Experience with Market-Based Environmental Policy Instruments*. Resources for the Future Discussion Paper 00-09. January 2000. Resources for the Future, Washington, D.C.

Sterner, T (2003). Policy Instruments for Environmental and Natural Resource Management. Washington, Resources for the Future.

Stoneham, G., V. Chaudhri, A. Ha, and L. Strppazzon. (2002). *Auctions for Conservation Contracts: An Empirical Examination of Victoria's BushTender Trial*. Paper presented at the 46th Annual Australian Agricultural and Resource Economic Society Conference, Canberra, Feb. 2002.

Sunraysia Rural Water Authority. (2002) *Refinement of the River salinity impact zoning system.* The Irrigator, volume 1, no 16, July 2002, pp 6-17.

Walker, C. and Greer, L. (1992) *The Economic Costs Associated With Lost Recreation Benefits Due To Blue-Green Algae In NSW: Three Case Studies*. Report Commissioned By The Blue-Green Algae Task Force, Technical Services Division, NSW Department of Water Resources.

Young, M.D. and D. Hatton MacDonald (2003). " An Opportunity to Improve Water Trading in the South East Catchment of South Australia". *Water Policy* 5(2): 127-146.

P:\PERU\NZ Environment Ministry\Final Report as Sent\as sent Feb 4, 2004\NZ Economic Instruments 26.doc