Improving Water Quality through Environmental Policies and Farm Management: an Environmental Economics Analysis of Dairy Farming in Karapiro Catchment

A thesis submitted in partial fulfilment of the requirements for the degree of Doctor of Philosophy at the University of Waikato

By

Thiagarajah Ramilan

Department of Economics

University of Waikato, New Zealand

2008

Abstract

This research explores the environmental and economic implications of nitrogen discharge abatement from dairy farms. This thesis develops a framework to analyse agri-environmental policies using bio economic modelling. A novel approach has been developed using farm survey data for catchment scale modelling and policy analysis.

Policy analysis is facilitated by various modelling techniques and software. Geographic Information Systems (GIS) are used to convert data from multiple sources to a common platform for policy analysis. Whole dairy farm system simulations coupled with a Metamodel built on the Overseer software are used to establish the relationship between farm economic returns and nitrogen discharges. This relationship is used to derive abatement costs and resolve policy implementation challenges. Data Envelopment Analysis is used to develop Environmental economic efficiency measures. Finally a stylized model is used to determine spatially optimal riparian buffer strips.

The results show that the differences in abatement costs and environmental efficiency between farming systems are significant. The adverse effects of information asymmetry can be effectively minimized by adopting differentiated incentives and target monitoring. Riparian buffers are a cost effective abatement tool that complement abatement at the intensive margin. Clear understanding of farm heterogeneity will help to design effective policies.

Techniques for the measurement of policy impact have been successfully developed and add significantly to our knowledge of the underlying relationships. The use of simulated data for agri-environmental policy analysis is versatile and is expected to have several valuable applications. These methods can be applied to other geographic areas and research domains. This thesis provides useful tools for policy makers seeking to develop empirically informed agri environmental policy.

Acknowledgements

First, I would like to express my eternal gratitude and profound appreciation to Professor Frank Scrimgeour, my mentor and supervisor, for his generous support, encouragement, outstanding guidance and inspiration at all stages of my research. Winter 2004, was the first time I met him, who later, welcomed me to Waikato Management School, an excellent research environment, as a PhD student and arranged funding for my study. I would also like to thank my co-supervisor Dr Dan Marsh for his advice and support. He generously spent time in reviewing thesis chapters and gave me a wealth of valuable comments, information and encouragement, and facilitated funding at the final stage of the thesis.

Besides my supervisors, Professor Riccardo Scarpa was a mine of information on empirical analysis and always willing to find time to assist. I am very grateful to the Chairperson of the Department of Economics, Professor Mark Holmes and the administrative managers Maria Fitzgerald and Leonie Pope for their kind facilitation in numerous ways. Financial support from the Department of Economics full scholarship throughout this study is gratefully acknowledged. Without which this study would not have been possible.

I would like to express my appreciation to DairyNZ, particularly to Matthew Newman for farm survey data and Gil Levy for the Whole Farm Model. I am very grateful to Fonterra especially to Barry Harris and Zachary Ward for arranging access to data. I also wish to thank many people in Agresearch and Environment Waikato for sharing with me a great deal of technical expertise and information; particularly Dr Reece Hill of Environment Waikato for soil classification and Dr David Wheeler of Agresearch for the Overseer software. My sincere thanks to Sanjay Waduwa of NIWA for teaching Geospatial analytical skills and Professor Chuda Basnett of Management Systems for Visual Basic Applications.

My thanks go to all the staff of WMS information technology for exceptional technical support, especially to Monica van Oostrom for formatting this thesis and Andrew Gera for technical and software support. I am very grateful to Jaki Heta of the Dean's office for her kind assistance in arranging supervisory meetings.

I thank my colleagues and friends in Orchard Park for the great support and good atmosphere, especially to Chris Hector and Kevin Old for helping me to appreciate Kiwi culture and Kartic Gupta for help in Stata programming. I am also very grateful to the Library staff particularly to inter loan services.

Finally, this thesis would not have been completed without the love, patience and inspiration from my wife Bavani and daughter Sona. I wish to extend my profound thanks to them, who always understood the challenges and hardships of this journey and rendered the greatest support, encouragement and tolerance.

There are many people I could not list here and I would like to thank for helping to my research and to my personal development during these years.

Data and software access

Data and software were obtained for this study under agreements designed to give effect to security and confidentiality. In accordance with these agreements the processed information has been used only for the production of this thesis and related scholarly publications. When required, specific confidentiality agreements were signed by the Chief supervisor of the study Professor Frank Scrimgeour on behalf of the University of Waikato and officers of the respective institutions. The data/software and institutions involved are as follows.

Data/ Software	Institution
Economic Survey of New Zealand Dairy Farmers	DairyNZ
Whole Farm Model	DairyNZ
On farm milk production	Fonterra Co-operative Group Limited
Agribase	Agriquality
Overseer®	Agresearch

Disclaimer

The results presented in this study are the work of the author, not the respective institutions.

Table of Contents

List of Tables

List of Figures

List of Acronyms

AE	Allocative Efficiency
BMP	Best Management Practices
CE	Cost Efficiency
CRS	Constant Returns to Scale
DE	Differential Evolution
DEA	Data Envelopment Analysis
DMU	Decision Making Unit
EBOP	Environment Bay of Plenty
EE	Environmental Efficiency
EW	Environment Waikato
GA	Genetic Algorithm
GAMS	General Algebraic Modelling System
GIS	Geographic Information System
LP	Linear Programming
LUC	Land Use Capability
MAF	Ministry of Agriculture and Forestry
MFE	Ministry for the Environment
NIWA	National Institute of Water & Atmospheric Research
NLP	Non Linear Programming
NZLRI	New Zealand Land Resource Inventory
NZSC	New Zealand Soil Classification
PCE	Parliamentary Commissioner for the Environment
REC	River Environment Classification
SFA	Stochastic Frontier Analysis
SKSE	Seasonal Kendall Slope Estimator
TE	Technical Efficiency
TLA	Territorial Local Authorities
UVIS	Uni Variate Imputation Systems
VBA	Visual Basic Applications
VRS	Variable Returns to Scale
WFM	Whole Farm Model

1. Introduction to thesis

1.1 Motivation

Declining water quality is widely considered to be one of the most important environmental issues facing New Zealand with farming being perceived as a major cause. Waikato region residents reported that water pollution was easily their most important environmental concern in each of four attitude surveys conducted by Environment Waikato. Technical and regulatory mechanisms to reduce water pollution; especially nonpoint source pollution from agriculture are the focus of an intensive research effort both in New Zealand and internationally. The research described in this thesis should help policy makers and farmers to identify the most cost effective options for achieving any given improvement in water quality.

Management of water quality is often inherently complex due to the large number of agents involved and because of the importance of spatial variability. At the same time, both cost and environmental impact must be evaluated simultaneously in order to evaluate alternative policies. Given these complexities, the integrated analysis and modelling described in this thesis provides a useful contribution to policy development.

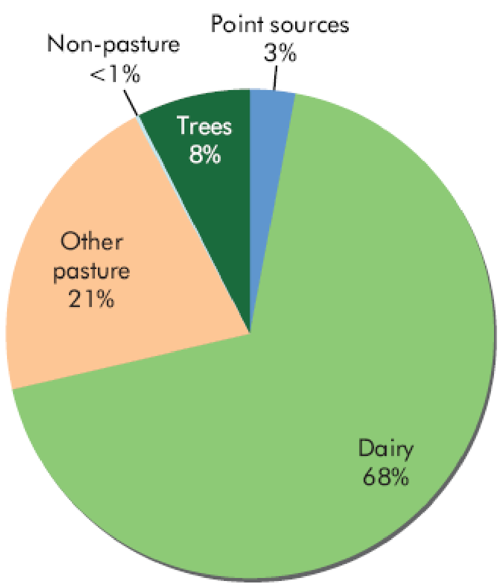
Nitrogen is a key nutrient for pasture productivity and farm nitrogen discharge is the major nonpoint source pollution. Dairy farming in New Zealand is an intensive form of land use, often involving high stocking rates and nitrogen fertiliser application rates, which elevate nitrogen concentrations in water (Ledgard, Penno, & Sporsen, 1999). Nitrate from cow urine patches is a major contributor to the leaching loss in grazed dairy pastures (Di & Cameron, 2002). The urine patches and mobile nature of nitrogen compared to other nutrients, leads to the potential for significant losses of nitrogen into water. Nitrogen contamination can make water undesirable for drinking; affect fisheries and reduce the aesthetic and recreational value of water. The Waikato River catchment has received much attention in recent years because of rising nitrogen levels with

68% of this has being attributed to nitrogen discharges from dairy farms (Figure 1.1).

According to the Parliamentary Commissioner for the Environment (2004), cow numbers and nitrogen fertiliser use per hectare have risen by 19 percent and 162 percent respectively during the decade from 1994 to 2004. Environment Waikato, the regional council for the Waikato region, reports that annual nitrogen fertiliser use in the region increased by 84 percent over the five year period 1997/98 to 2002/03 from 68 kilograms to 125 kilograms per farmed hectare.

Dairy farming has been an important component of the Waikato economy for many decades and its importance has increased over time with the conversion of additional land to dairy production. Warm climate, fertile soils and ample water supply make the region ideal for its flourishing dairy industry. Dairying is the dominant sector in the Waikato economy with respect to produce value and employment and uses 32% of total agricultural land. The dairy industry provides 17% of jobs, 24% of gross regional product and 35% of national dairy production (Hughes, 2007).

Figure 1.1 Sources of nitrogen to the Waikato River



Source: Environment Waikato (2008)[1]

[1] Based on modelled data provided by the by the National Institute of Water & Atmospheric Research. Technical details are provided in Elliot *et al*. (2005).

Pastoral nonpoint pollution has been largely free from regulation in New Zealand. But in recent times community pressure for better water quality has emerged (Brodnax, 2006) and has become a matter of public and political debate. Water quality concerns have triggered scientific research towards managing nitrogen in water. Scientific evidence indicates that farms may reduce their nitrogen discharge by reducing the intensity of production and changing management strategies. In the Waikato, polices for changing farm practices related to water quality have so far largely relied on voluntary measures. However Buchan, Meister, & Giera (2006) cited New Zealand (Bayfield & Meister, 2005) and overseas (Gunningham, Grabosky, & Sinclair, 2004) experience, that shows that voluntary approaches, while effective to a degree, are generally not enough to achieve the community's desired rate of progress without strong regulatory underpinning. Therefore in the future, dairy farmers are likely to face stricter environmental regulations either under standards or incentive based programmes. They are also aware that environmental issues will be among the greatest challenges they face in the near future. The dairy industry has identified addressing nitrogen losses into water as one of the key priority areas and realised the importance of the development of quantitative approaches to environmental management (Dairy Environment Review Group, 2006).

Falling water quality results in part from the failure of the market to allocate the socially desirable level of resources to abatement. Given that dairy farms are the engine of economic prosperity in the region, any proposed policies should minimise the adverse economic impact of achieving environmental goals. Understanding and quantifying the responsiveness of farms to alternative policies in different economic and geo-physical environments is essential to the development of cost effective solutions to nitrogen abatement. Economics can play an important role in guiding policies that dairy farms may face in order to pursue economic and environmental sustainability. This study developed quantitative models to generate information, guide policy development and monitoring of outcomes for dairy farms given their economic and environmental importance. This is consistent with the interests of the dairy industry and assists policy makers such as regional councils in developing cost effective programmes to improve water quality.

1.2 Research area and hypothesis

The empirical analysis is carried out in the catchment that includes the middle part of the Waikato River catchment from Lake Arapuni to Karapiro dam, plus contributing tributaries. This research area is referred to as the Karapiro catchment throughout the thesis. Being an enclosed geographic entity catchments have often been considered to be the most appropriate spatial unit for analysis of the interaction between agriculture and water quality. Lake Karapiro and Arapuni are identified as waters of national importance for tourism by the Ministry of Tourism. This catchment has been identified as a high priority areas for nutrient management (Environment Waikato, 2005c) and consists of approximately 400 dairy farms. Nitrogen discharges are the leading cause of water quality impairment in Lake Karapiro and its tributaries (Brodnax, 2006; Environment Waikato, 2005a). Although water quality is formed at the catchment scale, many of the immediate environmental impacts of agriculture occur as a result of decisions made at farm level. So this study focuses on economic and environmental variables at farm level.

The thesis examines implicit hypotheses that existing regional survey data can effectively be used to represent populations of farms in a small geographic area; heterogeneity exists among farms in the catchment in terms of economic and environmental impact and efficiency; information asymmetry and cost of policy implementation can be reduced by effective monitoring and interception technology is an effective tool to reduce nitrogen discharges.

1.3 Method of analysis

Environmental policy issues stem from agricultural production and its interface with the environment. Griffin and Bromley (1982) initiated the analysis of agri environmental policies based on non-point production functions. Various authors provide evidence of the effect of farm heterogeneity on the cost effectiveness of environmental policies (Martinez & Albiac, 2006; Newell & Stavins, 2003; Wossink, Lansink, & Struik, 2001). Thus designing policies for improving water quality is an empirical issue and requires knowledge of the nature of production and the fate of pollution. The complex interrelationships and time lags involved in various dimensions of agricultural production systems limit the use of experimental field trials in gaining understanding of the nature of these relationships. Bio-economic modelling is capable of tackling a component of this sort of information problem (Bennett, 2005). It provides useful insights on the relationships, which influence the heterogeneous set of farms and environmental resources.

However bio-economic models require intensive data on production, management, pollution and financial information. In New Zealand there is no such comprehensive data source that provides such data with spatial references. Therefore this study developed a spatial micro-simulation model was developed to generate a virtual population of dairy farms in the catchment. An integrated bio-economic model was also constructed including a mathematical programming model, meta model, Geographic Information Systems (GIS), using both the Overseer nutrient budget model and the DairyNZ’s Whole Farm Model and virtual population data. Although the Whole Farm Model and Overseer models are not specifically designed for evaluating the effect of policies, this study demonstrated that these are useful tools with modest data requirements and yet sufficiently robust to accurately describe the nature of dairy production and associated nitrogen discharges for catchment wide policy analysis. The model based estimates are used to analyse various policy issues such as estimation of abatement costs, standards and taxes, farm efficiency and riparian buffers at catchment scale.

1.4 Research objectives

The overall objective of this research is to gain insight into the environmental and economic implications of nitrogen discharge abatement. More specifically, the following issues will be addressed.

1. Development of a comprehensive framework for analysing proposed policies
2. Generation of a comprehensive data base to analyse catchment wide impacts
3. Exploring the impact of various policies on different farming systems
4. Exploring specific challenges of policy implementation including monitoring
5. Defining and measuring environmental and economic efficiency of farms

1.5 Chapter outline of thesis

This study consists of 9 chapters including this introductory chapter. The remaining parts of the thesis are organized as follows.

Chapter 2 describes the rationale for environmental policy; outlines the economic concepts behind environmental pollution; provides an overview of environmental policy for water quality improvement and discusses the importance of bio-economic modelling in capturing policy implications.

Chapter 3 characterises the catchment including the level of nitrogen pollution within water ways. This provides a platform for identifying policy challenges and analysis of environmental policy.

In chapter 4 a spatial micro-simulation model is developed to combine geo referenced data on farms in the catchment with economic variables from the annual dairy farm survey to generate comprehensive data for analysis.

Chapter 5 establishes the relationship between profit and pollution using optimisation techniques and derives abatement costs for farming systems by extending the utility of DairyNZ’s Whole Farm Model.

Chapter 6 examines the issues related to implementing environmental policy from a property rights, monitoring and transaction cost perspective. A model for effective monitoring to minimise information asymmetry is set up and empirically applied.

Chapter 7 defines and measures the environmental and economic performances of dairy farms using Data Envelopment Analysis (DEA). Then the variations in efficiency are explained based on characteristics that are hypothesised to influence environmental efficiency.

Chapter 8 develops a stylised model for optimum implementation of riparian fencing, which is followed by an empirical analysis using a virtual population of farms and analyses the impacts of potential land use changes.

Finally chapter 9 summarizes the findings and considers implications for policy and future research.

1.6 Contribution of thesis

This thesis contributes to the applied literature on many fronts, both methodologically and in relation to the study site. A comprehensive dataset is generated to model spatial heterogeneity at the level of individual decision making units. An applied analytical framework is developed for evaluating and implementing water quality policies on different farming systems. A stylised model for policy analysis is constructed. This facilitates precise consideration of riparian buffers on New Zealand dairy farms.

2. Agri-environmental policies for improving water quality

2.1 Introduction

Agri- environmental policies can have a significant influence on farming practices and water quality. Nitrogen pollution in water is an important environmental issue in the Waikato region, where the majority of manageable nitrogen discharges come from the agricultural sector particularly dairy farming. These discharges can be characterised as an environmental externality[1] associated with agricultural production as they contribute to deterioration of water quality. Social emphasis on water quality has resulted in the need for more rigorous integration of this externality with economic objectives in the management of farming systems. Therefore environmental externalities have been analysed with the view to design effective and efficient policies, which tend to internalise those externalities.

Griffin and Bromley (1982) initiated the analysis of agricultural pollution as a nonpoint externality by developing a theoretical framework, in which production externalities are expressed with a continuously differentiable nonpoint production function. They showed that under certainty, efficient environmental policies can be created based on correctly defined nonpoint production functions. Environmental policies can be regulatory, voluntary and market based. In the Waikato, policies for changing farm practices related to water quality have so far relied on effluent management rules and voluntary adoption of new practices. Policy tools to promote voluntary adoption include extension education, technical assistance and costs sharing.

This chapter explains the rationale for environmental policy; outlines the economic concepts behind environmental pollution; provides an overview of policy instrument

[1] An externality is any action that affects the welfare of an individual or group without direct payment or compensation

types highlighting generic properties and challenges; discusses the application of polices in the presence of heterogeneity and concludes with a review of future directions.

2.2 Rationale for environmental policy

While modern agriculture is very productive, its negative effects on the environment have become increasingly visible. Many of these are the result of practices aimed at reducing the per unit costs of production. This has resulted in increased intensity, more specialized production, and increased emissions of substances with negative effects on surrounding eco-systems. Input use intensity has increased in New Zealand agriculture over the years. According to the Parliamentary Commissioner for the Environment (2004), cow numbers and nitrogen fertiliser use per hectare have risen by 19% and 162 % respectively during the decade from 1994 to 2004. In the same period volume of milk production per hectare has risen by 34%. Intensification of production evolves in response to incentives. The New Zealand dairy industry's focus is to drive productivity improvements on dairy farms to maintain and enhance competitiveness. Productivity improvement fuelled by intensification, technological innovation and improved management systems is seen as imperative, if the industry is to remain profitable and sustainable. Severe environmental damage as a result of technological advancement is attributed not to bad science but to inadequate policies, institutions and management systems (Zilberman, Khanna, & Lipper, 1997).

In an economy the primary function of the market is to improve the well being of society. This occurs through the trading of goods and services. The pricing of these goods and services leads to the allocation of resources that best suits the demands, and hence satisfaction of society. In reality, however the price of goods and services does not always reflect scarcity due to imperfections in the market causing market failure. The failure to recognize and apportion the external costs of nitrogen discharge as a result of farming is an externality problem. This leads to sub-optimal economic outcomes. The key features of nitrogen pollution as an externality are the time lag involved between cause and effect and difficulties in identifying the producer (Pretty et al., 2000).

Inadequately specified property rights are the major cause of market failure (Randall, 1987). Efficient markets require restrictions on resource use in addition to rights that specify ownership. In the absence of such specifications, property rights are said to be attenuated. Attenuated property rights ultimately lead to over and under utilisation of resources, leaving society less well off than it might be with the existence of non-attenuated property rights.

When property rights are attenuated the full cost of land use activities are not captured. For instance when the environmental cost of nitrogen fertiliser is not reflected in its price, it leads to over consumption and consequently an increased level of nitrogen in water. Farms which do not bear the cost of damage they cause off farm can be described as “Free riders”(Legg, 2003). In contrast non attenuated property rights would ensure the full costs and benefits of consumption would accrue to the purchaser of the goods and services traded. Goods and services over which enforceable non-attenuated rights have been specified are said to be excludable and rival in consumption (Randall, 1987). Ideally, the role of environmental policy in correcting market failure is to specify non-attenuated property rights for all resources involved in the production process. This will lead to trade and result in prices that reflect the true price of the resources. Solutions of this nature to correct market failure were first proposed by Coase (1960) in his seminal article “The problem of social cost”. The Coase theorem states that under ideal circumstances, when polluters and pollutees bargain, the equilibrium level of pollution is independent of the allocation of property rights. He argued that provided property rights are well defined and transaction costs are low, the efficient level of pollution can be arrived at through bargaining between the owner of an environmental asset and respective users, irrespective of who has the initial property rights. The ideal circumstances infer perfect information about costs and benefits and the absence of transactions costs. Suppose, for example, that the "pollutee" has all the property rights to the environment, in the absence of any negotiation between the polluter and the pollutee, the former is not allowed to pollute, to do so would infringe upon the pollutee’s property rights. Such efficiency cannot be attained for most environmental problems related to agriculture due to the public good nature of the polluted resources, where non exclusiveness or non rivalry in consumption often make it impractical to specify property rights.

Even though the efficiency of markets may be improved by better specification of property rights, there are costs associated with the implementation and administration of the legislation required to specify the new set of rights. In addition, there are costs associated with market transactions. Only in cases where the benefits of trade outweigh these administration and transaction costs, would markets improve net social benefit. The highly variable nature of nitrogen pollution, the high costs of collecting data and non rivalry in consumption of the benefits of nitrogen discharge reduction makes market transactions unlikely.

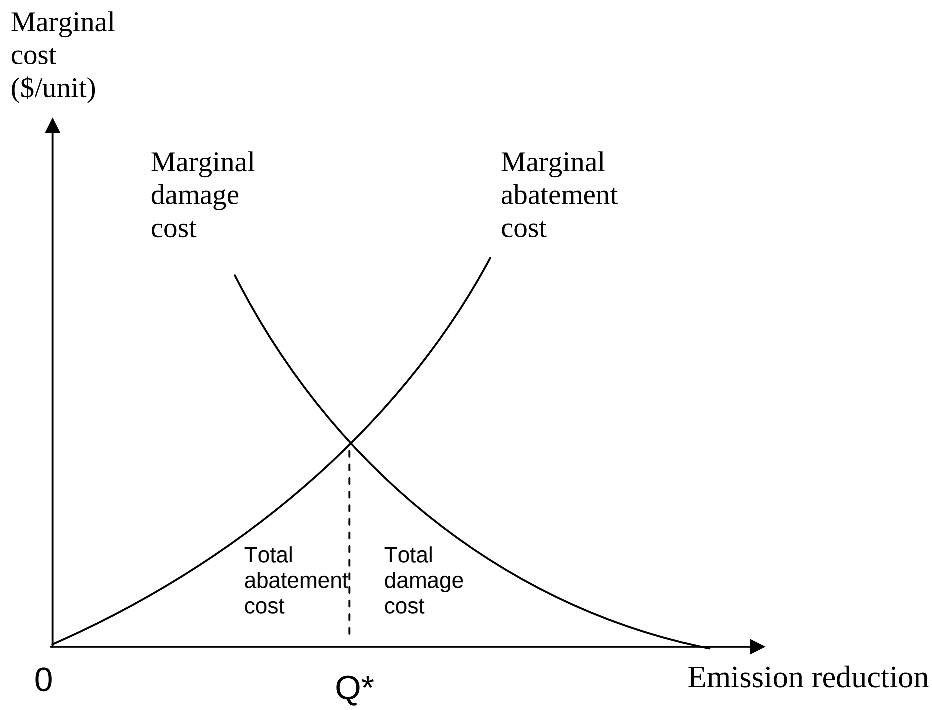
When market transactions fail to bring effective solutions, policy instruments may play a role in correcting market failure. However in the case of nonpoint pollution, in the presence of imperfect knowledge about nitrogen discharge and transportation, maximisation of social welfare is hardly ever achieved. In other words it is difficult to reach first best solutions. In this circumstance the best that can be achieved through policy intervention in the market is to move society from a sub optimal state to a better sub optimal state. Thus the role of policy is to influence the behaviour of economic agents to improve water quality and thus social welfare. The instruments discussed below attempt to internalise negative externalities caused by poorly defined property rights over the use of resources.

2.3 Conceptual framework for analysis of environmental policy

A framework for analysing environmental policy must consider cost and benefits of pollution abatement. Pollution abatement costs are defined as the cost of reducing nitrogen discharge to a particular level. The abatement cost function maps the minimum cost of achieving a given level abatement, where the abatement level is measured as the reduction in kilograms of nitrogen discharge below the unconstrained level. Figure 2.1graphs the marginal abatement cost (MAC) and marginal benefit of pollution abatement. Marginal abatement cost is positively sloped since the cost of pollution abatement increases at increasing rate. The reduction in economic values due to ambient pollution levels is referred to as economic damage and is reflected by the marginal damage curve. Because of the assimilative capacity of water bodies, at low levels of nitrogen discharge the damage is quite small while at higher levels the damage is significant. Therefore the marginal damage caused by each unit of nitrogen discharge increases with the amount emitted. This results in a negatively sloped marginal damage cost curve, which captures the trend that the benefits of pollution control increase at a decreasing rate. Economic theory suggests that the socially optimal level of pollution abatement is at a point where the marginal abatement cost and marginal damage is equal (at point Q*) (Hanley, Shogren, & White, 1997).

Figure 2.1 illustrates this conceptual framework.

Figure 2.1 Efficient level of pollution abatement



A common problem in determining the optimal pollution level is that the marginal damage cost function and marginal abatement cost are often unknown or contentious, so achieving an economically optimal level of pollution is often not practical. Given that the damage caused by nitrogen discharges into water is often not well known, a socially optimal level of pollution cannot be determined and policy instruments are not able to establish an optimal outcome. In economic analysis, therefore, a cost effectiveness framework is often used. Griffin & Bromley (1982) proposed cost effective ways of meeting an exogenous target for environmental quality. So a social planner can choose the policy instrument that minimises abatement and transaction costs. The empirical part of the chapter 5 quantifies and compares the relative differences in abatement cost under varying nitrogen discharge levels and tax scenarios in different farming systems[2].

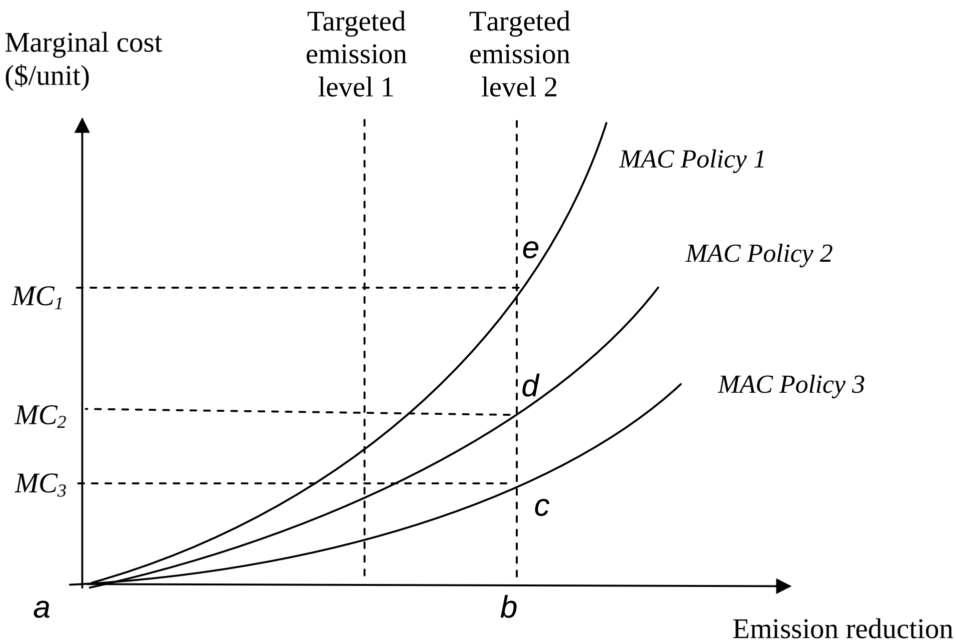
So exogenous specification of target pollution levels and the cost efficiency of achieving this under various policies are graphically illustrated in Figure 2.2. The limit on nitrogen discharges have often been politically or bureaucratically resolved based on scientific findings. For example Environment Waikato has proposed capping nitrogen levels in the Lake Taupo and reducing manageable discharges in the lake's catchment by 20 percent. Scientists regard this reduction as the minimum required in order to maintain the present level of water quality in the lake (Environment Waikato, 2005b). But the cost of achieving a given target can vary considerably depending on the policy adopted. Exogenous specification of pollution levels and the cost efficiency of achieving this under various policies are graphically illustrated in Figure 2.2. The total cost of abatement for policy 1 is represented by the area encompassed by *abe*, whereas the cost for policy 3 (*abc*) is much less. Similarly the marginal cost for the last unit abated is far higher for policy 1 (MC_1) than for policy 3 (MC_3).

The damage cost of water quality is often valued using a number of approaches that enable estimation of marginal benefit of abatement. Damage to water quality can be valued as a function of demand by individuals, municipalities, or industry based on the cost of water treatment, lack of biodiversity, eutrophication, decrease in recreation activities and so on (Bontems, Rotillon, & Turpin, 2005). The greater the demand for services such as recreation or industrial use, the greater their value and the greater the economic damages if impaired by pollution.

---

[2] Transaction costs are not modelled in this study

Figure 2.2 Cost effectiveness of policies

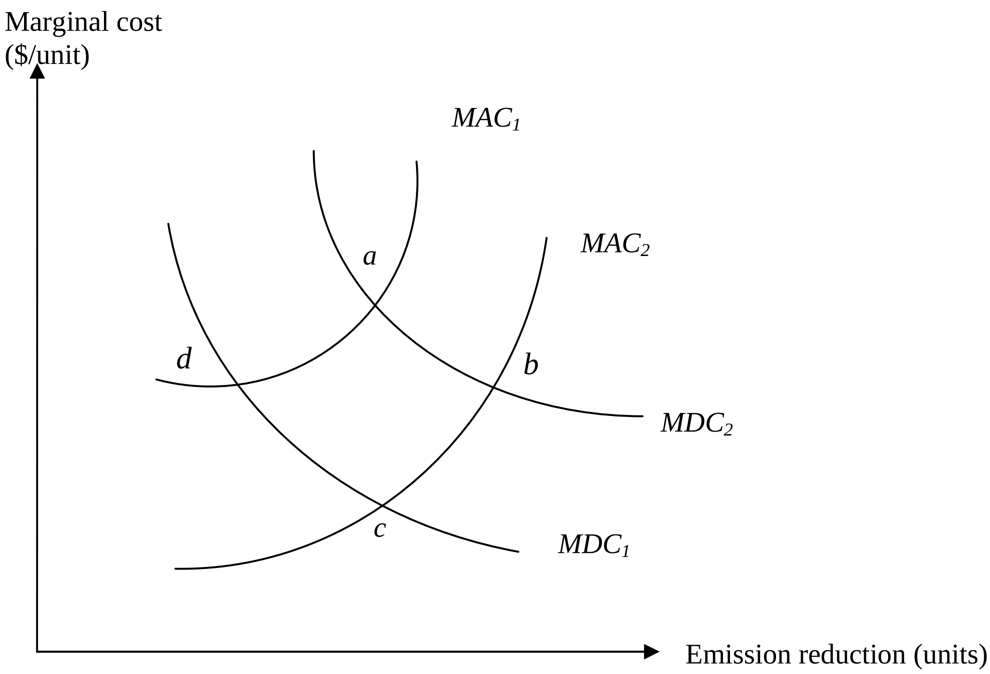


**MAC*- Marginal abatement cost

2.3.1 Spatial dimension

The quantity and cost of abatement of nonpoint source water pollution can vary considerably depending on spatial variation in soil, topographic, hydrologic and landscape features of fields and on the transport path for a pollutant (Qiu & Prato, 1999). The development of environmental policies for agriculture is thus complicated by spatial, temporal, and technological heterogeneity (Weersink, Livernois, Shogren, & Shortle, 1998). For instance, a farm with a low level of nitrogen discharge but located next to a water body may impose greater environmental damage than a farm with a high discharge level located further from the water body. When this occurs, cost effectiveness requires policy instruments to be targeted to individual farms. In addition, technological differences among farms may alter production levels, pollution potential and marginal abatement costs for nitrogen discharges.

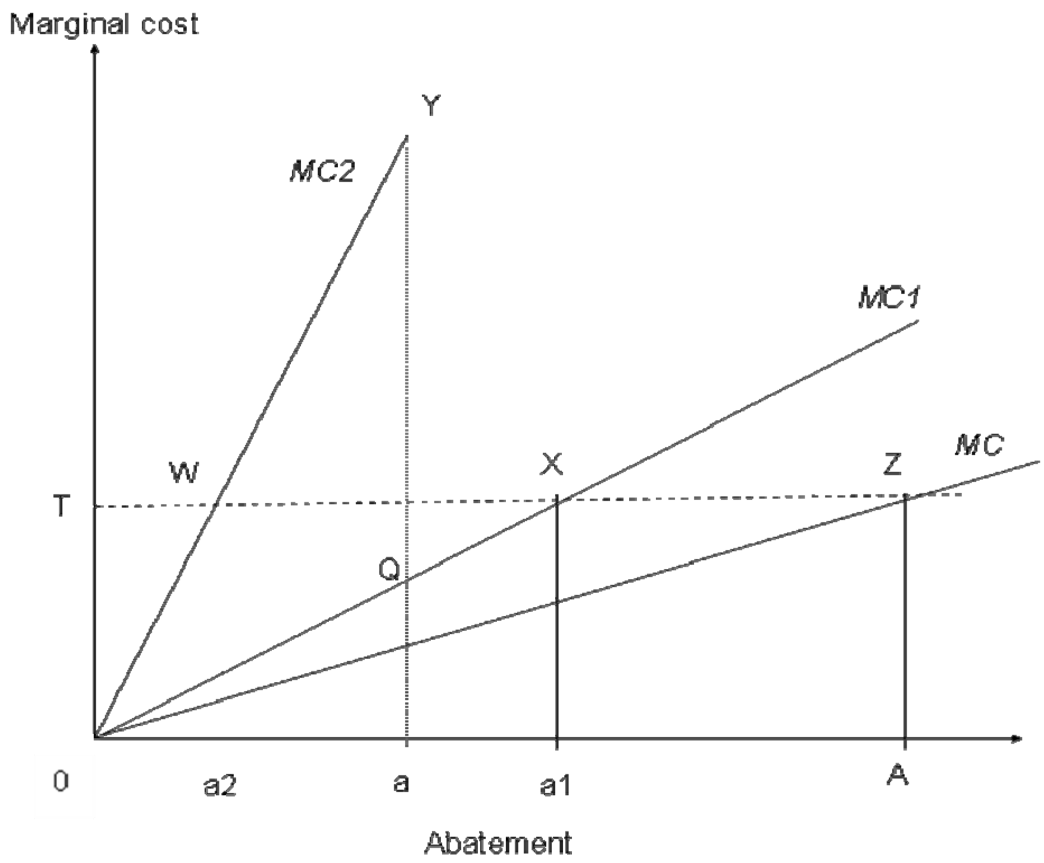
Figure 2.3 Spatial variations of agricultural production and pollution



Rygnestad (2000) illustrated the spatial variation of agricultural production and environmental impacts, using figures adapted from Baumol and Oates (1988); see Figure 2.3). Curves 1 and 2 describe different farms. He showed that A, B, C and D can be efficient solutions for different farms with spatial variation in both agricultural production and environmental vulnerability. In a situation where agricultural production and pollution potential varies, it is not possible to find a single optimal pollution level via a uniform environmental policy. However environmental polices are seldom spatially differentiated due to informational and monitoring costs. Instead uniform policies are implemented even though ignoring spatial differences in a catchment likely to increase the cost of abatement (Qiu & Prato, 1999).

The heterogeneity of farm abatement costs is of fundamental importance in the selection of optimal mitigation policy instruments (Newell & Stavins, 2003). Optimal environmental policy concentrates its efforts on those farms where abatement costs are low (Macho-Stadler & Perez-Castrillo, 2006); thus farm centric knowledge of abatement costs will facilitate selection of the most efficient option for nitrogen management. Chapter 5 considers the existence of spatial variability among farms in the catchment in terms of economic and environmental impact.

Figure 2.4 Abatement allocation



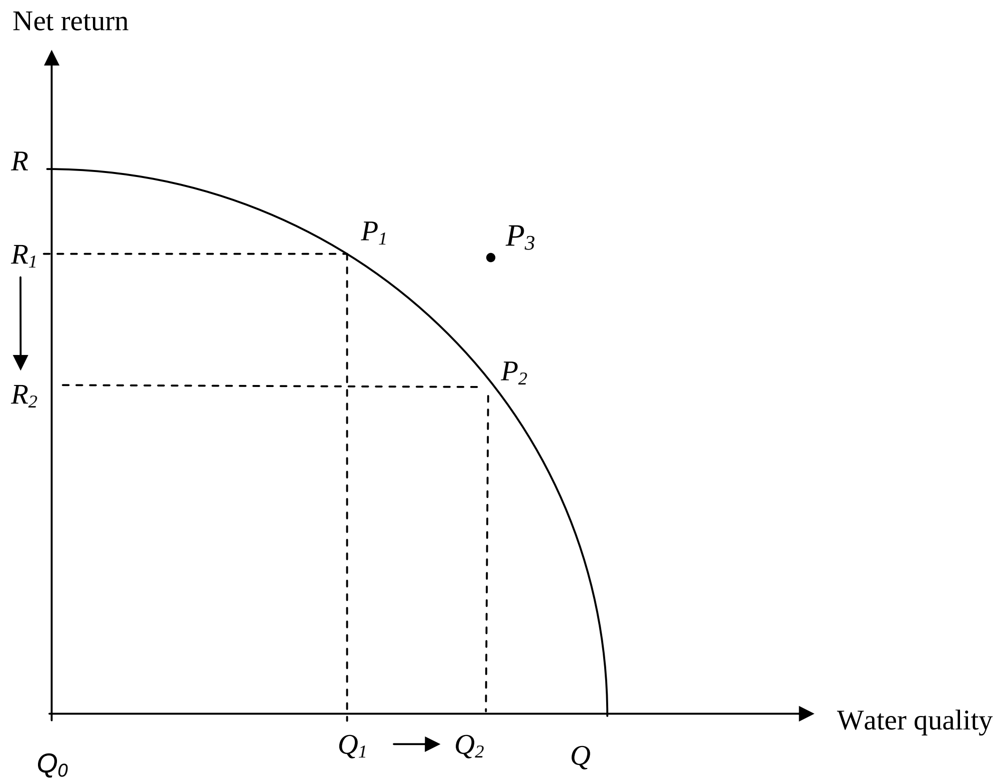
Source: Sterner (2003)

Figure 2.4 shows the allocation of abatement between two farms. Abatement could either be mandated at a target level for each farm or a market based policy instrument such as a charge T could be used, giving abatement of a_1 and a_2 respectively. Under the trading scenario, farm 1 with lower abatement costs will voluntarily undertake more abatement to create and sell surplus emissions credits or rights to farm 2, which finds that it is cheaper to buy these on the market than to incur the costs of abatement. Total abatement costs to the farms are the areas under the marginal cost curves (*MC1* and *MC2*) in Figure 2.4. When reductions in abatement are equal, costs are the sum of two triangles (*OaY)*. Under the market based allocation scenario, the farm with the lower abatement cost is responsible for most of the abatement, and aggregate costs are significantly lower (Oa_1X+ Oa_2W). In the presence of big differences in the abatement costs function of the two farms, an equalisation of the marginal costs of abatement can reduce aggregate costs significantly.

2.3.2 Policy impact on farms

The impact of policy on farms can be examined using the tradeoff between farm profitability (net returns) and water quality (Ribaudo, 1999). In Figure 2.5, movement along the curve represents changes in inputs at a given technology to achieve increasing levels of water quality. For instance, higher levels of water quality protection may necessitate a move away from conventional practices to ones using less nitrogen fertiliser and fewer cows in on dairy farms (from P_1 to P_2).

Figure 2.5 Farm level tradeoff between net returns and water quality



Under conventional technology better levels of water quality can be achieved only with a loss of net returns reflecting the fact that pollution control is typically costly.

However adoption of Best Management Practices (BMPs) may enable achievement of enhanced water quality without much loss of returns (point P_3).

2.4 Environmental policy instruments

A large number of policy instruments for reducing the environmental impacts of agricultural production can be identified. They can be categorized into the following groups; regulation, economic instruments, farm management choices, decentralised policies such as liability rules, property rights and moral suasion. Table 2.1 summarizes the key features of such policies and they are discussed in more detail below.

2.4.1 Direct regulation

Direct regulation can be implemented through either design standards regulating the way farmers produce and manage their resources or performance standards regulating the quantity of observable pollution resulting from production. Standards have an advantage over taxes because when standards are implemented, farms only bear the whole cost of abatement. Weersink, Livernois, Shogren, & Shortle (1998) showed that regulation or the threat of regulations on pollution levels can be used as a stick to promote the adoption of BMPs and uptake of other economic instruments such as tradable emission permits. Regulations depending on their design may either encourage or stifle research and development. Subsequent sections discuss the merits and application of standards in comparison with other policies, in more detail. When standards are implemented in a uniform manner in a catchment, they are likely to have differential effects on farms and so are likely to be opposed by property rights advocates (Qiu & Prato, 1999). The Waikato regional plan rule introduced last year is an example of direct regulation (Environment Waikato, 2007c). It requires any farm applying more than 60 kg of nitrogen fertiliser per hectare per year, or applying any fertiliser to the effluent irrigation area to have a nutrient management plan and to apply for a resource consent.

Table 2.1 Possible options for addressing nonpoint pollution in agriculture

Standards/ direct regulation

- Restrictions on stocking rate, fertiliser application, emission discharge levels
- Mandatory use of pollution control practices, compulsory adoption of environmental management plans
    - Require clearly defined legislation

Charges/ Subsidies/ Tenders

- Charges on inputs/outputs-
    - Require ability to discriminate charge/subsidy for polluting farms in affected region on inputs outputs that have a direct relationship with pollution levels
- Charges on estimated nitrogen emissions
    - Require accurate relationship between estimated emissions and readily observable inputs and site characteristics data
- Ambient charges
    - Readily monitored resource quality affected by a relatively homogeneous group of producers within a small area short time lag between emissions and environmental effects
- Cost-sharing or other subsidies for inputs or practices that reduce pollution
- Land retirement/land use change subsidies
- State grants to competitive discharge reduction practices
- Contracts involving adoption of conservation or nutrient management practices based on auctions

Tradable Permits

- Implemented on inputs or estimated emissions
    - Clearly-defined, homogeneous input related to environmental problem and sufficient number of polluters to establish market

Decentralized Policies

- Liability Rules
    - Applicable to infrequent polluting events with clear cause and effect link involving a few parties
- Non-Compliance Fees
    - Homogenous group of polluters with understood links between behaviour and environmental damages and
    - Fees related to damage need to be communicated to polluters ex ante
- Property Rights Definition
    - Privately-owned resource where institutional restrictions have prevented markets for environmental amenities

Moral suasion and education

- Dissemination of knowledge about environmental damages and code of practice for nutrient management etc
    - Farmers willingness to adopt

Collated from Shortle & Horan (2001), (Weersink, Livernois, Shogren, & Shortle (1998) and Gunningham & Sinclair (2005).

2.4.2 Price based instruments

Emission charges

Emission charges are fees levied on the nitrogen discharged from individual farms. The diffuse nature of nitrogen discharges makes estimating individual farm nitrogen discharges difficult and expensive unless modelling approaches are used. But there is likely to be political resistance to emission charges unless discharges can be clearly established.

Ambient charges

Segerson (1988) also Xepapadeas (1992) proposed a system for non-point pollution control based on the level of aggregate emission into the water body. When the targeted emission level is exceeded in the water body all farmers are charged. The major advantage of the system is easy monitoring as it does not require monitoring at the individual farm level. However, Weersink, Livernois, Shogren, & Shortle (1998) showed this strategy is best suited to environmental problems in small catchments with relatively homogeneous farms with readily monitored water quality and short time lags between polluting activities and pollution delivery. Under ambient, emission and input taxes, farms not only pay the tax but also bear the abatement cost (Qiu & Prato, 1999). In theory emission taxes can be made revenue neutral by reducing other taxes.

Input taxes

Griffin & Bromley (1982) and Shortle & Dunn(1986) illustrated the use of policy instruments based on management such as taxing input use and output produced. In contrast to emission based instruments they are easy to implement, but available evidence suggests that input taxes tend to generate revenue for the environmental authority rather than alter producer behaviour. For instance, Swinton and Clark (1994) found enterprise mix was unchanged in the range between a 121% and 780% increase in nitrogen input price. Giraldez and Fox (1995) assessed the cost of reducing nitrogen inputs in the USA by the same amount either through tax or a nitrogen ceiling. While the required level of a nitrogen tax was $49.70, use of a ceiling achieved the same reduction at a fraction of the cost ($1.81). Martinez and Albiac (2004) also found that standards outperformed taxes in their study; 1.20 Euro/kg of nitrogen tax resulted in 21.5 million Euros profit and 990 tons of nitrogen leaching. Meanwhile a nitrogen standard resulted in 23.8 million Euros profit and 634 tons of nitrogen leaching. Wu, Teague, Mapp, & Bernado (1995) reported that nitrogen use restrictions resulted in a 25 % reduction in nitrogen discharge and income loss of 16%, while a nitrogen use tax achieved the same level of reduction with 49% of income loss.

The effectiveness of input charges on reducing input use depends on the size of the tax and the proportion of total production costs made up by the input. The inelasticity of farm input demand implies that high tax levels are required to cause the desired reduction in input use. Empirical studies suggest that fertilizer use is very inelastic to price changes for example a one hundred percent increase in nitrogen price, reduced nitrogen use only by 2% (Johnson, Adams, & Perry, 1991). They found that the elasticity of nitrogen losses with respect to nitrogen price was less than 0.1. While higher tax rates would be required to induce dairy farmers to substantially reduce nitrogen use, this may not be politically feasible (Hefland & House, 1995). A further complication is caused by the spatial variability of input use on environmental impact, which means that the input tax should in principle vary with location and application method (Zilberman, Khanna, & Lipper, 1997). Also, taxing inputs like fertiliser, does not take account of any environmentally oriented farm management practices that may limit nutrient runoff. A mere promising approach may be to use taxes to create a price differential between conventional fertilisers like urea and other environmentally preferred types of chemicals such as nitrification inhibitors.

2.4.3 Incentives

Rather than imposing a charge on inputs associated with a polluting residual, an alternative often used is to offer financial incentives. Financial assistance normally takes the form of grants, loans and tax allowances. These subsidies are advocated as a means of easing the financial burden on farmers thus increasing the probability of adoption of measures for which the social benefits are greater than private abatement costs. Incentives are used to encourage the adoption of better management practices and nutrient budgeting in the state of Virginia (OECD, 2007). In New Zealand assistance is focussed on fencing of riparian margins (Ministry for the Environment, 2003). If a financial assistance programme is targeted to more pollution prone areas, social benefits can be increased. Even though discriminatory input taxes are infeasible, targeted input subsidies are possible and could be based on a region's effective demand for environmental quality. Incentives can be used as flipside of taxes to promote environmentally friendly practices. An obvious problem with incentives schemes is that they often face budgetary constraints.

2.4.4 Tenders

Even though financial incentives to all land holders may be perceived to be equitable, such a policy would not be cost effective would be poorly targeted, and would probably have high transaction costs (Gunningham & Sinclair, 2005). According to Gordon (2003) tenders involve a system of auctioned grants for the voluntary improvement of environmental quality. This approach respects the existing allocation of property rights and does not seek to impose obligations without compensation. Farmers who usually have the best knowledge about their own property, tender on a competitive basis, outlining proposed management actions and the payment they would require from the regulator to undertake them. The regulator assesses each tender in terms of its cost and the anticipated impact on nutrient emission levels and using cost minimization as the basis for ranking tenders. This approach can result in significant cost savings and economic efficiency gains. These gains enable either greater total emissions reduction to be achieved with a limited budget or the freeing up of regulator resources for other programme. An example of environmental tendering is “EcoTender” implemented in Australia, which aims to achieve environmental benefits such as salinity reduction and water quality improvement (Eigenraam, Strappazzon, Lansdell, Beverly, & Stoneham, 2007).

2.4.5 Decentralised policies

Decentralised policies allow farms to resolve environmental problems through negotiation or through definition of rights responsibilities under the legal system.

Liability rules make polluters liable for the damages they cause. Under strict liability, a farm would be held liable for any damage resulting from its production behaviour regardless of the care taken to avoid the damage (Weersink, Livernois, Shogren, & Shortle, 1998). Under a negligence rule, farms are only liable if appropriate actions were not taken to prevent the damage. Under liability rules expectation of paying damages, motivates changes in farm behaviour. Provided farms have accurate expectations of costs and they correspond to actual damage costs, liability rules can be an efficient means of obtaining socially desired farm practices. However the application of liability rules tends to be limited to situations, in which a small number of participants are involved and the incidence of pollution is infrequent. Further the polluter needs to be specifically identified and the cause and effect relationship between the pollution and damage must be established. In agriculture these conditions can only be met for a limited number of localised situations such as accidental spillage of dairy farm effluent. Since most environmental problems in agriculture are more diffuse, liability rules do not provide a comprehensive solution but can be used as a complementary mechanism alongside other instruments.

2.4.6 Moral suasion

Moral suasion is based on the premise that farmers will voluntarily adopt pollution control practices if they are fully informed about the economic and environmental consequences of their current practices. The most common instrument of this kind in New Zealand to address environment problems in farming is the provision of information on environmentally benign farm management practices. For example DairyNZ has produced a number of advisory publications to promote nutrient management, design of standoff and feed pads and disposal of farm dairy effluent and the New Zealand Fertiliser Manufacturers Research Association produced a Code of Practice for Nutrient Management. Environment Waikato, through environmental education programmes also attempt to raise awareness within the community about appropriate land management practices and streamside management that reduces nonpoint source discharges. Horizons Regional Council (2007) developed an applied resource management strategy for farmers, which aims to enable better understanding of the farming operation and its impact on the environment.

Despite the increasing emphasis on these voluntary compliance programmes, their effectiveness is uncertain. While moral suasion approaches rank extremely highly in terms of political acceptability, they are reportedly very low on environmental effectiveness (Gunningham & Sinclair, 2005). Adoption is more likely by farmers who feel a greater sense of moral responsibility, not necessarily by those contributing to environmental problems. Environment Waikato (2005a) also remains concerned that the BMPs being promoted will not reduce nutrient loads enough to offset the impact of land intensification. Given current levels of non compliance with the resource consent rules of the regional plan for dairy effluent (Ministry for the Environment, 2007), voluntary adoption is unlikely when these practices are costly. When pollution abatement is costly and does not provide other benefits farms tend not to invest in pollution control technology or abate emissions in the absence of regulations (Bontems, 2007). In this circumstance moral suasion should be accompanied by cost sharing arrangements and incentives.

2.4.7 Tradable Emission permits

Tradable emission permits involve establishing marketable property rights for discharging nitrogen into the environment. They define the amount of nitrogen that a farm is allowed to discharge and the sum of all permits is equal to the total level of emission permissible within the region of interest. Farms that exceed their permit limits can be subject to non-compliance fines and so have an incentive to purchase permits from another farm or firm. Conversely, a farm that finds abatement less costly might find selling its permits and reducing its emissions more profitable than keeping its permits and polluting. Competitive trading should lead to a cost effective allocation of pollution abatement among farms.

Trading provides an incentive to invest in developing abatement oriented BMPs because farmers can anticipate revenue through the sale of pollution reduction credits (Rousseau, 2001). Mean while trading render an opportunity to those, who can not reduce the level of pollution at a lower cost. The attractiveness of tradable emission systems is that they shift decisions about the design and location of pollution control from the regulator to the farm. In contrast deciding a tax rate on emissions or inputs requires full information on polluters' response, which is not likely to be known even by the polluters themselves in the short run. Another appealing feature of tradable emission systems is that they allow for adjustment in the initial allocation and subsequent control of supply.

In New Zealand the possibilities of introducing tradable emission permits in the catchments of Lake Taupo and Rotorua has been explored (Kerr, Lauder, & Fairman, 2007). However high transaction costs in identifying and exchanging the credits between farms makes the system less appealing. However use of existing trading platforms for general goods, such as "Trademe" can be explored. Further the method of initial permit allocation is contentious. Therefore methods that minimise transaction costs need to be developed such as use of a suite of web-based tools like "Nutrient net" to facilitate market-based approaches to improving water quality (Greenhalgh & Selman, 2006).

2.5 Appropriate application of policy

Since a single economic instrument is unlikely to be strictly preferred over all policy options over all conditions, the optimal strategy for any given situation will probably involve a mix of instruments (OECD, 2007; Weersink, Livernois, Shogren, & Shortle, 1998). Economic instruments could be used in conjunction with other environmental policy choices, moral suasion and direct regulation.

Direct regulation can be the most practical way to deal with certain kinds of environmental issues like prevention of livestock access to water bodies, which could be monitored. The acceptability of regulation can be enhanced through financial assistance programmes. Combined tax and subsidy schemes have been used elsewhere to improve water quality (Qiu & Prato, 1999). In these schemes, farms which adopt abatement measures are given tax rebates.

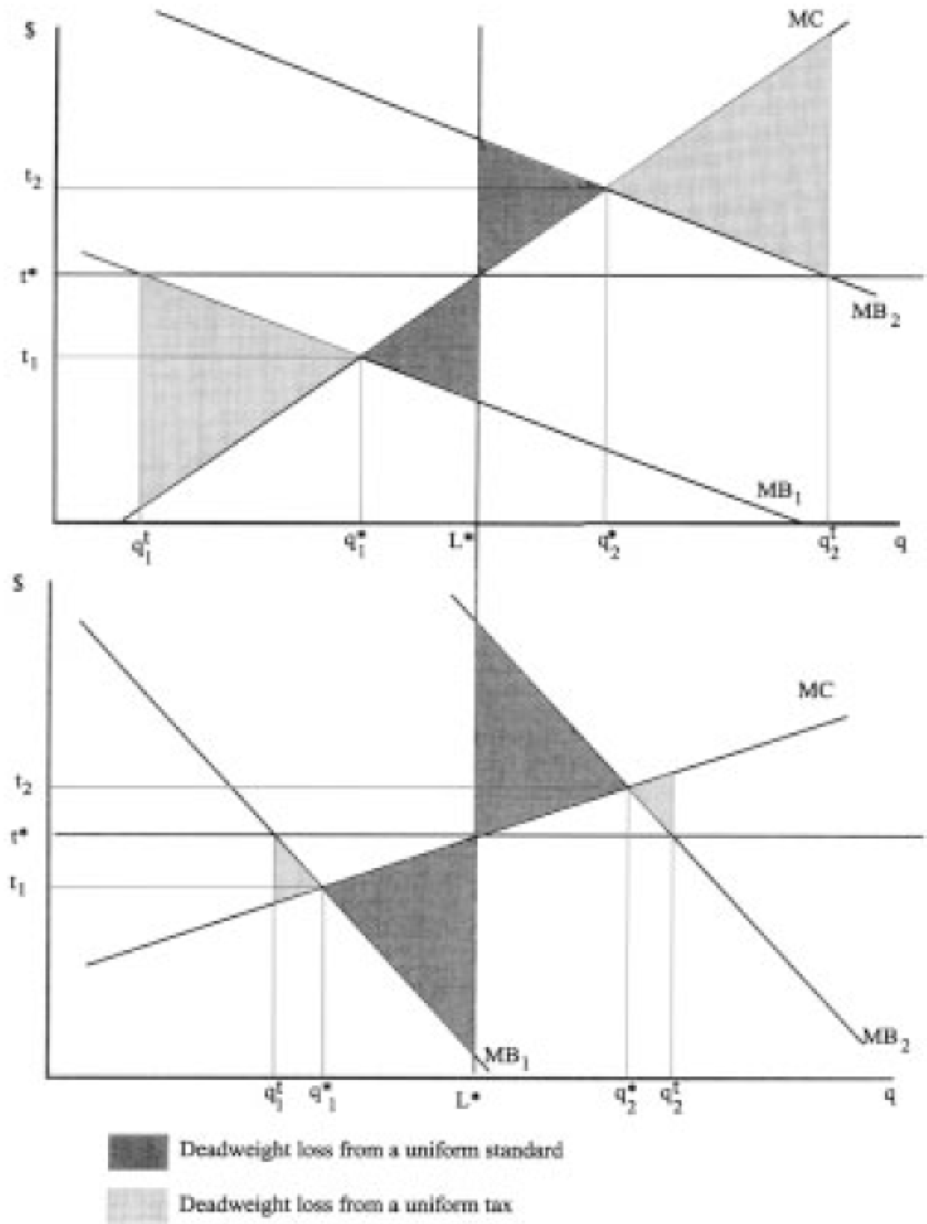
Mandatory approaches like emission or input use taxes or restrictions have drawbacks in terms of inflexibility or high transaction costs. While voluntary approaches designed to entice farmers to adopt BMPs via incentives and moral suasion fail to provide adequate environmental protection. Segerson & Wu, (2006) examined combining voluntary approaches with a background threat of imposing a retroactive ambient tax, when nutrient reduction targets are met voluntarily. This approach induces cost minimizing abatement without the need for farm specific information about pollution related features. This policy can therefore be both more effective than a pure voluntary approach without a threat and involves lower information and transaction costs than a pure ambient tax.

2.5.1 Impact of heterogeneity

The heterogeneity of abatement costs is fundamental in the choice of optimal mitigation policy instruments. Three major sources of abatement cost heterogeneity can be distinguished based on level of activity, emission factors and flexibility of substitution between production activities (De Cara, Houze, & Jayet, 2005).
In the presence of farm heterogeneity, the policy instrument and level of instrument have to be differentiated to reach first-best solution as the relative slopes of the marginal costs and benefit curves vary across farms (Wu, 2000).

Wu & Babcock (2001) illustrated the factors, which could affect choice of policy instrument. They found that taxes are efficient when marginal profits are more sensitive to fertiliser use than marginal pollution cost and marginal profits are higher than the tax rate. The upper panel of Figure 2.6 shows that when the marginal cost function is steeper than the marginal profit functions, a uniform tax results in a larger dead weight loss than a uniform standard. The lower panel shows the opposite. The negative correlation between marginal profits and marginal costs also favours a tax. The intuition behind this result is that under a tax, producers will apply more inputs to land with a high marginal profit, when marginal profits and marginal costs are positively correlated, land with a higher marginal profit tends to have a high marginal pollution costs. On the other hand, when marginal profits and pollution costs are negatively correlated, land with a higher marginal profit tends to have lower marginal pollution costs. Therefore a tax, which allows more input use on more productive land, tends to be a better instrument. Low application efficiency of inputs leads to a high marginal pollution cost, because low efficiency implies that much of the applied fertiliser is lost to the environment. Marginal cost curves are steep because the environment is likely to receive a larger proportion of the pollution as more is applied. Further low input application efficiency causes low and relatively flat marginal profits. A combination of flat marginal profit and steep marginal costs favours standards. When the marginal cost function is steeper than the marginal profit function, a uniform tax results a larger dead weight loss than a uniform standard. When the marginal cost function is flatter than the marginal profit function, the outcome tends to favour a tax. These authors showed that the positive correlation between marginal profits and marginal costs favours standards. Meanwhile a negative correlation favours the tax. Therefore geophysical and production factors need to be considered in choosing appropriate policies.

Figure 2.6 The effect of relative slopes on deadweight losses



Source: Wu & Babcock (2001)

2.6 Role of technology in policy implementation

Well designed environmental policies can build on valuable synergies with technological development. Environmental policies also provide incentives for innovation in management practices. While improving management practices can contribute to lower abatement costs thus increasing the feasibility of environmental policy implementation.

2.6.1 Proxies for nitrogen discharges

The diffuse nature of nitrogen discharges makes the design of effective environmental policy for agriculture difficult. Many farms contribute to pollution making it difficult to separate damages caused by individual farms and assign liability. The costs of monitoring agricultural pollution are high because of the complexity of production and environmental processes, which make it difficult to directly infer emissions from observable inputs (Braden & Segerson, 1993). The difficulty in measuring actual farm nitrogen discharges within a reasonable time and cost has made the application of economic instruments to such issues particularly challenging. One approach is to use input levels as proxies for nitrogen discharges. However nitrogen discharges are inherently stochastic because of geophysical and weather variables. Therefore observation of inputs cannot provide a reliable forecast of nitrogen discharges. The diffuse nature of agricultural nitrogen pollution and the time lag before it appears in the water body, necessitates the use of a simulation model. Developments in information technology enhance the ability to estimate nitrogen discharges under different climatic and geographic conditions using different software applications.

Use of estimated emissions is appealing, because emissions from individual farms cannot be monitored with reasonable timeliness, accuracy and cost (Wu & Babcock, 1999). Horan, Shortle, Abler, & Ribaudo (2001) cited many studies that showed nitrogen emission proxies were more effective than standards or taxes applied to nitrogen inputs. They stated that trading programmes for which nonpoint permits are defined in terms of estimated emissions are more effective than those based on input use. This is because emissions are a better indicator of environmental pressure than inputs. Reliable proxies enhance the feasibility of certain types of economic instruments, like emission charges and lower monitoring cost. In New Zealand, Overseer, a farm scale nutrient budget simulation model (Wheeler, Ledgard, DeKlein, Monaghan, & Carey, 2003) has the potential to contribute to emission based environmental policy initiatives (Dake, 2007; Horizons Regional Council, 2007). Information derived from nutrient budgeting can readily be incorporated into Overseer to estimate the nitrogen discharge of a farm. In principle such estimates could be made for the forty percent of Waikato dairy farms that use nutrient budgeting (Ministry for the Environment, 2007). However current simulation models cannot provide sufficiently accurate estimates of the complex fate and transport of most agricultural pollutants to water bodies, so improvements will be necessary, if this indirect approach is to withstand legal challenge and gain political legitimacy.

One famous application of nutrient budgeting is the Dutch Mineral Accounting System (*MINAS*). This policy requires a detailed accounting system for farm nutrients for farms having more than 2.5 livestock units per hectare. The system requires recording and reporting of all inputs used and outputs produced in the farm. Sales of output and input purchases are allowed only from accredited firms. Nutrient surpluses are differentiated according to soil type and land use. The OECD report on Instrument Mixes for Environmental Policy (2007) states that this system was effective for decreasing nutrient surpluses in dairy farming due to the availability of alternative methods of reducing nutrient surpluses. However high administration costs, lengthy judicial procedures due to fraud attempts, frequent changes introduced without much planning and failure to provide viable alternatives to handle surplus nutrients in intensive farming systems like poultry and swine made the whole system less attractive.

Geographic Information Systems (GIS) and satellite imagery have the potential to reduce the cost of monitoring and improve the accuracy of physical simulation models by providing accurate information, which could potentially provide a legal basis for prediction of nitrogen discharge levels for any given farm from observable practices or farm records.

2.6.2 Best management practices (BMP)

Concern with environmental pollution has led to technological developments aimed both at remedying and preventing their future occurrence. Technological breakthroughs potentially reduce the cost of abatement by reducing the form and level of discharge into the environment from farming systems (Bontems, 2007).

Management strategies for reducing nitrogen discharge can be categorised into two major classes, namely source reduction and interception strategies (Ribaudo, Heimlich, Claassen, & Peters, 2001). A range of BMPs have been researched in New Zealand pastoral farming. A detailed description of these strategies can be found in Ledgard & Menneer (2005). The source reduction strategies adopted to manage nitrogen discharges are reduction of fertiliser application and stocking rate, nitrification inhibitors and restricted grazing during winter (Grazing off). Major interception technologies are fencing of riparian margins, creation of wetlands and conservation reserves and feed pads and winter/standoff pads.

Leached nitrogen from urine and dung patches and applied fertilizer nitrogen are reported to be the major potential sources of nitrogen from cattle grazing systems (Ledgard & Menneer, 2005). Nitrate discharge from the plant root zone is seasonal. Monaghan *et al* (2007) found that 60% of total nitrogen discharge occurs during the winter due to low plant nutrient uptake and as well as high drainage. Best management practices like Wintering pads are useful to reduce these seasonal nitrogen discharges. Nitrification inhibitors and feed pads are reported to be a promising mitigation strategy for nitrogen (Wilcock et al., 2006). A recent research report on the impact of management changes on farm profitability and environmental outcomes, also identifies nitrification inhibitors, standoff pads and optimised nutrient use as having positive financial impacts on farms (Water Programme of Action, 2007). Keeping dairy animals on wintering pads during late autumn and winter reduces nitrogen leaching by 50-60% (Chadwick, Ledgard, & Brown, 2002). De Klein & Monaghan (2005) reported that wintering pad systems reduced nitrogen leaching by 14-44% with the largest reductions achieved in South Island catchments. Nitrification inhibitors prevent the accumulation of nitrogen in a movable form and are capable of reducing nitrogen leaching by 60% (Di & Cameron, 2002). In another study, Wilcock et al.(2006) reported that combining winter pads with nitrification inhibitors has the potential to reduce nitrogen losses by 51%. However the effectiveness of nitrification inhibitors is strongly affected by environmental conditions especially temperature. Further the effectiveness is based on small scale experimental trials and the challenge for the future is to quantify the effects across the whole range of soil and climatic conditions existing in New Zealand (Edmeades, 2004).

Designing policies to encourage adoption of environmentally friendly farming practices requires analysis of adoption decisions. Cooper & Keim (1998) used survey data to estimate payment levels needed to induce farmers to adopt alternative BMPs.

2.7 Conclusion

There is an extensive range of environmental policy instruments including emissions charges, tradable emissions permits, and tenders, subsidies for emissions reductions, performance standards and moral suasion. In theory a policy instrument that is effective in internalizing externalities, should resemble the social optimum as closely as possible. If a policy is the first best solution then it should replicate the social optimum exactly. To be the first best, the policy should be differentiated at farm or parcel level, because the marginal cost of and benefits of farming and the effectiveness of policy differ by geophysical variables as well as by production characteristics. However transaction costs involved in information gathering and monitoring favour less differentiated policies. So in practice no policies discussed here are first-best solutions. However, most empirical studies show that targeted, information intensive policies for nonpoint pollution control outperform undifferentiated uniform policies (Carpentier, Bosch, & Batie, 1998; Fleming & Adams, 1997). Berntsen, Petersen, Jacobsen, Olesen, & Hutchings (2003) used farm model to evaluate the environmental and economic consequences of different taxes on farm nitrogen use on Danish farming systems. They concluded that efficient taxation schemes should differentiate between types of farms. The choice between uniform and differentiated policies depends on the relative slopes of marginal control costs and marginal benefit functions. So that appropriate policy instruments must be developed to address water quality issues at the local level. Optimal environmental policy will minimises nitrogen discharge at low abatement cost to producers and will have low administrative and monitoring costs for enforcement.

The conceptual framework for environmental policy and subsequent discussions shed light on the importance of abatement and damage costs and substituting damage costs with exogenous discharge limits. The cost effectiveness of polices depends on the level of abatement required (Iho, 2005). The implications of specified limits on the level of pollution and farm income need to be evaluated in an ex ante manner. Shortle & Horan (2001) reported that effective design of policies requires farm specific knowledge of profit and environment.

The appropriate choice of a policy depends not only on its cost efficiency, but also on the implementation issues of particular abatement measures. These are often linked to transaction costs and political acceptability. Goulder & Parry (2008) reviewed the literature on instrument choice and reached a number of general conclusions. They found that no single instrument clearly outperform others in all dimensions; there are significant trade-offs between degree of fairness in the distribution of impacts (and political feasibility) and cost effectiveness. Hybrid instruments which combine the best features of different instruments are attractive. For instance Fischer and Newell in their paper on climate policy (2008) found that achieving a given emissions reduction through one instrument alone involved considerably higher costs than employing two instruments. Overall, Goulder and Parry suggest that identification of the best policy instrument involves art as well as science.

Legg (2006) stated that “The analysis of the cause effect linkages between policy measures and environmental outcomes is complex and too little is yet known to make strong recommendations on appropriate mixes of policy measures and market actions or to make definitive judgments about good policy practice”. Developing a policy that combines the strengths of regulatory and economic instruments requires information on the economic consequences of cause and effect relationship in agri-environmental systems. Therefore empirical estimation of the impact of alternative environmental economic policies is important for effective policy development. Bio-economic modelling is capable of tackling a component of this sort of information problem (Bennett, 2005). An economic model based on mathematical programming, drawing on estimates of behavioral parameters from econometric studies, simulation models and scientific experiment could provide valuable insights into the information required. In particular more location specific empirical research is needed, using an integrated modelling approach at the catchment scale that captures environmental and economic impacts. The research has led to the development of useful tools model the policy implications on farm. The reminder of this thesis largely deals with data generation, bio-economic modeling and its potential uses for policy analysis. However the choice of appropriate policy requires estimates of the marginal benefits of policy as well as the marginal costs of implementation. Non-market valuation is typically needed to measure benefits of a proposed policies to restrict nitrogen loadings at varying levels in a catchment (MacDonald, Connor, & Morrison, 2004). Hence, bio-economic modeling needs to be complemented with farm surveys and non market valuation studies.

3. Characterisation of the catchment and nitrogen pollution of water: A platform for environmental policy analysis

3.1 Spatial dimension

Agri-environmental policy analysis exploring the interaction between agriculture and water quality has included studies at a range of spatial scales including national, regional and catchment. In general, a catchment has been considered an appropriate spatial unit for modeling policies to manage diffuse source pollution (Just & Antle, 1990; Kampas & White, 2003; Novonty, 1999). Schou, Skop, & Jensen (2000) justified the appropriateness of using the catchment scale on the basis of its definite boundaries and negligible pollutant flow between the catchment and other areas. Its importance is evidenced by a growing literature on spatially referenced environmental policy at a catchment scale (Berntsen, Petersen, Jacobsen, Olesen, & Hutchings, 2003), and many environmental agencies confine their environmental policies to catchments (Johansson, Gowda, Mulla, & Dalzell, 2004). Improved spatial information may yield large benefits to society because nonpoint pollution is heterogeneous and diffuse (Carpentier, Bosch, & Batie, 1998), and this itself has an effect on the cost effectiveness of environmental policies (Martinez & Albiac, 2006; Wossink, Lansink, & Struik, 2001). Therefore considering spatial heterogeneity at a catchment scale is important in modelling nitrogen abatement policies.

Spatial heterogeneity can be attributed to geophysical and production variability. Geophysical variables such as proximity to water bodies, soil type, topography and distribution of water margins, and production variables such as input use intensity and farm size, influence nitrogen discharge into water. Shortle & Horan (2001) showed that spatially differentiated policies are efficient when there is a considerable variation in pollution contribution from farms due to physical and management differences. It is important to establish the variations and levels of nitrogen pollution within a catchment, and in the presence of heterogeneity, targeting particular farms or areas may pave the way for tailor-made abatement policies.

Geographic information systems (GIS) have been widely used (Mapp, Bernado, Sabbagh, Geleta, & Watkins, 1994; Munier, Birr-Pedersen, & Schou, 2004; Opaluch & Segerson, 1991; Schou, Skop, & Jensen, 2000; Yang & Weersink, 2004) to represent the spatial dimension in agri- environmental policy analysis. GIS provides the opportunity for greater realism, comprehensiveness and relevance in agri-environmental policy modelling (Bateman, Ennew, Lovett, & Rayner, 1999). In this study the spatial analytic capability of GIS is used to define the catchment boundaries and its characteristics.

3.2 Application of Geographic Information System

This section provides an overview of GIS, which is a computerized information management system designed to capture, store, integrate, analyse and display data from a geographic perspective. GIS uses geo referenced data, aligning geographic location data to a known coordinate system by latitude and longitude. It enables data to be combined from different sources in a consistent manner to derive useful information by viewing, querying and analyzing. Richer data sets are able to be developed for site specific characteristics.

There are a number of different data formats that can be imported into a GIS. The most common form is vector data, which includes point, line, and polygon data. In this study farm boundaries are categorised as polygon[1] data, flowing rivers are line data and water quality monitoring locations are point data. Vector data have attributes associated with them, and these can provide information about ownership, polygon names, collection dates, sources, special codes, soil type and land use. The attributes are stored in a database file and are linked to the vector information. In a GIS, users can access the attributes to perform analyses and query the vector data.

---

[1] In a vector-based geographic representation a polygon is a continuous two-dimensional object, which may be homogeneous or divided internally into areas with different characteristics. Each polygon is encoded in the database as a sequence of locations that define the boundaries of each closed area in a specified coordinate system.

The catchment has been delineated by using the digital database of river flows known as the River Environmental Classification (REC). Catchment characteristics are generated by combining layers of different data sources in GIS. Using these facilities, joint distributions for attributes such as land use, soil type and topography are created. The details of geo-spatial analysis and data sources are discussed at a greater detail in the subsequent chapter on spatial micro-simulation.

3.2.1 Application software

A software application known as ArcGIS is used for analysis. ArcGIS Desktop identifies patterns, relationships, and trends in the data that are not readily apparent in databases, spreadsheets, or statistical packages. It was developed by the Environmental Systems Research Institute (ESRI) in the USA.

ArcGIS Desktop is a suite of integrated applications including ArcMap, ArcCatalog, and ArcToolbox (ESRI, 2007). Analysis of data was carried out using these tools in an integrated manner. Analysis consists of mapping, area calculations, data editing and compilation, data management, visualization, and geo-processing. A typical geo-processing operation involves manipulating input data through operations like geographic feature overlay, feature selection and analysis.

3.3 The Catchment

Nutrient discharge into water bodies degrades water quality. Agricultural land use has contributed to the increased levels of nitrogen in Waikato water bodies. The contribution of pastoral agriculture to the water quality degradation has been well recognized (Parliamentary Commissioner for the Environment, 2004). Nitrogen is a significant contaminant in the Waikato River. In recent years nitrogen concentrations in the Waikato river tributaries have increased at an average of 2.5 percent per annum (Vant & Smith, 2004).

The catchment delineated is situated within the boundaries of the broader catchment of the Waikato River, which has been identified as one of the water bodies in the region with a high priority for nutrient management (Brodnax, 2006; Environment

Waikato, 2005c). The approximate geographic area of the whole catchment is 436,000 hectares and includes all land that drains into the Waikato River from the outflow of Lake Taupo to the Karapiro dam (Environment Waikato, 2007a) .

As a rational compromise between sufficient variability and computational convenience, the catchment examined in this study covers only part of the broader catchment. It includes the middle part of the Waikato River catchment from the Karapiro dam to Lake Arapuni, plus contributing tributaries. The catchment is bisected by the main stem of the Waikato River (Figure 3.1). It comprises approximately 151,678 hectares, with an annual average precipitation of 1200-1600 mm/year. It has considerable spatial variability in terms of physiographic parameters such as topography and soil type. The geopolitical boundaries of the catchment fall within the local authorities of Waipa, Otorohanga and South Waikato, with the South Waikato covering the largest (78%) portion of the catchment land area. Integrated aerial and satellite imagery with a transparent topographical map overlay shows the land use, townships and road network within the catchment (Figures 3.2 and 3.3).

Figure 3.1 Location of the catchment

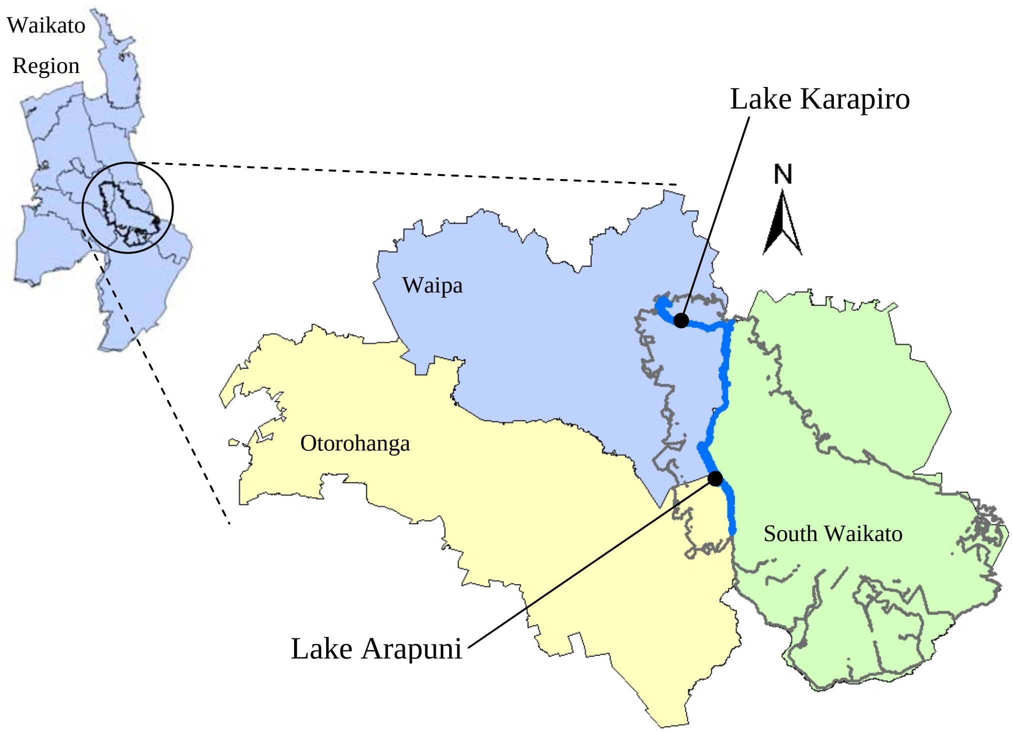


Figure 3.2 Topographic map of the catchment

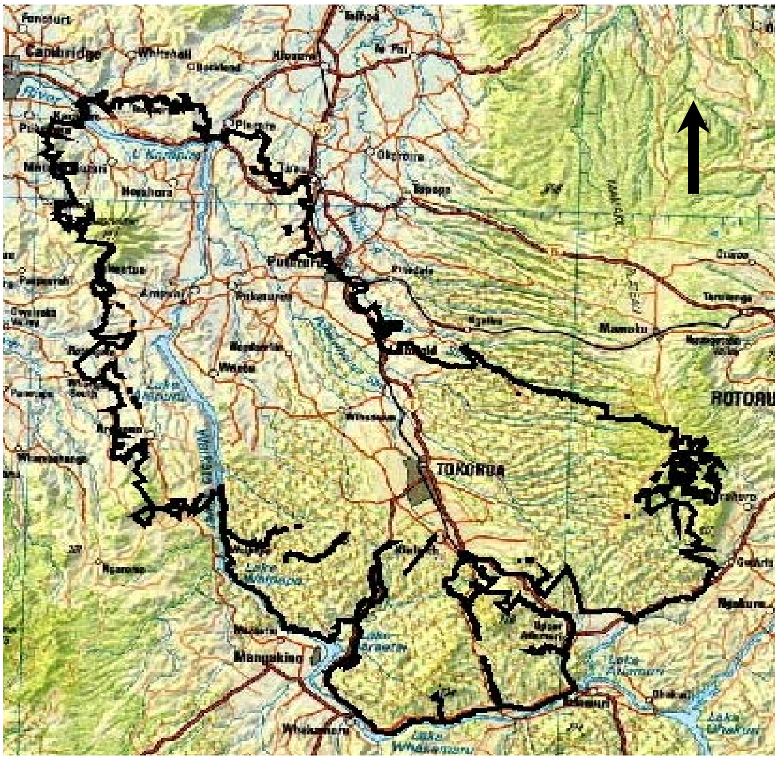
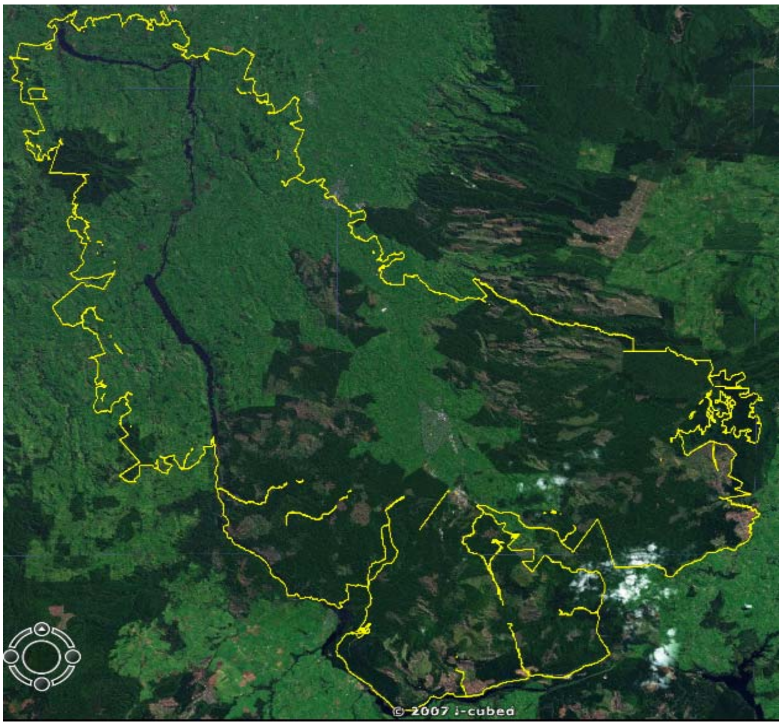


Figure 3.3 Satellite imagery of the catchment



The rate and concentration of water and contaminants entering the water body is influenced by catchment specific geology, rainfall, land use and topography. The following sections briefly describe the land use and nitrogen pollution in the catchment.

3.3.1 Land use

Land use in the catchment is predominantly pastoral, with dairying as the major pastoral farming activity. Dairy farming in New Zealand is an intensive form of land use, often involving high stocking rates and fertiliser application rates which generate elevated concentrations of nitrogen in water (Ledgard, Penno, & Sporsen, 1999). Dairying is considered to contribute considerably to the problem of nitrogen discharge to water bodies (Ledgard, De Klein, Crush, & Thorrold, 2000). Nitrogen discharge from average Waikato dairy farm is 36 kgN per ha. It is about three times higher than that of an average sheep and beef farm, which is 13 kgN per ha (Ledgard & Power, 2006). Meanwhile nitrogen losses from undisturbed plantation forestry is around 2 kgN per ha (Davis, 2005). This study, therefore, focuses primarily on dairy farming and its nitrogen discharges. Besides dairying there are considerable pine forests in the catchment, and these have the potential to be converted to pastoral farming (Environment Waikato 2007b).

Environment Waikato is concerned about any increase in the nutrient status of the Waikato River and tributaries from existing and anticipated land use changes in the catchment between Karapiro Dam and Taupo gates. The catchment is a good area for simulating the impact of current as well as future land use changes.

The smallest area unit for which time series data is available is the territorial local authority. Stocking rate and average milksolids per cow have increased in all three territorial local authorities (Table 3.1 and Figures 3.4 and 3.5), providing an indication of the intensification of land use over this time.

Table 3.1 Production statistics at territorial local authority level

	Total herds	Average herd size per farm	Average effective hectare per farm	Average cows per ha	Average milksolids per ha	Average milksolids per cow
Waipa						
2000-2001	755	239	82	2.93	911	314
2001-2002	727	257	89	2.92	904	308
2002-2003	686	271	96	2.88	935	323
2003-2004	667	284	94	3.07	991	323
2004-2005	635	295	97	3.08	956	309
2005-2006	609	302	99	3.08	1013	329
2006-2007	596	313	102	3.1	1041	335
Otorohanga						
2000-2001	464	258	90	2.92	897	313
2001-2002	451	280	99	2.85	857	300
2002-2003	434	285	104	2.78	862	309
2003-2004	420	303	102	2.96	933	315
2004-2005	414	309	105	2.95	909	308
2005-2006	404	313	107	2.95	940	317
2006-2007	391	318	110	2.95	975	329
South Waikato						
2000-2001	438	251	87	2.94	909	313
2001-2002	429	269	95	2.89	892	308
2002-2003	418	276	101	2.79	920	330
2003-2004	404	290	100	2.95	983	333
2004-2005	394	297	103	2.95	951	323
2005-2006	384	303	104	2.95	1003	338
2006-2007	377	323	110	2.99	1036	346

Data source: New Zealand Dairy Statistics (2000/01-2006/07)

Figure 3.4 Average stocking rate in TLA

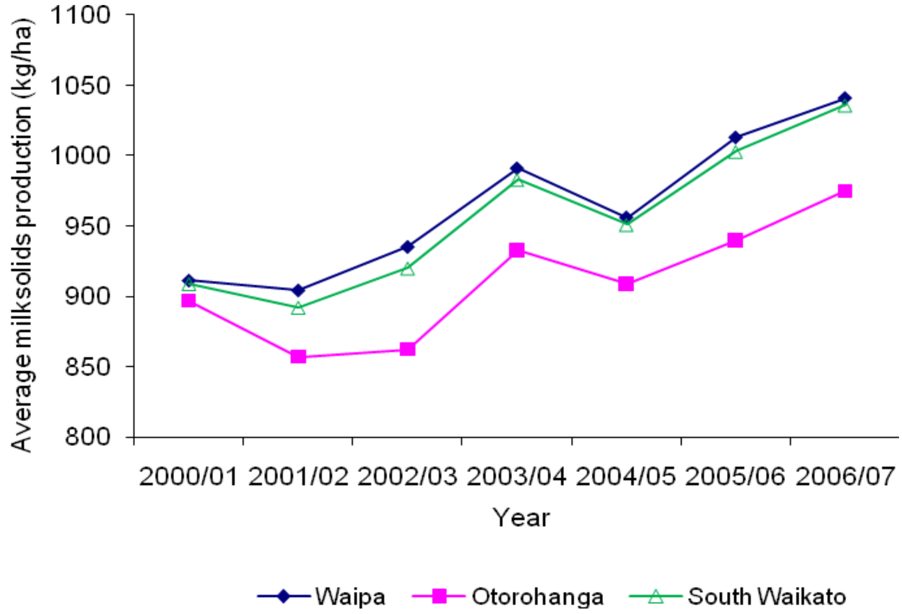
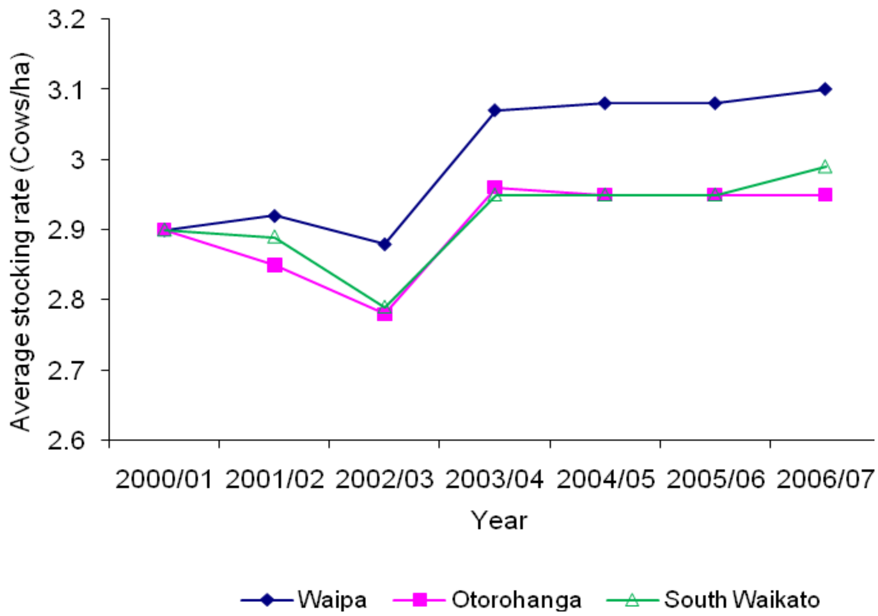


Figure 3.5 Milksolids production per hectare



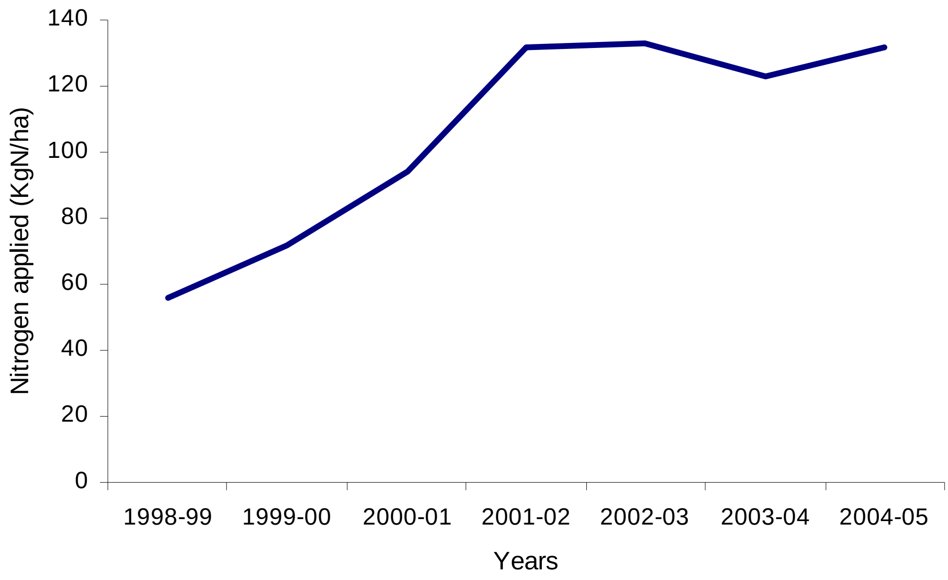
Vant (1999) studied the relationship between nitrogen losses and stocking rate in Waikato farms and found a strong correlation. He expressed the relationship as follows.

Nitrogen load (kg/ha/y) = 10.28 x dairy cow stocking rate (cows/ha) + 2.241

The correlation between stocking rate and nitrogen yield suggests that the intensity of livestock farming in a catchment is a key factor in determining nitrogen discharge into surface waters. In addition average farm size has increased over time. This can be attributed to the merger of farms, and a reduction in the number of herds in the territorial local authorities is consistent with this interpretation. An increase in the total extent of dairy farming due to land use conversions, combined with the increase in the intensification of farming can contribute to increased nitrogen contamination.

Technological advances may lead to the increased intensification of dairy farming. Some of the increase in milksolids production per cow can be attributed to improvements in the genetic merit of herds and pasture. However intensification of input use has been a key driver of productivity improvements in the short run. According to a Waikato based farmlet trial, increases in nitrogen fertilizer from 200N per ha to 400N per ha raised milksolids production per cow by 6 percent (McGrath, 1998). The Economic Survey of New Zealand Dairy Farmers shows that nitrogen use per ha in dairy farming increased more than two fold from 56 kgN per ha to 134 kgN per ha in Waikato from 1998 to 2005 (Figure 3.6). This increase in nitrogen fertilizer use over the years can be attributed to relatively low prices of nitrogen fertiliser. In recent years Nitrogen use became more affordable as a result of a rise in the ratio of milk price to nitrogen cost (Thomas, Ledgard, & Francis, 2005).

Figure 3.6 Average nitrogen use on dairy farms



Data source: Dexcel, Economic Survey of New Zealand Dairy Farmers (1998/99-2004/05).

3.4 Nitrogen pollution in the catchment

3.4.1 Level of nitrogen pollution

Rising nitrogen levels in water have been a growing concern because of their potential and actual effect on public health and the environment. High levels of nitrate can pose a health risk, and this is reflected in the Drinking Water Standards for New Zealand which are set at a Maximum Acceptable Level (MAV) of 0.13 grams per cubic meter for nitrate-nitrogen (Ministry of Health, 2000).

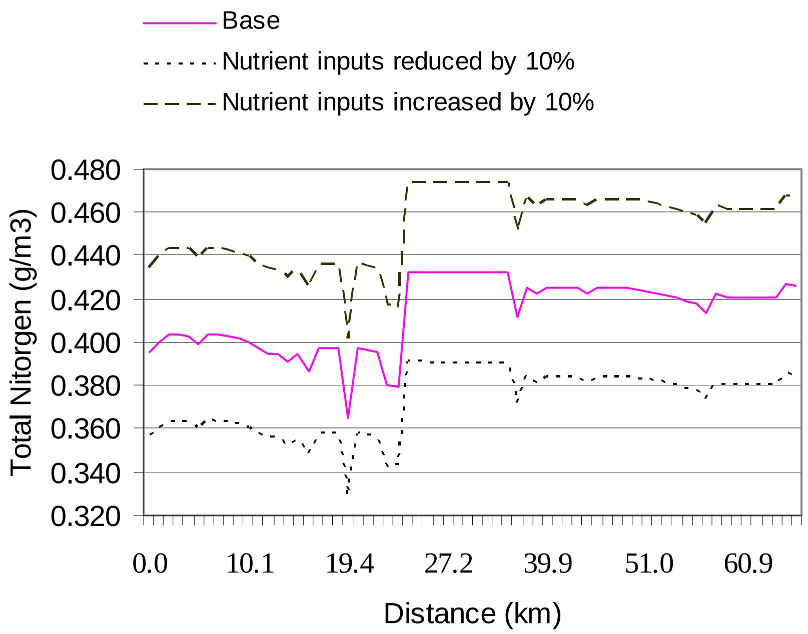
The mobile nature of nitrogen (compared to most other nutrients), leads to the potential for significant losses of it into the environment from nitrate leaching to water. Total nitrogen, defined as a summation of nitrate nitrogen and Total Kjeldhal nitrogen, has been used an indicator variable for nitrogen content in water (Atasoy, Palmquist, & Phaneuf, 2006). Ideally, total nitrogen levels in water should be less than 0.5 grams per cubic metre to prevent excessive growth of nuisance plants. Environment Waikato categorises rivers and streams with total nitrogen levels above 0.5 grams per cubic metre as undesirably nutrient-enriched, between 0.5 and 0.1 as satisfactory and less than 0.1 as excellent (Beard, 2007).

Increased levels of nitrogen in the water induce the growth of algae during hot weather. This is known as Eutrophication. It results in lower visual clarity, changes in colour of water and dwindling dissolved oxygen. Lower dissolved oxygen level causes death of fish species. These changes will have a serious impact on the recreational value of the water.

The quality of water along the Waikato River changes from a near pristine status at Lake Taupo to a pale colour when it passes the Karapiro Dam. The overall water quality in the main stem of the river can be attributed to good quality water leaving Lake Taupo as this account for the most of the water in this part of the river. Water quality deterioration in the catchment can be attributed mainly to the water quality of inflowing tributaries (Environment Waikato, 2007b). We can expect that continuing deterioration in the condition of the inflowing tributaries will eventually result in deterioration in the main stem of the river.

The impact of land use changes is not realised immediately as there can be a time lag of years between the losses of nitrogen from the land surface until the appearance of nitrogen in the surface water. Contaminants that travel primarily via groundwater tends to take longer to impact on surface water. Nitrogen is one such contaminant, and once it reaches surface water, there is no practical means of limiting its eventual impact. Simulation models are therefore used to predict nitrogen levels in water as a result of land use changes. Rutherford (2005) modelled the impact on water quality of upstream changes in input and land use along the Waikato River catchment. Figure 3.7 was produced by compiling the data points within the catchment from Rutherford's model. These data points are from Waipapa/Ngarahu Stream to Little Waipa Stream. Reduction of nutrient application reduced the total nitrogen content in water by 10 percent. Lowered nitrogen levels are likely to reduce the algal growth and have the potential to enhance the clarity of the water because of the lower chlorophyll content. Increase in nutrients is likely to reverse this trend. Upstream land conversion of 20,000 ha of pine to dairy could cause a 22% rise in the total nitrogen content in the water and a consequent rise in Cholorphyll content. Rutherford predicts that a 40,000 ha land conversion to dairy could result in a 44% increase in the total nitrogen content of the water.

Figure 3.7 Simulated nitrogen content along the main stem of the river



Data source: Rutherford (2005).

3.4.2 Water quality trends

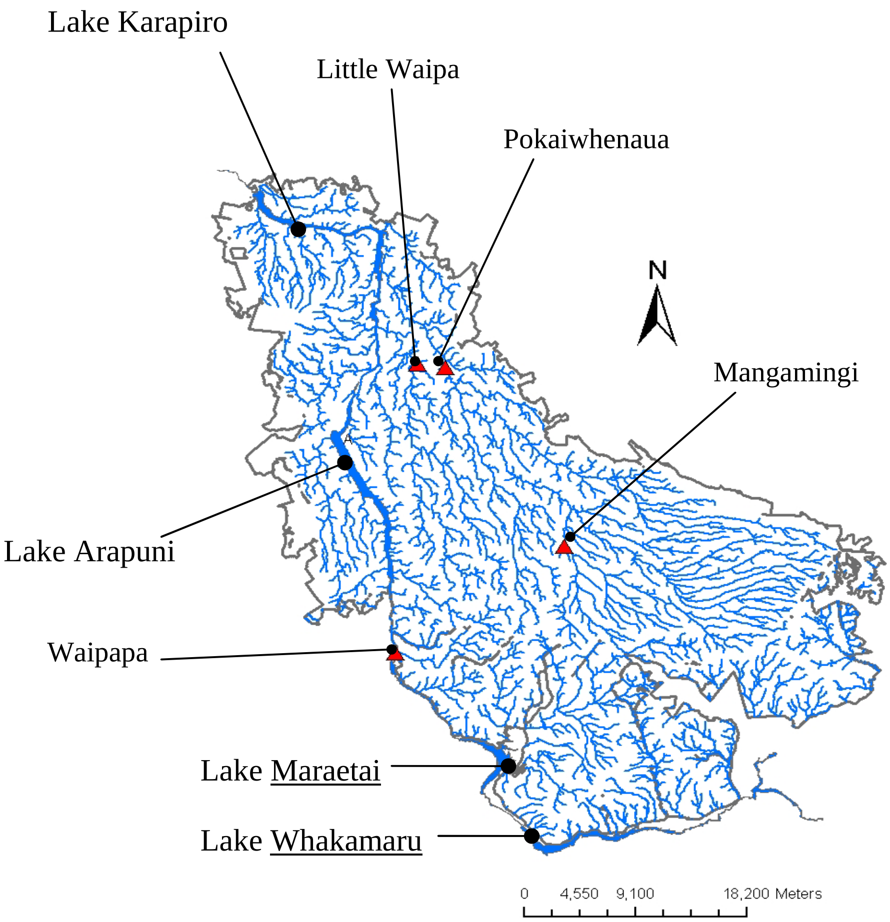
A time series of nitrogen concentrations in river water was supplied by Environment Waikato from their network of monitoring locations. Figure 3.8 shows monitoring locations which are derived by intersecting catchment boundaries on the GIS layer of Environment Waikato's Regional Water Quality Monitoring of Streams Programme[2]. The monitoring data consists of total nitrogen level at monthly intervals from 1995 to

---

[2] Environmental data location information sourced from Environment Waikato database and may be subject to Privacy regulations. Copyright Reserved.

2006. The Waipapa monitoring point is located on the main stem of the river. Other points are located on the tributaries.

Figure 3.8 Water quality monitoring locations[3]

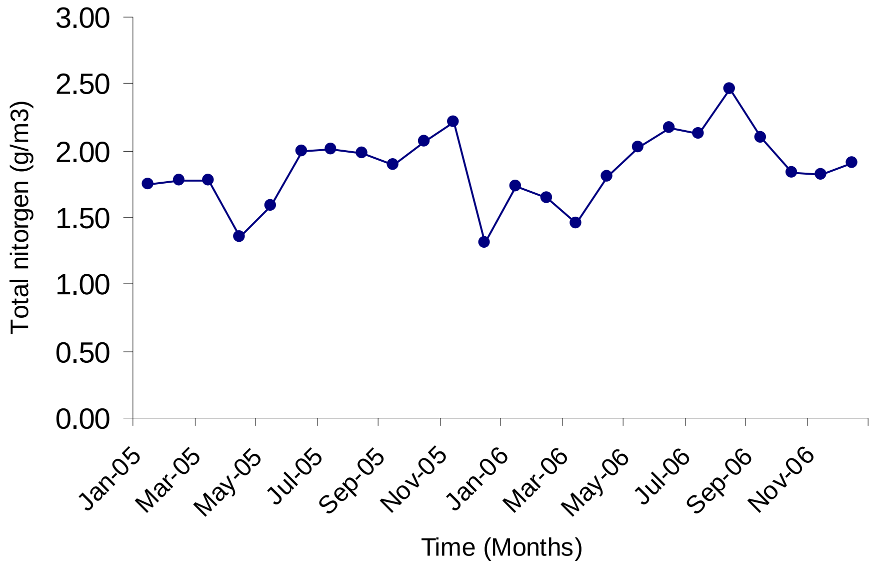


Total nitrogen concentration in water shows a cyclical pattern over time (Figure 3.9). In a year, total nitrogen level peaks during the winter and falls during the summer. This annual fluctuation in nitrogen is attributed to lower downward movement of nitrate nitrogen from farm lands in summer as Evapotranspiration exceeds precipitation. Plant biomass in the water bodies in summer is at a maximum (Ledgard, Sprosen, Brier, Nemaia, & Clark, 1996; Wilcock et al., 1999) and this

[3] The catchment includes only Lakes Karapiro and Arapuni, the contributing tributaries to the lakes, and the section of the Waikato River between these lakes.

tends to absorb the nitrogen in the water and release it during the decaying process. This trend is reportedly common for time series data with predominantly agricultural influences (Worrall & Burt, 1999).

Figure 3.9 Annual cycle of nitrogen concentration in water



Seasonal KendallTest

A Seasonal Kendall Test is prescribed to identify time trends for data with seasonal cycles, as illustrated in Figure 3.10. The Seasonal Kendall Slope Estimator (SKSE), derived from this test, is the median of all possible combinations of slopes for each of the months of the year. Results for all Januarys are compared with one another, but they are not compared with those from the other months. The SKSE estimator does not require any distributional assumptions. The detailed formulas for the Seasonal Kendall test are described in Gilbert (1987). These formulas are estimated using the Excel macro developed by Vant & Smith (2004) for empirical estimation. The SKSE is estimated using raw data instead of using flow adjusted data, which is not readily available for these monitoring locations. However nitrogen discharges from non-point sources are not significantly affected by differences in flow rate (Rutherford, 2005). Further, the U.S. Geological Survey's water resources study shows that in general nitrogen concentrations in surface water are not related to stream flow (Schnoebelen, Becher, Bobier, & Wilton, 1999).

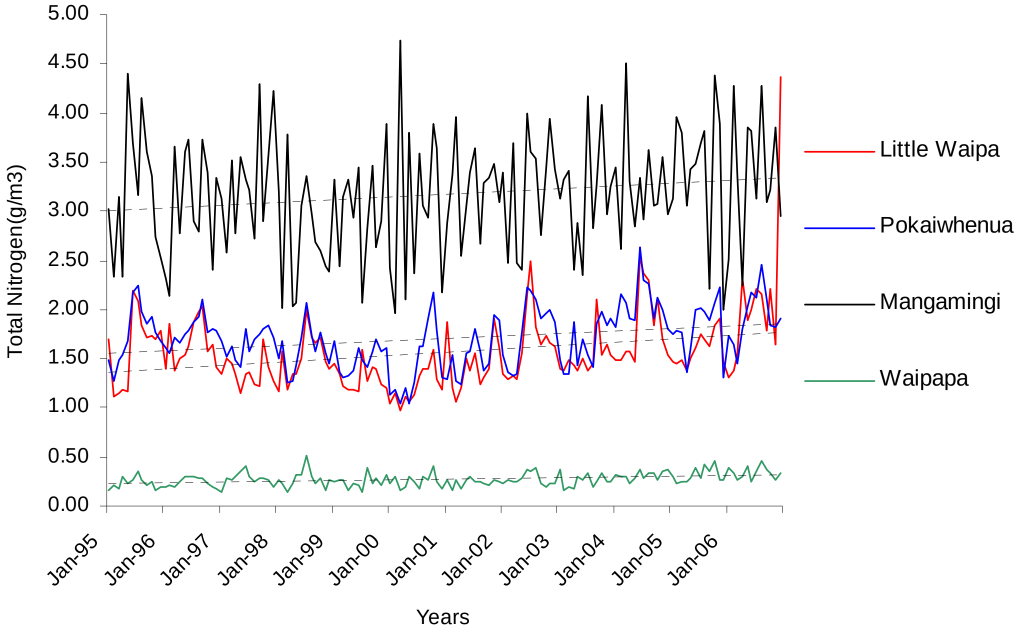
Positive SKSE slopes result from an overall increase in the values of a water quality variable, while negative slopes result from an overall decrease. Slopes are conventionally expressed in “water quality units/time”. For example, analysis of a record of concentrations in g/m3 gives a slope in units of (g/m3)/year. The trend test calculates the probability of getting a trend slope. Smaller P values indicate a significant trend. Higher SKSE values indicate less change in land development over time. The P-value is calculated by comparing the total number of increasing monthly slopes with the total number of decreasing slopes. If the net result is close to zero, the p-value will be large, so the slope can be regarded as being due to chance. Conversely, a large difference between the numbers of increasing and decreasing slopes produces a low *p*-value, meaning the slope is unlikely to be due to chance. Table 3.2 shows the trend (SKSE Slope) and respective p-values for the data for the period 1995 to 2006.

Table 3.2 SKSE test results

Location	SKSE Slope	P-Value
Little Waipa	0.023	0.280
Pokaiwhenaua	0.023	0.094
Mangamingi	0.006	2.282
Waipapa	0.008	0.000

The SKSE results indicate that the trend of total nitrogen concentration is positive. The P-value is not significant at location Mangamingi. It can be attributed to the dramatic improvements in the quality of waste water from nearby Kinleith pulp and paper mill over the last decade (Kim & Smith, 2006). However Mangamingi has the highest nitrogen loadings on average. Perhaps improvements in the water quality Total nitrogen tends to increase at a greater rate in streams in more developed parts of the catchment, reflecting increased leaching losses from areas of pastoral farming following intensification. For instance the increase in nitrogen content at Waipapa is significant, as indicated by lower and positive SKSE value and significant P-value. The time series of the total nitrogen content is presented in Figure 3.10.

Figure 3.10 Time series of total nitrogen



This graph shows the spatial heterogeneity of nitrogen loadings in water. As discussed before in winter nitrogen levels tend to peak. The Waipapa monitoring location shows the lowest total nitrogen concentration. This indicates relatively good water quality in the main stem of the river. The Mangamingi location shows relatively high levels of total nitrogen content, which may be attributed to its pumice soil. In all monitoring locations except Waipapa the total nitrogen level exceeds the maximum acceptable level for drinking water (0.13 g per cubic meter) and threshold level for triggering algal bloom. Therefore Little Waipa, Mangamingi and Pokaiwhenaua can be described as hot spots within the catchment. Given that water quality monitoring point Waipapa located in the main stem of the river, which largely receive better quality water from Lake Taupo and relatively smaller size of the catchment, entire catchment should be treated as a hot spot.

Nonpoint source pollution is complex and affected by antecedent soil conditions, timing of fertilizer application, land cover, location, duration and intensity of precipitation (Becher, Kalkhoff, Schnoebelen, Barnes, & Miller, 2001). Finding hydrological reasons for these spatial discrepancies in nitrogen loading is beyond the scope of this thesis.

If the trend shown by the nitrogen time series can be ascribed to changes in land use, this would suggest that the catchment has experienced considerable land use intensification throughout this period. Land use intensification over time, as confirmed by local authority indicators, may be linked to the rising nitrogen content of the water.

3.5 Conclusion

The Waikato River and its hydro lakes and tributaries have been experiencing nitrogen contamination problems from agriculture for some time. Pastoral agriculture, particularly dairying, is the major land use in this area. There has also been a trend towards increased nitrogen concentrations in the water body over time. This is an indication that the catchment is prone to nitrogen contamination, and this may be linked to the intensification of dairying. Nitrogen in the water is likely to increase even faster with pine to pasture conversions. It is clear that actions to mitigate the problem are necessary if the water quality is to be maintained or improved. The catchment considered is a good platform to study, design, develop, and apply an analytical framework to evaluate the potential economic and environmental effects of policies. The catchment land use and geophysical diversity are ideal for creating representative hypothetical farms to examine agri-environmental policies. The lessons learned from this policy analysis platform can be extended to wider geographic scales as well.

4. Micro-simulation – a novel approach to using farm survey data for catchment scale modeling and policy analysis

4.1 Introduction

The slow evolution of environmental policy has been attributed to lack of data and the complexity of ecological and economic systems (Adamowicz, 2007). Policy analysis is undertaken at all scales from local to international, but the catchment has generally been considered to be the most appropriate spatial unit for analysis of the interaction between agriculture and water quality. Modelling of alternative policy instruments to control nitrate pollution is often carried out at this scale (Kampas & White, 2004) and many environmental protection agencies work at the catchment level acknowledging that water quality problems can best be solved at this level (Johansson, Gowda, Mulla, & Dalzell, 2004). Catchments also have the advantage of definite boundaries and negligible pollutant flow between the catchment and other areas (Schou, Skop, & Jensen, 2000).

Accurate estimation of pollution at the catchment level is a fundamental requirement for effective modelling of policy implications. While several studies have modelled farm nutrient discharges at the catchment scale, the impact of policies on individual decision making units and the reactions of those units to these policies have generally not been analysed. Instead the entire catchment has been treated as a single decision making unit. (Aftab, Hanley, & Kampas, 2007; Borisova, Shortle, Horan, & Abler, 2005; Chalmers & Crabtree, 1999; Taylor, 1992). However Brady (2003) did consider spatial differences within a catchment at a broader scale.

The catchment approach tends to overlook spatial interactions across decision making units. Although water quality is formed at the catchment scale, it is usually farms that must take action to improve water quality and it is farms that face the economic impact of environmental policies. This provides a strong rationale for a realistic farm centric modelling framework for agri-environmental policy analysis. This argument is further supported by Newell and Stavins (2003) who suggest that modelling spatial heterogeneity at the level of individual decision making units can be useful for designing market based policy instruments and De Cara, Houze, & Jayet (2005) who stress the usefulness of a farm centered approach in abating green house gases. However modeling all farms at the catchment level is a data intensive process that posses a number of challenges. One difficulty is caused by the fact that farms are generally not surveyed at the spatial unit at which ecosystem services function (Lant et al., 2005).

The objective of this chapter is to extend the usefulness of existing farm economic and geo-physical data to analyse the economic and environmental impact of various policies using a spatial micro-simulation. The method involves an early application of this technique to combining survey and population units to model environmental policy outcomes. The application focuses on a catchment in the Waikato region of New Zealand where water quality problems posed by nitrogen from the dairy farming sector are a major concern (Environment Waikato, 2005a). The model applies to all dairy farms[1] within the catchment since they are the major decision making units that can respond to policy changes.

Ballas, Clarke, Dorling, Rigby, & Wheeler (2006) define micro-simulation as the creation of simulated population micro-data sets for the analysis of policy impacts at the micro level. This approach has often been used for regional or local approaches to policy analysis (Ballas, Clarke, Dorling, Rigby, & Wheeler, 2006; Ballas, Clarke, & Wiemeres, 2006; Lymer, Brown, Payne, & Harding, 2006) and in welfare analysis and medical science but the only agricultural application is provided by Vrolijk (2004) who combined data from sample farms with the Netherlands agricultural census to obtain regional farm production characteristics. It is believed this chapter

[1] Ideally, the cumulative effect of all land uses in the catchment would be considered. However in the catchment of interest, dairy farming is responsible for a high proportion of nitrogen exports, hence only dairy farms are considered in this study.

describes the first use of micro-simulation to model virtual population of farms for analysing water quality related policies at a catchment scale.

The remaining parts of this chapter are organised as follows, with method followed by respective results in each section. Section 2 describes the rationale for micro-simulation approach. Section 3 describes an overview of the modelling process, discusses geo-spatial analysis and missing data estimation. Micro-simulation is explained in section 4. Later sections briefly discuss application, potential uses and limitations.

4.2 Rationale

Policy analysis in New Zealand is generally carried out at the national and regional level and at the level of the farm since this is the main decision making unit in New Zealand agriculture. There is a tendency, however for policy analysts to consider the impact of proposed policies on ‘typical farms’. Such an approach can provide a misleading abstraction from reality in a sector that is far from homogeneous and where few ‘typical farms’ exist. In reality farms exhibit considerable heterogeneity in all important variables from managerial ability and attitudes to soil type and use of inputs, to profitability and risk aversion. It is essential to take account of this farm level heterogeneity if the impact of policies is to be effectively modelled. It is also highly desirable that these different variables be combined in one data set in order to link geographical and management issues in agri-environmental analysis (Fais, Nino, & Giampalo, 2005).

Various authors provide evidence of the effect of farm heterogeneity on the cost effectiveness of environmental policies (Martinez & Albiac, 2006; Wossink, Lansink, & Struik, 2001). Martinez & Albiac (2006) showed that standards were cost efficient relative to input taxes in landscapes with low productivity and higher nitrogen discharges, because they achieve a high level of abatement of nitrogen emission but only cause moderate reductions in farm profit.

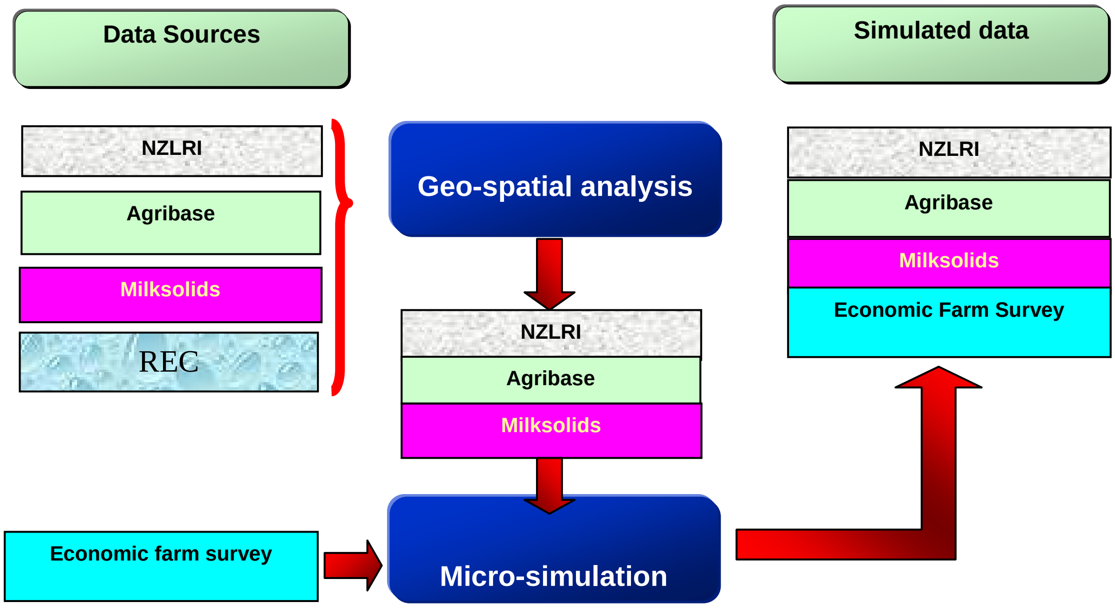
Farm survey data can enable analysts to take account of heterogeneity among farms, for example Muniz & Hurle (2006) coupled farm surveys with mathematical programming, to model the actual farm level decision making process. However catchment level application of survey data sets is a major challenge since surveys usually collect data for geopolitical units. For instance in New Zealand, agricultural production statistics such as the Economic Survey of New Zealand Dairy Farms, and the Sheep and Beef Farm Survey and Farm Monitoring Reports use farm surveys to produce statistics on a national or regional basis while agri-environmental policy analysis requires farm level data at the catchment scale. Another problem associated with direct use of survey data for catchment analysis is that there may be too few survey farms in the catchment of interest and that the exact location of these farms is unavailable for privacy reasons.

In New Zealand there is no single data set that provides spatially referenced farm management data so the development of a method of extending the usefulness of existing survey data is of particular interest especially when collection of additional data is difficult due to financial and time constraints. The development of an efficient method for linking different data sources should also enable a more comprehensive and reliable understanding of catchment level issues.

4.3 Data and methods

Geographic Information Systems enable the combination of geo-referenced data from different sources in a consistent manner to derive information on site specific characteristics. The major data sources used in this micro-simulation are illustrated in Figure 4.1 and reviewed in this section. Catchment modelling is implemented using two software systems, ArcGIS (ESRI, 2005) and Stata (StataCorp, 2005). ArcGIS is used for the data handling and spatial analysis functionality, allowing explicit representation and analysis of farms' bio-physical features and map based visualization. Stata is used for identifying predominant geographic attributes within each farm, merging of different data sources and for micro-simulation.

Figure 4.1 An overview of spatial micro-simulation



4.3.1 Geo-spatial analysis

In order to demonstrate the micro-simulation method, a catchment of the Waikato river was delineated using the New Zealand River Environment Classification (REC)[2]. The catchment stretches over 155,303 ha, from Lake Karapiro to Lake Arapuni and has been identified as part of the catchment in the region with highest priority for nutrient management (Brodnax, 2006; Environment Waikato, 2005c). The main stages of the geospatial analysis are illustrated in the Figure 4.2.

The catchment boundary layer was intersected with Agribase, a spatial land use database that includes the boundaries of farm and forestry land parcels and stock numbers for each farm property in the catchment. Properties which fall across catchment boundaries may create difficulties for administration and enforcement of catchment based policies, but leaving such properties out of the catchment would

---

[2] REC organises information about the physical characteristics of New Zealand's rivers by the source of flow for the river water, the catchment geology and land cover and maps this information by river segment for the New Zealand river network.

complicate the achievement of nutrient targets. Therefore these whole properties are defined as being within the catchment. The resulting catchment land use layer is illustrated in the Figure 4.3. Light green and conical symbol areas are pastoral and forest land respectively. Dairy farming and forestry are major land uses accounting for 82 percent of land use. The distribution of catchment land uses are described in Table 4.1.

Figure 4.2 Stages of Geo-spatial analysis

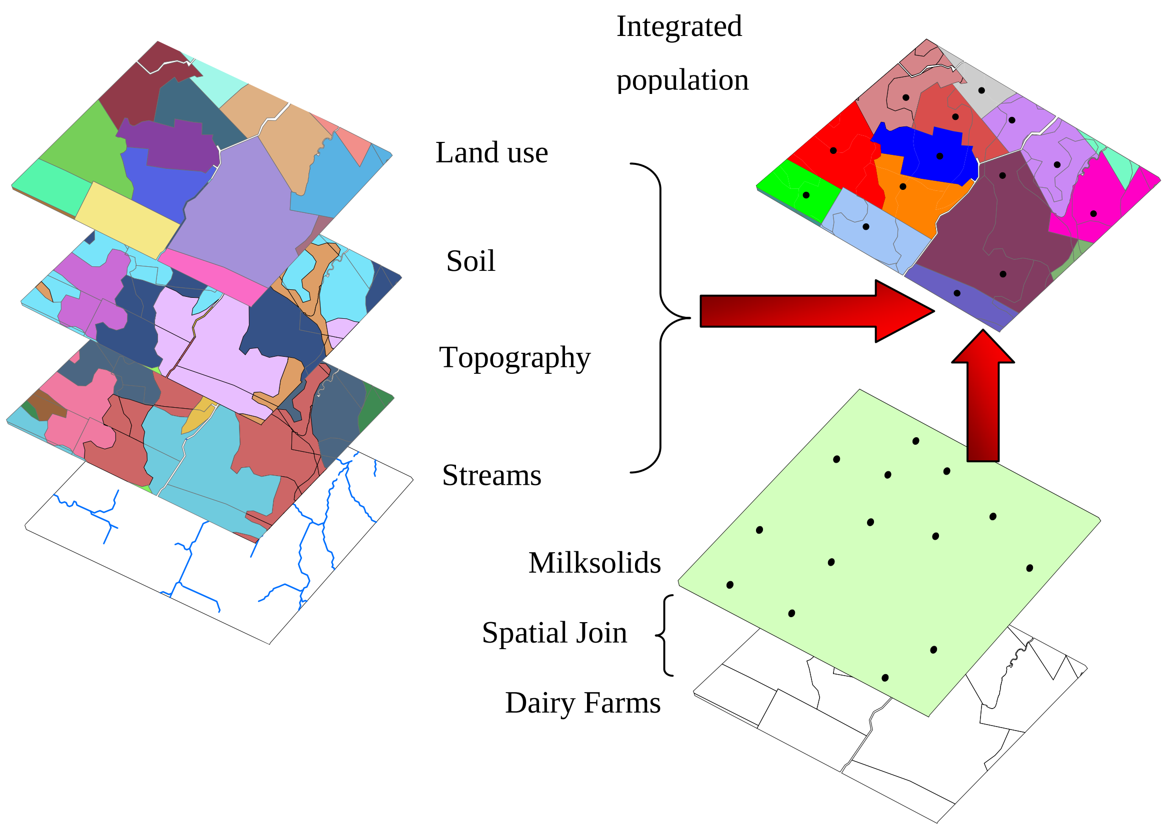


Figure 4.3 Catchment land use

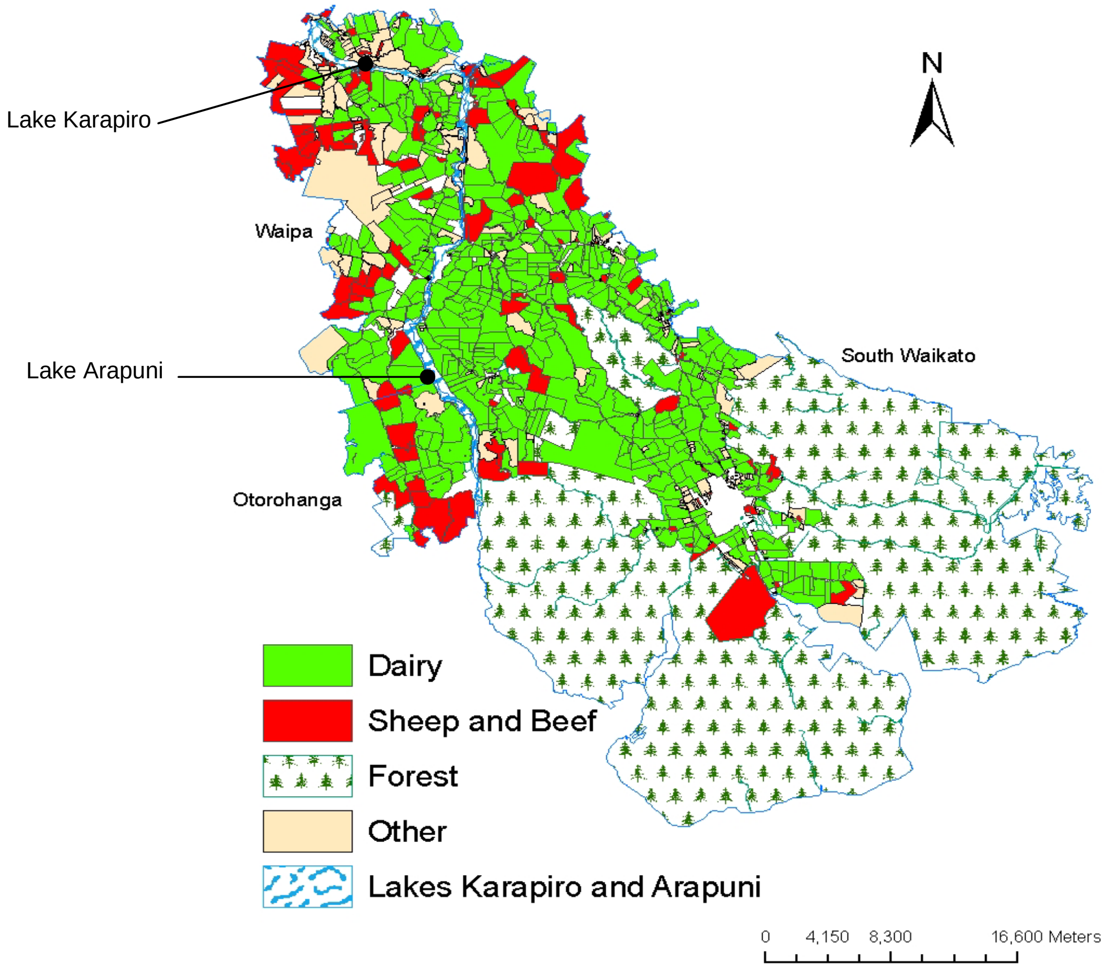


Table 4.1 Major land uses in the catchment

Land use	Number of properties	Area (ha)	Percent land use
Dairy	370	52887	34.3
Lifestyle	183	856	0.6
Beef	62	3708	2.4
Dry stock	48	4627	3.0
Grazing	40	1124	0.7
Sheep and Beef	38	8635	5.6
Deer	12	686	0.4
Sheep	11	1654	1.1
Forestry	6	74711	48.4
Native bush	3	2881	1.9
Idle land		1968	1.3
Other	30	696	0.5

4.3.2 Classification of soil type and topography

The potential for nitrogen loss varies with soil physical properties and topography (VanDyke, Bosch, & Pease, 1999), for example steeper slopes and some soils are more prone to nitrogen leaching (Judge & Ledgard, 2004; VanDyke, Bosch, & Pease, 1999). Parker & Litchenberg (2004) showed the effect of proximity to water bodies, soils and topography on nitrogen discharges. Therefore making an accurate representation of these factors is important. In the next stage geo-physical characteristics from the New Zealand Land Resources Inventory (NZLRI) were intersected with catchment land use to derive the distribution of soil and topographic classes across farms. NZLRI provides detailed spatial information on land geo-physical attributes within the catchment broken down into 3393 polygons each of which describes a parcel of land in terms rock, soil, slope, erosion and vegetation attributes.

Using geo spatial statistical techniques in ArcGIS and Stata the area of each polygon[3] that intersects each farm boundary was calculated to identify predominant soil type and the topographic feature of each property. Soil type based on the New Zealand Soil Classification (NZSC) described by Hewitt (1998), is an attribute within the NZLRI. Details are provided in Appendix 1.1. Each farm consists of numerous soil subgroups but these are aggregated into major soil orders and soil groups to meet the data requirements of Overseer, a farm scale nutrient budget simulation model which accounts for variability in soil type and topography (Wheeler, Ledgard, DeKlein, Monaghan, & Carey, 2003). Table 4.2 provides classification details for the soils in the catchment.

Overseer was used because of the cost and time lag involved in directly measuring nutrient discharges from farms. The Overseer model is widely used in New Zealand

[3] Polygons are continuous two-dimensional objects, encoded in the database as a sequence of locations that define the boundaries of each closed area in a specified coordinate system. The attributes of each polygon is stored in the database as well.

to estimate farm nitrogen discharges and has been used to design rules regarding nitrogen discharges from farms by regional councils (Horizons Regional Council, 2007).

In NZLRI, topography is categorized into seven different classes (Newsome, Wilde, & Willoughby, 2000). The major topographic categories found in the catchment being described in Table 4.3. Details of this categorisation can be found in Appendix 1.2. The distribution of major soil types and topographic classes across the catchment is illustrated in Figure 4.4. 'Volcanic' and 'pumice' are the predominant soil types with 'easy' and 'rolling' being the major topographic classes.

Table 4.2 Classification of major soil types in the catchment

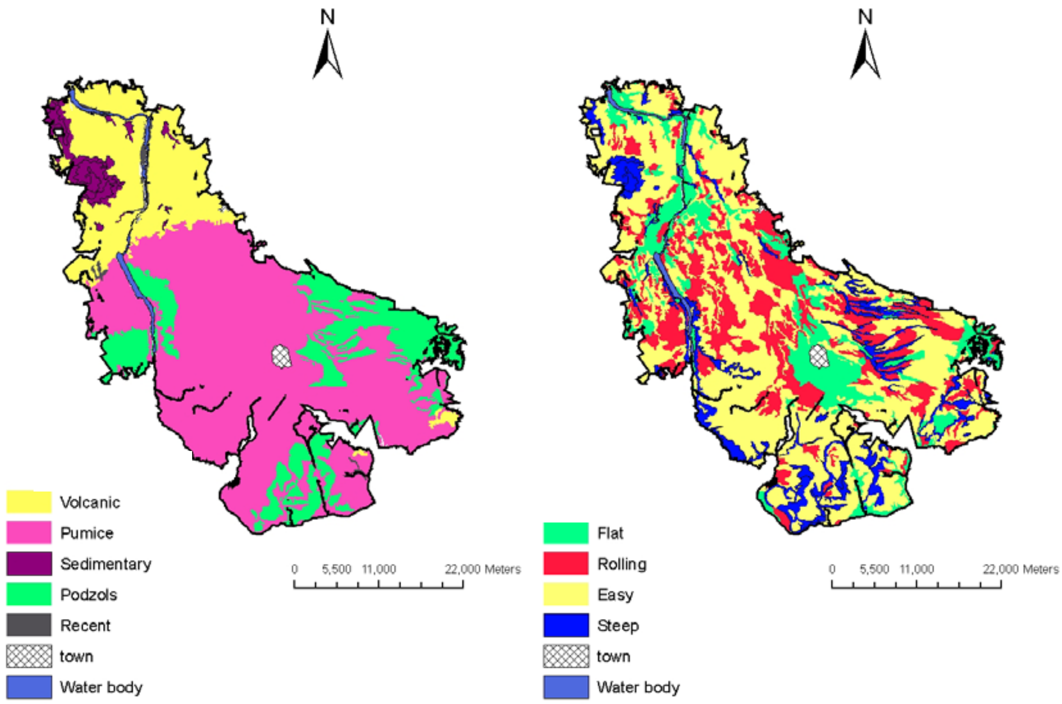
Soil group (Overseer)	**Soil order**	**Soil sub group**
Sedimentary	Brown	BOT
	Gley	GOT GRO
Volcanic	Allophanic	LOT LOT-GOT LOV
Pumice	Pumice	MIW MOI
Peat	Organic	OHM
Recent	Recent	RFM ROM RST RTT
Sands	Raw	WX
Podozols	Podzols	ZOH ZOT

Source: Hewitt (1998)

Table 4.3 Major Topographic classes in the catchment

Topography	Slope (Degrees)	Slope categories
Flat	0-7	A---B
Rolling	8-15	C
Easy hill	16-25	D---E
Steep hill	>25	F---G

Figure 4.4 Soil type and topography



4.3.3 Production variables

The major production variable on New Zealand dairy farms is milksolids production representing the combined weight of milk fat and milk protein (Dexcel, 2006). Milksolids production is a major determinant of dairy farm income in New Zealand where farms are paid according to the quantity of milksolids produced. About 92% of the gross farm revenue of Waikato dairy farms is derived from milksolids (Ministry of Agriculture and Forestry, 2005a). Data on milksolids collected from Fonterra's milk collection points within the catchment for three consecutive years from 2003 to 2006 was made available for this study under stringent privacy conditions. Fonterra is the dominant processor of raw milk in New Zealand and collects around 96 percent of milk produced in New Zealand (Ministry of Agriculture and Forestry, 2005b). The milk collection points are georeferenced[4] using Global Positioning System (GPS) co-ordinates to create a vector layer of milk collection points[5]. The vector layer consists of 346 collection points. Milksolids production data for each point was stored in the attribute table of the vector layer.

The milksolids vector layer was spatially joined with the farm boundaries data using the spatial join function in ArcGIS. Spatial join constructs a table in which fields from one layer's attribute table are appended to another layer's attribute table based on the relative locations of the features in the two layers. When more than one milk collection point falls in a single farm polygon these points are aggregated within the polygon in order to generate milksolids production per farm. As a result of the spatial join 317 dairy farms were joined with milk collection points. This accounts for 87% of farms in the catchment. The remaining 47 (12%) out of 364 farms lacked milksolids production data. One obvious reason for this is the lack of geo referencing for 31 milk collection points. Another possible reason may be the existence of dairy farms not supplying milk to Fonterra.

Discarding farms lacking milksolids production could lead to loss of substantial information on catchment-wide nitrogen discharges since nitrogen discharges are largely a function of farmed area, farming intensity and the location of farm relative to water bodies in the catchment (Johnes, 1996; Vant & Smith, 2004). Since finding missing data directly is likely to be difficult and expensive, an estimation method is used as detailed below.

---

[4] Geo referencing is the process of aligning geographic location data to a known coordinate system by latitude and longitude to create an image file of points and polygons.

[5] Point data specifies a location by a coordinate pair of XY values.

4.3.4 Missing data estimation

Missing data can be estimated using various techniques such as mean imputation, regression and multiple imputation. Mean imputation typically uses the mean of complete case observations to impute missing data (Allison, 2002) but tends to reduce the variation in the population.

In another approach the data set is used to regress variables and create a regression model with missing values being imputed from predicted values. The relationship between the dependent variable milksolids and independent variables namely farm size and number of cows are explored to select variables and the appropriate functional form. In this case the linear functional form was found to be the most appropriate as follows (equation 1):

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + E \qquad (1)$$

Y Milksolids per ha

X_1 Stocking rate (number of cows per ha)

X_2 Farm size (ha)

β_0 , β_1 & β_2 are intercept and slope parameters. E –Error term with $N(0, \delta)$

The intercept term accounts for other factors which influence farm production. The regression parameters are estimated by least squares applied to complete farm records for stocking rate, farm size and milksolids per ha. A similar approach was used to estimate missing values for a continuous variable by Penn (2007). Parameter estimates and t statistics of the regression are given in equation 2. Given the cross sectional nature of the data the model has a good fit (Adjusted R^2 of 0.52 good for a cross sectional regression). Replacements for the missing values were generated from the fitted model (Figure 4.5 a).

$$\underset{(3.11)}{Y= 166.95} + \underset{(17.57)}{38.05\, X_1} - \underset{(3.11)}{0.16\, X_2} \qquad (2)$$

*Figures in parenthesise are t statistics.

According to the p –values of the intercept term (0.002), slope parameters farm size (0.27) and stocking rate (0.000), the stocking rate has significant influence in determining the milksolids production per ha. This is consistent with production theory. The weak and negative relationship between farm size and productivity suggests that milksolids production per hectare falls as farm size increases. This is consistent with existing findings for New Zealand dairy farming (Glassey, 2001). The next section outlines another method of estimating missing data known as multiple imputation.

Multiple imputation

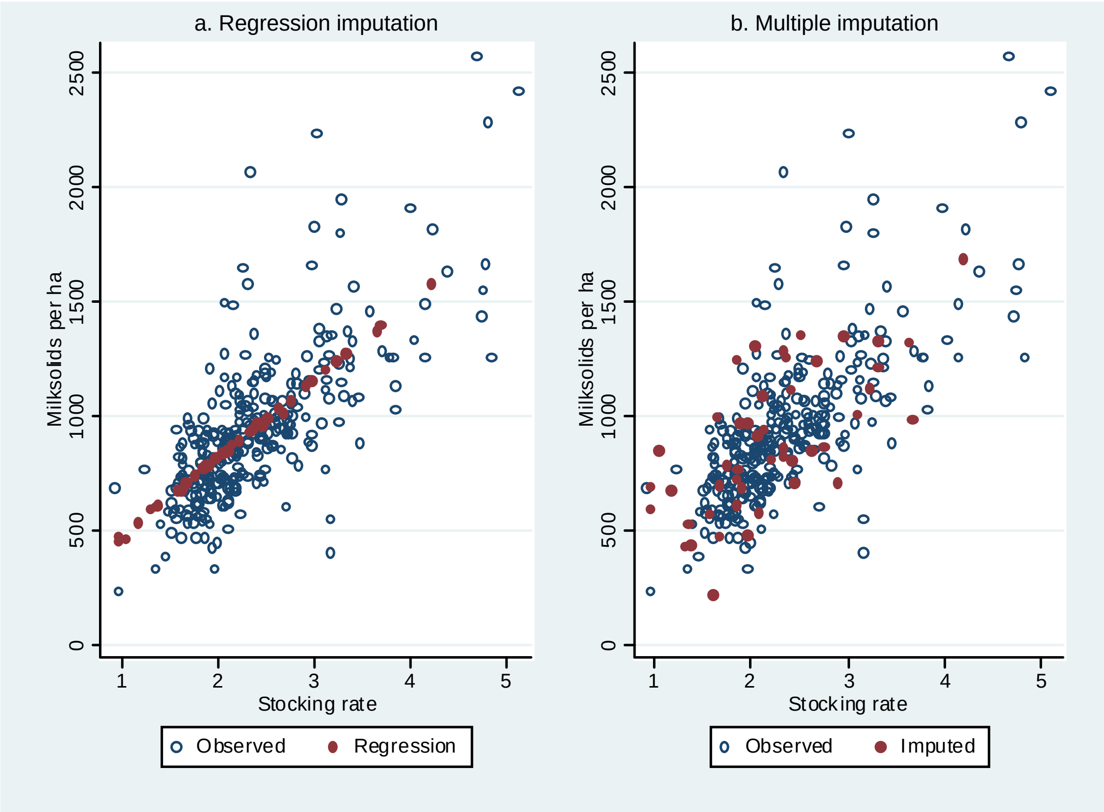
Multiple imputation produces better outcomes than regression based on single imputation when more than 5% of data is missing (Schafer, 1999) and so is used to replace missing milksolids data. Multiple imputation is mainly used to deal with non responses in surveys (Rubin, 1987) but has been broadly applied to a variety of missing data settings. Lokupitiya, et al (2006) used multiple imputation to estimate missing values for crop yield data in the United States census of agriculture and agricultural statistical surveys.

Several computational methods have been developed for multiple imputation. The univariate imputation algorithm defined by Van Buuren, Boshuizen, & Knook (1999) is most appropriate for missing data where only dependent variable observations are missing. It provides a procedure for imputation of missing values for a single dependent variable (milksolids per hectare) given a complete set of independent variables (stocking rate and farm size). Missing values for continuous variables are estimated using linear regression. In the first step a vector of beta coefficients and residual variance are estimated by regressing the non-missing values of dependent variable on the current completed (original) version of independent variables. Thereafter a random value is drawn from the posterior distribution of the residual standard deviation to account for uncertainties in beta coefficients. These uncertainty accounted beta coefficients are used to predict missing values using independent variables. This process is repeated *m* times. This process assumes the dependent variable is normally distributed and there is no systematic difference between complete and incomplete records. A skewed dependent variable tends to produce implausible imputations. This is one of the reasons for choosing Milksolids production per ha as a dependent variable rather than milksolids production per farm which is highly skewed. Technical details of the algorithm for creating multiple imputations are detailed in pages 689-691 of Van Buuren et al (1999).

A computational algorithm, *UVIS* (Univariate Imputation Systems) has been developed for this procedure by Royston (2005). In this process OLS is used to predict continuous variables and the functional form detailed in [1] above is retained for prediction. In our case data is missing for 12 percent of the population so imputations are executed five times based on Rubin (1987) and Schafer (1999). Rubin demonstrated the relative efficiency of an estimate based on *m* imputations and the percentage of missing data. For instance if 10% of data is missing, imputing 5 times achieves 98% efficiency. Missing values imputed using multiple imputations are illustrated in the Figure 4.5 b.

Milksolids production is largely determined by stocking rate and management practices, but the effects of inherent soil fertility and topography on pasture production are also important (O'Connor, 1982; Roberts & Morton, 2004). These factors are not captured in the multiple imputation process adopted here due to the low number of observations in some soil and topography classes. While multiple imputations cannot substitute for quality primary data collection the method adopted here provides a pragmatic and cost effective solution that enables catchment scale modelling.

Figure 4.5 Missing data imputation



4.3.5 Riparian margins and location of farms

In addition to the soil and topographic features and production variables described above, farm location relative to streams and waterways influences nitrogen discharges to surface water. For example, N deliveries have been found to be higher for fields located closer to streams and with steeper slopes (Peng & Bosch, 2001).

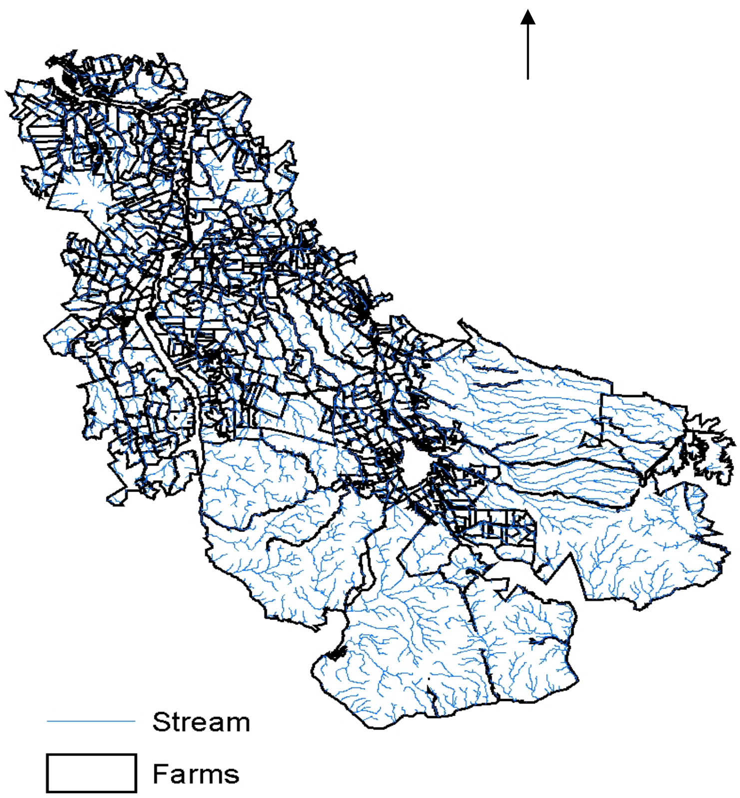
In this section the stream length within each farm is estimated by intersecting the REC data base with farm boundaries (Figure 4.6). The stream length attribute is then aggregated within each farm[6] and multiplied by two to get the length of riparian margin in each farm. The minimum distance between each farm and the main stem of

[6] Dissolve function in ArcGIS is used for aggregation

the river is calculated by estimating the distance between the Centeroid[7] of each farm polygon and the main stem[8] of the water body.

The length of stream on each farm averages 1.59 kms or 3.18 kms of riparian margin per farm, see Table 4.4. The distance to river values and the length of riparian margin can be used with appropriate transmission coefficients to estimate nitrogen discharge from farms into the river and to enable analysis of the impact of interception technologies on nitrogen abatement.

Figure 4.6 Distribution of streams



[7] A Centeroid is a polygon's mean centre which is based on the weighted average of it's x and y geographic coordinates.

[8] The centre line of the lake and river polygon on the topographic map.

Table 4.4 Descriptive statistics of farm riparian margin and distance to river

Variable	Mean	Standard deviation	Minimum	Maximum
Distance to river(km)	6.46	5.20	0.20	18.38
Riparian margin length(km)	3.18	2.44	0.00	13.40

4.4 Micro-simulation

Micro-simulation models enable analysis of the spatial implications of policy changes at the micro level. The technique is new and is increasingly used for merging population and survey data to synthesize micro data for a virtual population. This micro data may then be used to analyse the effect of alternative policies both on individual units and on the population as a whole.

In the micro-simulation procedure adopted for the Karapiro catchment, the geo referenced farm population is supplemented with additional microeconomic information from farm surveys using a statistical matching technique. This procedure will now be described in detail.

4.4.1 Catchment farm population and farm survey data

The data available for farms in the catchment are limited to milksolids production from Fonterra, farm size and stock (cow) numbers from Agribase and soil and topographic attributes from NZLRI. Stock numbers reported in Agribase are not broken down by age class so it is assumed that the number of milking cows in each farm is 80 percent of the total number of cattle. This is based on the typical replacement herd size (20 percent of stock) on Waikato dairy farms (Ministry of Agriculture and Forestry, 2005a).

The Economic Survey of Dairy Farmers is the major micro economic data source for dairy farming in New Zealand. This nationwide survey was conducted annually by DairyNZ, which is the principal research and extension arm of New Zealand's dairy industry. The Economic Survey covers detailed physical inputs, financial variables and capital structure. Farm level survey data for years 2003/04 and 2004/05 for the Waikato region was provided by DairyNZ for the purpose this study, covering an average of 150 farms in each year.

Owner operators comprise the largest group of dairy farms, accounting for 65% of the New Zealand total (Livestock Improvement Co-operation, 2008). Around 60% of survey farms are owner operated most of the remainder being operated under sharemilking arrangements. Following Neal (2004), sharemilker farms were removed from the data set in order to avoid inconsistencies resulting from reporting of variable shares of costs and revenues as sharemilker farms' operating structures vary from less than 20% share to over 50%. The farm economic survey data set thus consisted of data for 175 owner operator farms for the years 2003/04 and 2004/05. The data set includes 120 attributes for each farm covering physical input use and financial variables, All financial variables were adjusted to 2004/05 prices using Consumer Price Indices (CPI). Since the dedicated data collection from catchment farms is constrained by time, cost and infrastructure, there is an advantage for acquiring additional information about the farms in the catchment using the strength of the Economic Farm Survey.

4.4.2 Method of micro-simulation

Economic Farm Survey data is used to simulate variables for farms in the catchment as follows. For each farm in the population[9], a farm in the survey sample is selected which resembles the population farm as closely as possible, based on certain variables. These auxiliary variables are variables must be known for all farms in the survey and the population. In this study milksolids, number of milking cows and farm size are known for both survey and population farms. Survey farms are matched to population farms based on the minimum difference between selected auxiliary variables known as imputation variables.

---

[9] all dairy farms in the catchment

Selection of variables

The selection of imputation variables is of crucial importance to the quality of the simulated end result. Simkin, Verwaart, & Vrolijk (2005) used a genetic algorithm to select imputation variables but stressed the importance of human expertise in the selection of the imputation variable. The imputation technique assumes that if a farm is similar based on imputation variables, then it is likely that other variables are also similar. For this to be true there must be a logical relationship between imputation variables and other variables. The key variables, which are important in the analysis of economic and production aspects of dairy farming have been identified from the literature.

In New Zealand, milksolids, farm size, stocking rate, variable costs, labour, feed supplements, dairy farm operating profit[10] , total revenue, net operating assets, and fertilisers applied are most commonly used in economic studies (Jaforullah & Premachandra, 2003; Jaforullah & Whiteman, 1999; Kompas & Che, 2006; Neal, 2004; Reinhard, Lovell, & Thijssen, 1999). Milksolids, farm size and number of milking cows are good indicators for production, intensity of land use and enterprise size and strongly influence input use, capital requirements and pollution potential in agricultural production systems.

Vrolijk (2004) suggests that analysis of the correlation between auxiliary variables and variables which are only known for survey farms may be useful in selecting imputation variables. Therefore the strength of the relationship between the auxiliary variables and other key variables in the Economic Farm Survey is examined by analysing correlation coefficients. Analysis reveals strong correlation between the hypothesised auxiliary variables and other key variables (Table 4.5) so all three auxiliary variables were selected as imputation variables. The lower correlation between brought in feed and farm size may reflect the ability of larger farms to

---

[10] Dairy farm operating profit is a measure of dairy farm profitability indicating dairy operating return after an allowance for the value of change in dairy livestock numbers, non-paid labour and management, supplementary feed inventory change, owned run-off adjustment and depreciation (DairyNZ, 2008).

produce more feed on farm. In the presence of many common variables, selection of variables and allocation of relative weights to the variables can be implemented using stepwise regression (Decoster, Standaert, Standaert, Valenduc, & Van Camp, 1998.).

Table 4.5 Correlation between auxiliary and other important variables

Variables	**Milksolids (kg)**	**Farm size (ha)**	**Number of cows**
Nitrogen fertilizer applied per farm (kg/ha)	0.74	0.79	0.71
Feed brought in (dry matter)	0.54	0.46	0.63
Economic farm surplus ($)	0.67	0.64	0.70
Feed costs ($)	0.59	0.50	0.72
Labour costs($)	0.82	0.76	0.80
Net operating assets	0.80	0.78	0.79

Distance function

Once imputation variables are selected, an appropriate distance function must be specified to calculate the distance between imputation variables in the population and in the survey. Euclidean distance is generally used to measure the linear distance between two points in geometry, when all variables are measured in the same unit. However Euclidean distance is extremely sensitive to the scales of the variables involved. Since the scales of imputation variables involved in this study vary, it is inappropriate to use the generic form of the Euclidean distance function, particularly when applying weights to differentiate the importance of imputation variables. Because of this scale effect, some variables would be penalised more than others, purely due to the scale of the variable and not its importance in the matching process.

In this circumstance, normalisation is often used so that some attributes do not arbitrarily get more weight than others (Lymer, Brown, Payne, & Harding, 2006; Vrolijk, 2004; Yoshizoe & Araki, 1999). The normalised Euclidean distance accounts for the scale differences among imputation variables.

The distance function is defined following Decoster & Van Camp (2000) as follows:

$$D_{jk} = \sum_{i \in M} W_i \frac{\left| X_{ij} - X_{ik} \right|}{\sigma(X_{ij})} \tag{3}$$

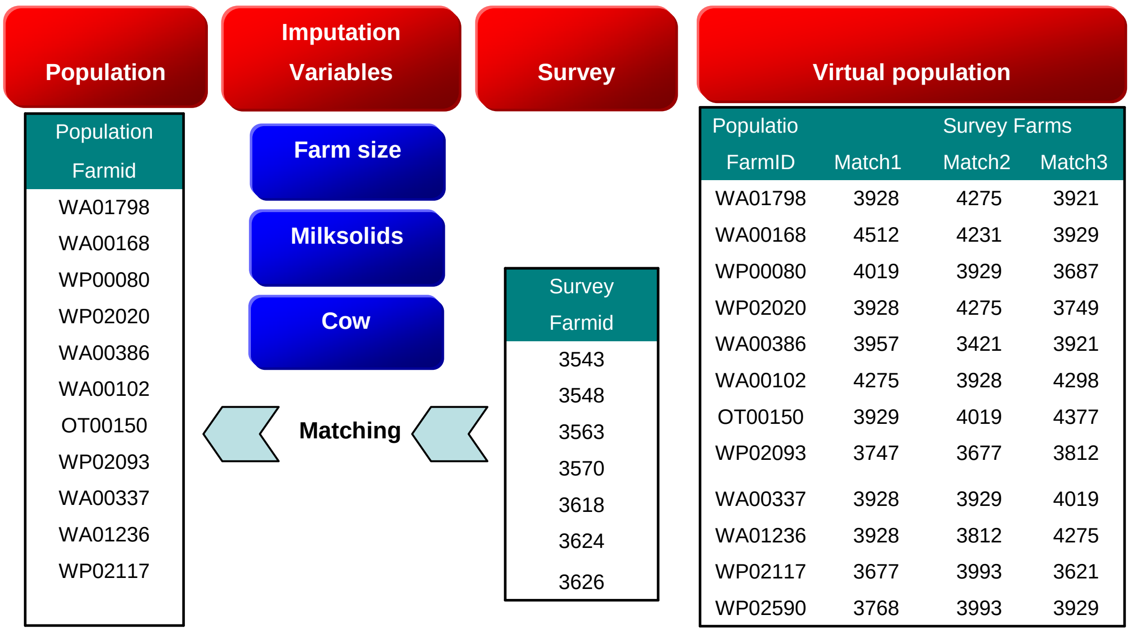
M	is a set of auxiliary variables used in the matching process
X_{ij}	the value of the auxiliary variable i for the population unit j
X_{ik}	the value of auxiliary variable i for the survey unit k
W_i	Weight of auxiliary variable i in the distance function
$\sigma(X_{ij})$	Standard deviation of the auxiliary variable i in the population unit j

The Euclidean distance between the population and survey imputation variables is normalised when divided by the standard deviation of the respective population variables. The survey farm which most resembles the population farm is then defined as the farm which has the minimum sum of normalised Euclidean distance. This farm is considered to be representative of the population farm. For each farm in the population, the three most similar farms are selected from the survey sample.

Imputation was implemented by a series of tailor made algorithms developed using the programming option of Stata statistical software with equal weight being assigned to each imputation variable. In the matching process, which is illustrated in Figure 4.7, farms in both the population and survey data are identified using unique identifiers (FarmID). The population consists of 364 farms while the survey covers 175 farms. The distance between each imputation variable is calculated for every population farm against all sample farms giving 191,625 distances. Distances for each farm are summed and ranked, the farm with the shortest distance being defined as the first best survey farm. After ranking two different types of matches are carried out. Firstly, each farm in the population is represented by the most similar survey farm (first best survey farm), described as a single match. Some survey farms are matched to more than one population farms. This results from the high degree of homogeneity among farms in the catchment.

As an alternative to a single match, multiple matching was also carried out. In multiple matching a set of 100 farms are created using random combinations of the three best matching farms. Then the variable values for each 100 farms are averaged to create a new matched farm for each member of the population. Multiple matching avoids the issue of identical matched farms. However multiple matching compromises the accuracy of the matching as it tends to tradeoff first best matches for second and third matches. Further multiple matching will not handle discrete variables for instance type of milking shed. Therefore choice of the matching method depends on the proposed analysis and the nature of the data on survey farms.

Figure 4.7 Matching process between population and survey farms



In Table 4.6 the descriptive statistics of the imputation results is presented along with the real farm population data. The values estimated by the imputation procedure are similar to the real values and there are no significant differences between the means and standard deviations of real and virtual populations. The accuracy of the imputation process was also examined by mapping each farm imputed value with real values where the degree of overlap of the real and imputed values indicates the accuracy of the matching process, see Figure 4.8.

Table 4.6 Descriptive statistics of imputation results and farm population

Variables	**Real**		**Single match**		**Multiple match**	
	Mean	Standard deviation	Mean	Standard deviation	Mean	Standard deviation
Milksolids (000 kg)	100.73	63.94	95.14	52.04	93.96	48.05
Farm area (ha)	113.88	71.53	106.62	60.15	103.56	56.21
Milking cows	252.68	151.75	273.14	161.50	272.28	154.13

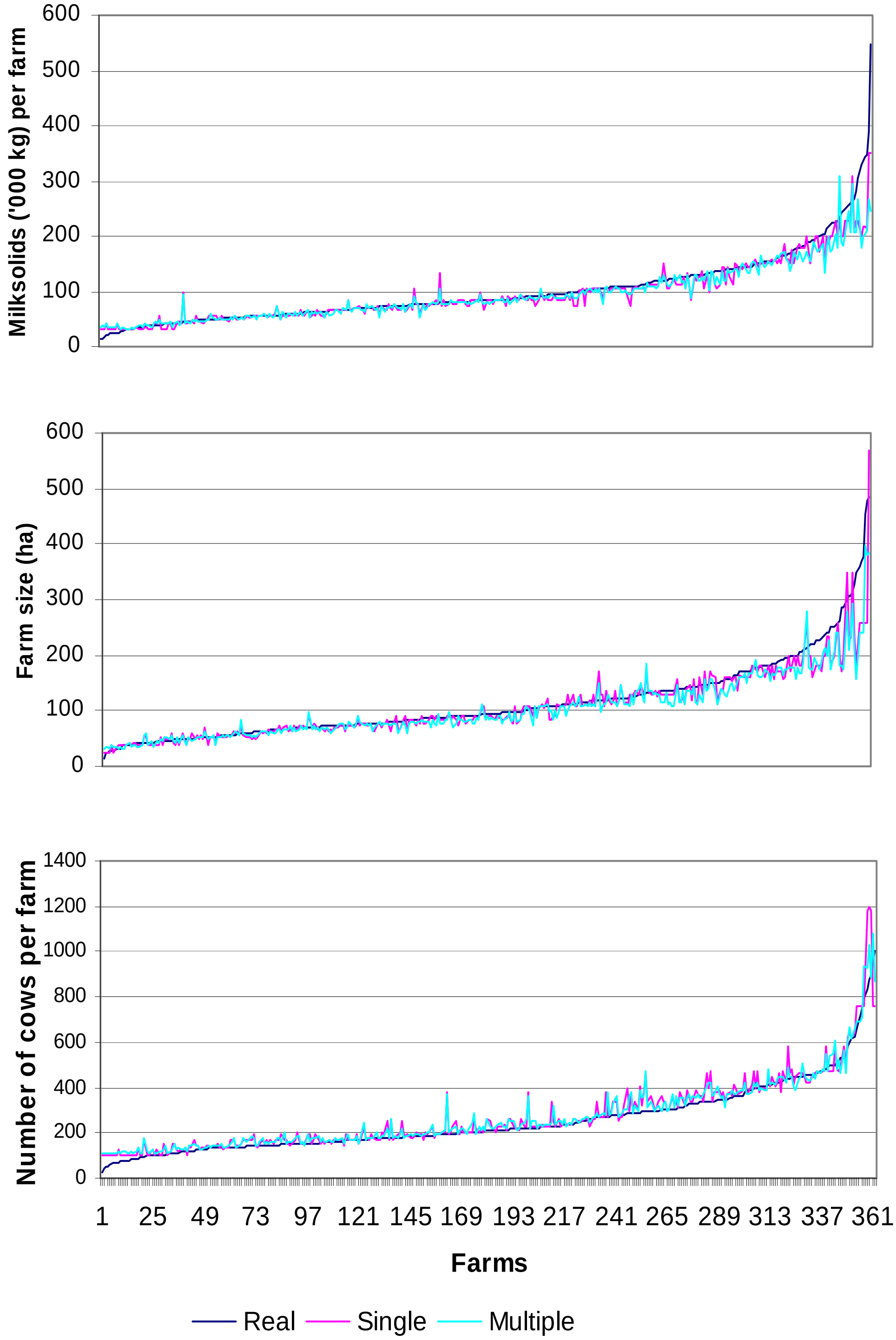
These distributions indicate an excellent match between real and virtual farms, using either single or multiple matching, except for the smallest and largest farms accounting for 3 percent of the population. The differences between the real and simulated variable catchment totals are compared in Table 4.7 where it can be seen that estimated milksolids production and total farm area are 5 to 6 percent lower than the real population totals. The estimated total number of milking cows is around 8 percent higher than the real value, this may be due to inconsistencies in the age structure of animals on Agribase.

Potential problems which could arise in circumstances like estimation of catchment wide nitrogen discharges from farms due to the difference in total farm area, could be alleviated by extrapolating per ha estimates of the discharge to real farm area.

Table 4.7 Comparison of estimated and real variables for the catchment

Variables	**Real**	**Single matching**	**Multiple matching**
Milksolids ('000 kg)	36500	34400	34000
Farm area (ha)	41226	38598	37490
Milking cows	91470	98877	98564

Figure 4.8 Comparison of real and simulated values of variables

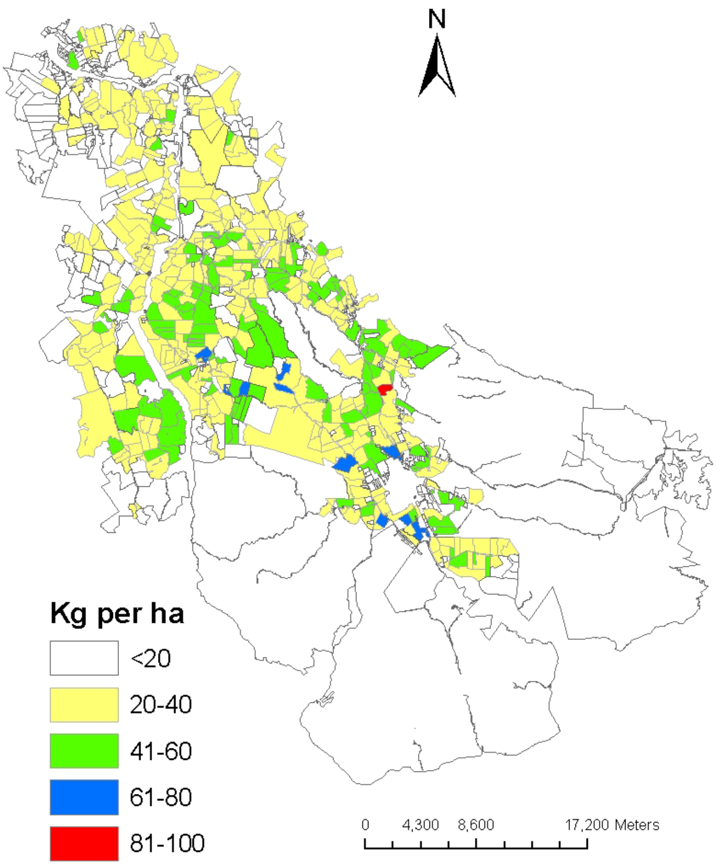


4.5 Application

In order to demonstrate the potential use of the virtual population, a simple application has been performed using single matched farms in which we map dairy nitrogen discharges and farm returns across the virtual farms in the catchment.

Nitrogen discharges are estimated by the Metamodel[11] built on the Overseer estimates. In this exercise farms in the catchment were assumed to be one of two predominant soil types and topographic classes, Pumice- Rolling and Volcanic-Easy[12]. Figures 4.9 and 4.10 indicate a large variability in nitrogen discharges per ha and dairy farm returns.

Figure 4.9 Simulated dairy farm nitrogen discharges per hectare

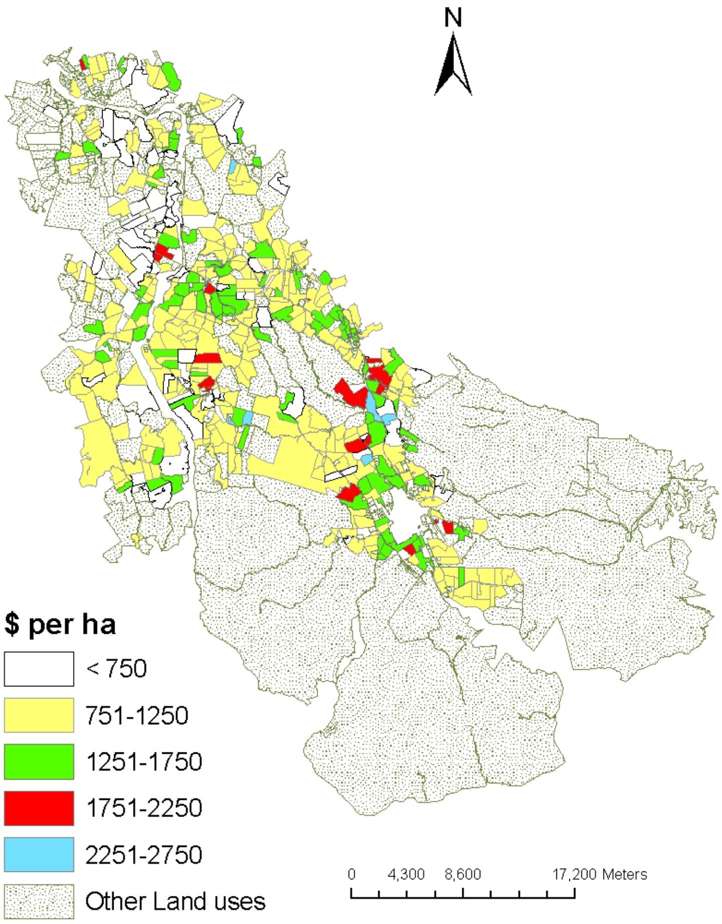


---

[11]As direct application of the Overseer over hundreds of production systems is time consuming a metalmodel is used. Details of the meta model are discussed in chapter 5.

[12] Pumice and volcanic soil types account for around 50 per cent of dairying land area

Figure 4.10 Simulated dairy farm returns per hectare



The nitrogen discharge information generated would allow estimation of overall discharges into the main stem of the river, when accompanied by appropriate distance based transport coefficients. Since individual farm nitrogen discharges and returns are commercially and socially sensitive, synthetic farm data may be regarded as a pragmatic compromise between reality and privacy for analysing policy.

4.6 Potential uses

One of the major advantages of micro-simulation models is scale flexibility since data can be aggregated or disaggregated to the preferred scale (Hynes, Morrissey, & O'Donoghue, 2006). For instance synthetic micro-data can be used to create catchment specific representative farming systems. Mathematical programming models can then be applied to these farming systems to simulate responses to policy then these responses can be extrapolated to the entire catchment. This was the approach taken by Payraudeau & van der Werf (2005) in extrapolating farm emissions from a linear programming model to a whole region.

Integration of more virtual data into simulation models can produce more attribute rich farm data that would enrich our knowledge of farming systems and its responses to policy shocks. For instance the simulated data can be fed into DairyNZ's whole farm model to predict responses to policy changes and into the nutrient budget model Overseer to quantify the impact of adopting best management practices such as winter pads, herd homes and effluent disposal systems and nitrification inhibitors.

Virtual data may also provide a useful source of information for simulating nitrogen trading using multi agent simulation models such as CORMAS (Bousquet, Bakam, Proton, & Le Page, 1998). Nitrogen discharge data may also be used to estimate the environmental and economic efficiency of farms based on nonparametric and parametric methods as reported in Coelli, Lauwers, & Van Huylenbroeck (2007) and Reinhard, Lovell, & Thijssen,(2000).

4.7 Limitations

The matching process was not effective for farms at the ends of size spectrum e.g extremely smaller or larger farms. Perhaps collecting data directly from such farms may be a pragmatic solution. Variables describing farm size and stock numbers, which often change in the short to medium time period may not be represented accurately, since frequent update of Agribase is expensive and time consuming. However enhanced use of remote sensing and satellite imagery could help to alleviate this problem.

Lack of validation for all variables is a disadvantage of virtual data. Given that the rational for creating virtual catchment data is that it does not currently exist, lack of validation may be justified.

Survey farms do not contain information on land quality. If survey farms were geo referenced it would have enabled identification of geographic attributes of the farms thus more effective matches could have been made with population farms considering geophysical attributes. However it can be justified that matching based on production potential, stock numbers and farm size sufficiently represent soil quality.

The model assumes that each farm has homogeneous soil and topography for computational convenience. However considering intra farm heterogeneity may be unrealistic due to the difficulties in restricting mobility of animals within certain soil and topography. Further it makes policy implementation difficult.

The model developed does however omit factors such as individual farmers' attitude, intrinsic knowledge base and experience and risk aversion etc. These are important influences on decision making which we recognise and fully aware as a limitation of the model. Direct data collection from the farms is clearly optimal given no cost, time and effort constraint. However, given such constraints the approach taken substantially overcomes data limitations.

4.8 Conclusion

The method developed with a modest amount of information can be employed to analyse environmental policy in a spatial context. The model integrated attributes of geo referenced farm population with Economic Farm Survey for dairy farmers. The Economic Farm Survey until now has mainly been used for production economic analysis. This approach shows the potential use of the survey data for spatial environmental policy analysis. Synthetic data would provide better understanding of catchment farms for policy analysis. It can be extremely valuable to regional councils and other policy making organisations that want to minimise the economic impact of environmental policies.

Using the Economic Farm Survey data for catchment level modelling saves time and resources for dedicated data collection. In addition the Economic Farm Survey is more reliable as DairyNZ (Dexcel) has greater resources and better access to dairy farms and annually produces farm statistics with national and regional scope. This is especially so with the recent changes introduced to data collection and management with the establishment of Dairy Base Economic Survey as a substitute to the Economic Farm Survey, which intends to target more farms with more specific details such as feeding regime and best management practices. There is a potential for extending the method to build a spatial dynamic micro-simulation model given the availability of time tagged population variables and panel survey data.

The virtual population results in this chapter has been used to derive marginal abatement costs of different farming systems in chapter 5; model cost effective policy implementation in chapter 6; measure environmental and economic efficiency in chapter 7 and model optimum riparian buffers in chapter 8.

5. An integrated simulation model to assess economic and environmental impacts of dairy farm systems

5.1 Introduction

This chapter presents an integrated simulation model to assess economic and environmental impacts of dairy farm systems with a focus on the nitrogen pollution in the catchment as described in Chapter 3. Alternative methods for abatement of nutrient discharges can be broken down into changes at the intensive or the extensive margin of production (Xabadia, Goetz, & Zilberman, 2006; Yiridoe & Weersink, 1998). Changes at the extensive margin involve retiring farmland or changing the size of the effective farm area by establishing riparian margins and wetlands, or by changing the use of production units. At the intensive margin, abatement can be realised by a reduction in the level of nitrogen input per unit of land. In dairy farming this would involve adjustments to the level of nitrogen fertiliser application and to the stocking rate. Limiting stock numbers has been proposed as a method of reducing nitrogen discharges into water in ecologically sensitive water bodies in New Zealand (MacDonald, Connor, & Morrison, 2004). The modeling approach described in this chapter focuses on changes at the intensive margin.

Just & Antle (1990) emphasize the importance of modelling agri-environmental policies at a disaggregated level to capture the heterogeneous nature of the physical environment and economic behaviour among farms. For instance nitrogen loss potential increases with the rise of soil slope and higher on soils more prone to leaching (VanDyke, Bosch, & Pease, 1999). Establishing the cost of nitrogen discharge reduction at farm level is essential for making informed policy decisions, and solutions to pollution control problems require a knowledge of farm specific abatement cost (Yiridoe & Weersink, 1998). Since farm heterogeneity plays an important role in abatement cost variations, a farm level approach is particularly useful in determining optimal environmental policy instruments. Aggregate approaches, which rely on national or regional aggregated models, tend to underestimate this variability (De Cara, Houze, & Jayet, 2005). Optimal environmental policy concentrates its efforts on those farms where abatement costs are low (Macho-Stadler & Perez-Castrillo, 2006); thus farm centric knowledge of abatement costs will facilitate selection of the most efficient option for nitrogen management. De Cara, Houze, & Jayet (2005) classified heterogeneity into the following three categories; activity related, emission-factor, and flexibility of substitution of production activities. In this chapter heterogeneity of farming systems is represented by production structure, soil and topography, which are important variables in determining the production and pollution potential of farms.

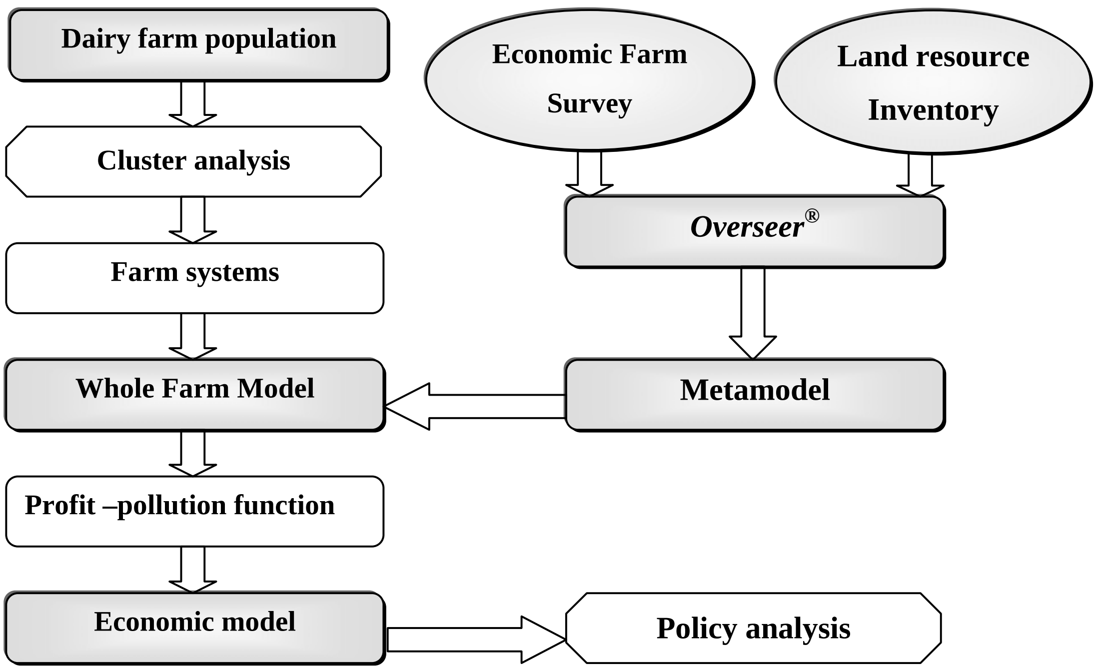
There is a complex relationship between changes in farming practice and results in terms of nitrogen discharge reduction. Moreover, results are observable only over a long time frame, and farmers lack the necessary knowledge to determine the changes that must be achieved. In this situation simulation models can be effectively used to evaluate changes in farming practices under different policies. Models for assessment of new policies need to incorporate both environmental and economic effects. Such models can help policy makers assess the trade-offs between economic and environmental objectives and thereby facilitate informed policy choices. Modeling dairy farms is a challenge because of the complex interaction of cows, management, paddocks and climate. For this reason, use has been made of the Whole Farm Model has been used to simulate policies (Cabrera, Breuer, Hildebrand, & Letson, 2005).

This study extends the value of the DairyNZ Whole Farm Model (WFM)(Beukes et al., 2005) by building and integrating a nitrogen discharge function using a meta modelling technique. Then a hybrid model was created by merging the merits of differential evolution and mathematical programming systems to overcome the inherent disadvantages of individual approaches.

The modeling framework developed in this analysis includes three components: a meta model, the WFM and mathematical programming. Firstly meta models for nitrogen discharges for specific soil types and topography were estimated using the simulated nitrogen discharges of the *Overseer* nutrient budget model. Second, the Meta models were incorporated into the WFM. Then WFM was calibrated to the farm systems derived from cluster analysis of catchment farms. Profits and respective nitrogen discharges obtained from the differential evolution-based optimization process of WFM were assembled to form a profit-pollution frontier. This frontier was subject to constrainted optimisation based on a non linear mathematical programming model, which predicts producer responses to alternative pollution control policies. Figure 5.1 gives an overview of the modeling framework. The model components are discussed in detail later.

The main focus of this study is to simulate responses to environmental policies aimed at intensive margin changes in different farming systems. A range of different farm types is represented in order to allow for biophysical and economic variations.

Figure 5.1 Overview of the modelling framework



5.2 Model setup

An ex-ante evaluation of the cost effectiveness of the measures requires an understanding of the abatement cost function of the farms involved. Abatement costs associated with the different farming systems were derived by gradually tightening the constraint on nitrogen discharge. It is assumed that farmers are perfectly competitive and risk neutral. Dairy farm profit per unit area is assumed to be a function of nitrogen discharges as a result of chosen input use intensities. It is represented by the profit-pollution function $f(x, \theta)$ where x is nitrogen discharge. θ is geo-phyiscal factors of the farm. The function is assumed to be concave, with a negative second derivative at all points ($f`>0, f``<0$). Farm land differs in the environmental consequences of production. Let the private per hectare optimum of the profit maximizing farmer with farm type θ be as follows (Equation 1).

$$Max \ f(x,\theta) = \beta_1 x + \beta_2 x^2 \tag{1}$$

Subject to $x \leq x^R$

x^R is the level of restriction of nitrogen discharge.

Given this production and pollution relationship, the management problem for the individual farm can be summarized by the following Lagrangian (Equation 2)

$$\varsigma(x,\lambda) = \beta_1 x + \beta_2 x^2 - \lambda(x - x^R) \tag{2}$$

The Lagrange multiplier λ indicates the rate of change of the maximum value of the objective function with respect to the parametric changes in the value of constraint. In other words the value of an extra nitrogen discharge to the producer. The Kuhn-Tucker conditions for this simple problem with constant returns production technology show that, in the absence of environmental policies, the producer will tend to apply inputs in order to maximize returns and this will result in higher levels of nitrogen discharge into water.

Three different policy scenarios are simulated. In the first scenario farm systems are subject to nitrogen discharge restrictions. In the second, a tax on nitrogen discharges is applied. In the third scenario a joint policy instrument is evaluated, in which a tax is imposed on nitrogen discharge that exceeds the amount allowed under nitrogen discharge restrictions. For the purposes of this study it is assumed that the size of the farming operation would not change as a result of the tax regime, although in practice this could happen. The following stylized forms illustrate the three policy scenarios.

Scenario 1 $Max\ f(x)$ subject to $x \le x^R$

Scenario 2 $Max\ f(x) - \tau(x)$

Scenario 3 $Max\ f(x) - \tau(x - x^R)$ subject to $x \le x^R$

5.3 Study area and data

The catchment in this study covers part of the Waikato River, including Lakes Arapuni and Karapiro. Analysis here is restricted to dairy farming which is the predominant agricultural land use in the catchment, occupying 72 percent of total agricultural land. There are 370 dairy farms, covering 52,877 ha. Assessment of nonpoint pollution abatement costs at farm level poses a challenge due to data limitations (Helin, Laukkanen, & Koikkalainen, 2006). Farm data therefore has been obtained from a virtual population of catchment farms derived in the chapter 4.

Each farm has its own specific soil and topographic characteristics and production structure which require different solutions, so modeling at the individual farm level is computationally demanding. In addition to this, implementation of perfectly differentiated emission policies at the individual farm level imposes an informational burden and implementation difficulties on the social planner. Unless this problem can be overcome, it is useful to bring farms with similar characteristics together into homogeneous groups, so that a series of common recommendations can be made. Grouping farms using statistical clustering techniques is a recommended practice (Hazell & Norton, 1986) for obtaining a reasonable representation of a range of farm production structures and bio-physical variables. Three distinctive farming systems in terms of production structure, soil and topography have been identified, based on latent class cluster analysis (Vermunt & Magidson, 2005). The farming systems are classified as intensive, moderate and extensive based on input use intensity and production. Table 5.1 describes the key variables of each farming system. The intensive farming system applies higher levels of nitrogen fertiliser to produce more grass and uses a greater amount of brought in feed to feed a higher number of better performing cows in order to produce more milk out of per unit of land area. In the extensive farming system a lower number of animals are kept per unit of land area and those animals are mostly fed with a limited amount of farm grown pasture and a lesser amount of brought in feed. The moderate farming system stands between these two extremes. The level of intensification is affected by geo-physical features of the farm, level of management skills, capital availability, risk preferences and life style choices.

The effect of relative distances of farms to the main stem of the river has not been taken into consideration in this study, as nitrogen is considered a uniformly mixed assimilative pollutant i.e the damage caused by the pollutant depends only on the amount discharged into the medium and is relatively insensitive to where the emissions enter the medium, or how long it takes to reach the medium (Tietenberg, 2006b).

Table 5.1 Descriptive statistics of the farming systems

Variables	**Moderate**	**Extensive**	**Intensive**
Farm size (ha)	106	136	92
Stocking rate(Milking cows/ha)	2.7	2.2	3.3
Annual Pasture requirement (kg/ha)	12975	9756	13950
Brought in feed (tons per cow)	0.5	0.6	2.12
Milksolids (Kg/ha)	919	788	1910
Soil type	Volcanic	Pumice	Volcanic
Topography	Easy	Rolling	Easy

5.4 Meta model for nitrogen discharge

The diffuse nature of agricultural nitrogen pollution and the time lag before it appears in a water body necessitates the use of a simulation model to derive quantifiable measures of the fate of nitrogen discharges in water. *Overseer*® is an nutrient budget model for decision support (Wheeler, Ledgard, DeKlein, Monaghan, & Carey, 2003). It simulates annual farm nitrogen discharges into water and has already been calibrated for New Zealand farming systems (Thomas, Ledgard, & Francis, 2005). Information on nutrient leaching derived from recent work on the central North Island lake catchments of New Zealand has been incorporated into *Overseer* (Clark, 2007).

Direct use of *Overseer*, however, is inefficient and impractical when a large number of unique combinations of farms are simulated in the optimisation process. Therefore a simplified approach called Metamodelling has been adopted here. A Metamodel is a statistical response function that approximates the outcomes of complex simulation models (Wu & Babcock, 1999). Metamodelling has been widely used to create nutrient discharge functions for policy analysis (Goetz, Schmidt, & Lehmann, 2006; Hefland & House, 1995; Martinez & Albiac, 2006). Metamodels were built by regressing the nitrogen discharges that resulted from simulating 100 randomly selected farms (taken from DairyNZ's Economic Farm Survey for Waikato in 2004/2005). Separate simulations were carried out for soil and topography combinations of the farming systems. In accordance with the guidelines for dairy farm effluent application (Dexcel, 2007b), it was assumed that 20 percent of each farm was subject to effluent application. The supplementary feed policy was kept on a par with the supplementary feed policy of the WFM.

5.4.1 Empirical specification for the Metamodel

In order to obtain useful predictions it is important to choose the right combination of inputs and the proper functional form for a Metamodel. According to the literature many functional forms are used to estimate Metamodels for nutrient discharge. For instance Martinez & Albiac (2006) built a quadratic Metamodel based on the Erosion and Productivity Impact Calculator (EPIC). Yiridoe & Weersink (1998) used a simple quadratic functional form in terms of nitrogen fertiliser use to predict nitrogen discharges based on CENTURY biophysical model simulations. Hefland & House (1995) used a square root function to fit a Metamodel of nitrate effluent from crop production using EPIC. Dake, Mackay, & Manderson (2005) fitted a full second order polynomial model in terms of stocking rate for dry stock farming using *Overseer*.

In cattle grazing systems nitrogen discharged from urine and dung patches, and applied fertiliser are the major sources of nitrogen (Ledgard & Menneer, 2005; Ledgard, Penno, & Sporsen, 1999)[1]. Monaghan *et al*,. (2007) show that even though nitrogen excreted by animals is the primary source of nitrogen leaching and run off, use of nitrogen fertiliser indirectly contributes to nitrogen leaching by boosting pasture production and therefore stocking rate. Stocking rate and nitrogen fertiliser application were selected therefore as variables for estimating the Metamodel. In reality there are of course other sources contributing to nitrogen discharge, including brought in feed resources (Ledgard & Thorrold, 2003). These were excluded from the Metamodel to avoid complexity, as feed resources are categorised into 7 major groups and many sub groups in WFM, resulting in 60 categories**.** Instead, feed resources are capped in the WFM simulations.

Various functional forms were fitted, including quadratic, exponential and square root functions. The exponential functional form (Equation 3) statistically out performed other functional forms, and this is consistent with the functional form fitted for experimentally measured nitrogen discharges against total nitrogen input by Ledgard and Menneer (2005).

$$\ln(Y) = \beta_0 + \beta_1 N + \beta_2 SR \tag{3}$$

Y = Nitrogen discharge KgN ha per year
N = Nitrogen applied to paddocks in kg ha per year
SR = Number of animals per ha.

The parameters of the nitrogen discharge function were estimated using ordinary least squares regression. Table 5.2 presents the parameters of estimated nitrogen discharge functions by soil and topographic categories. All coefficients are highly significant. Goodness of fit of the discharge function was very good, as indicated by the adjusted coefficients of determination. The assumption for homoscedastic errors could not be rejected on the basis of the Breusch-Pagan / Cook-Weisberg test for

[1] Experimental evidences are provided in Appendix 2.1 and 2.2

heteroskedasticity, since the chi-squared test statistic is above the critical value of 0.0654. Colinearity between variables is tested using the Variance Inflation Factor (VIF) (Baum, 2006). The rule of thumb states that there is evidence of colinearity if the largest VIF is greater than 10. We can rely here on the conditioning of the model as the maximum VIF (1.08) is far smaller than 10 for functions in both soil/topography classes. The empirical form of the Metamodel can be interpreted as a percentage change in the dependent variable as a result of unit change in the independent variable. For instance if β_2 is 0.26 then a 1 unit increase in SR will produce a 26 percent (approx.) increase in *E(Y).* The estimated discharge function is graphically presented in Figure 5.2. We can see from this, that the effect of stocking rate on nitrogen discharge is greater than that of nitrogen fertiliser. Also, both have a large combined effect.

Figure 5.2 Nitrogen discharge function

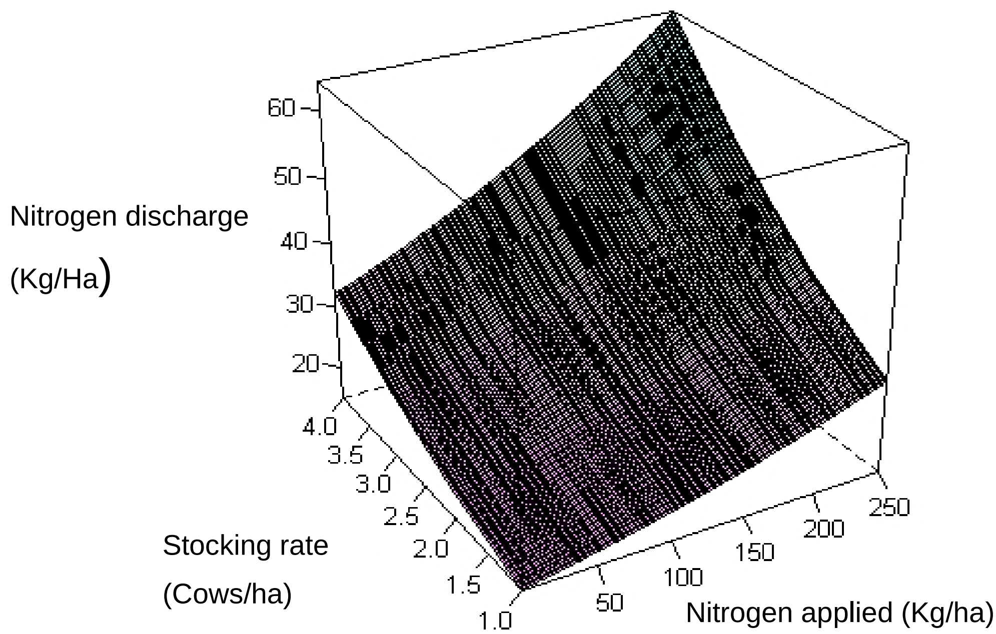


Table 5.2 Parameters of Metamodel

	Volcanic/ Easy			Pumice /Rolling		
	coefficient	t stat	P value	coefficient	t stat	P value
Intercept	2.42	66.21	0.000	2.52	90.72	0.000
Nitrogen	$2.81*10^{-2}$	34.29	0.000	$2.86*10^{-2}$	45.82	0.000
Stocking rate	0.26	20.71	0.000	0.27	27.56	0.000
Adj R^2		0.98			0.98	

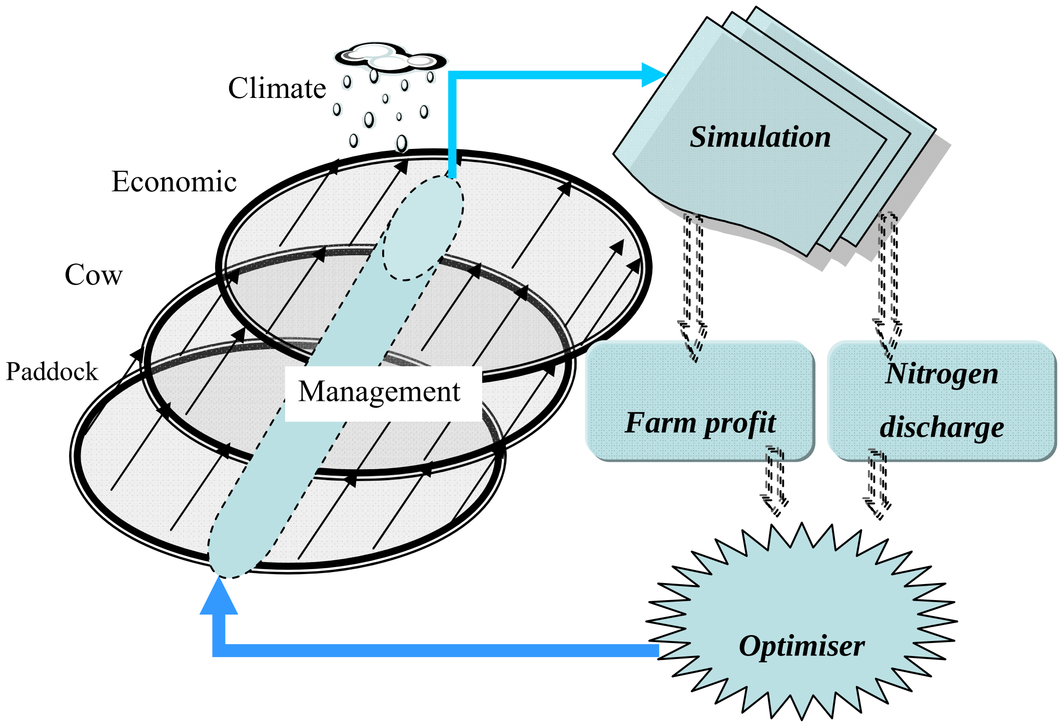
5.5 Dairy NZ's Whole Farm Model (WFM)

The Whole Farm Model (WFM), developed by Dairy NZ, was used to ascertain possible farm responses to environmental policies. The WFM is a computer model that simulates New Zealand dairy farms. The model framework is explicitly designed to facilitate incorporation of existing and future sub models by using an object oriented framework implemented in Visual Works Smalltalk language. WFM components consist of climate, management, cow, paddock and economics sub models (Beukes et al., 2005). These sub models simulate complex interactions of climate and pasture growth, cow metabolism and management regimes and resultant economic output. An overview of the WFM structure used for this study is illustrated in Figure 5.3.

The pasture model used is the McCall Pasture Model, based on the work of McCall & Bishop-Hurley (2003). The cow sub model used is Molly, a dynamic model consisting of differential equations describing the nutrient metabolism of cows under New Zealand conditions (Palliser, Bright, Macdonald, Penno, & Wastney, 2001). The economics component is similar to that specified in the Economic Farm Survey of New Zealand Dairy Farmers. Dairy farm operating profits are calculated using the economic module of the WFM. The revenue of the farm is primarily derived from the sale of milksolids. The details of the economic component can be found in Neal, Drynana *et al.*(2005). The WFM simulates optimum farm responses to different policies and calculates the dairy farm profit by executing differential evolution (DE)-based farm optimisation. The process of WFM optimisation is discussed in the following sections.

The WFM is calibrated to represent each farming system, with outputs and inputs being verified with data for individual systems. The feed policy of the model is fixed for each farming system. The pasture production potential of paddocks is calibrated using the Pslope (α) parameter in the pasture model (McCall & Bishop-Hurley, 2003). Pslope is used as a proxy for inherent soil fertility and represents the efficiency of solar radiation. Cow production potential in terms of milksolids production per ha per year under different systems is calibrated by manipulating the lactation period, live weight and PV milk. PV milk is a measure of a cow's genetic production potential and was developed by the Livestock Improvement Corporation (Beukes et al., 2006).

Figure 5.3 Overview of the Whole Farm Model analysis

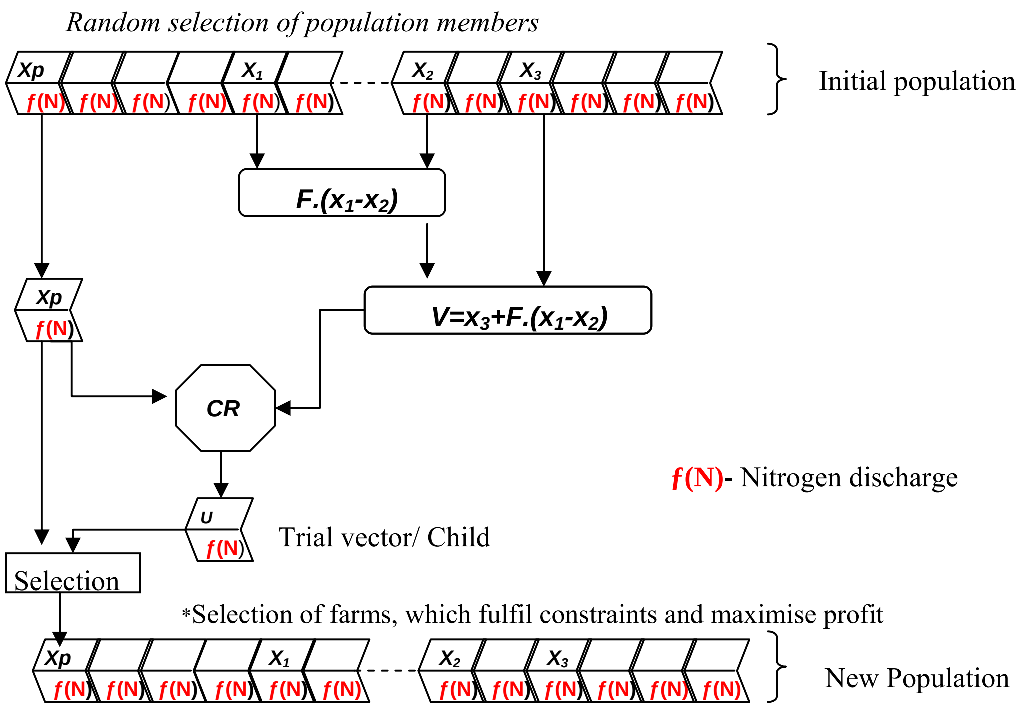


An understanding of nitrogen response is important if the production potential of farms is to be realised, as soil heterogeneity, climate variability and time of application can have major implications for nitrogen fertiliser use. The nitrogen fertiliser response of the pasture is modelled using a decision tree model for pasture growth developed by Zhang & Tillman (2007). This is an empirical model based on data collected from experimental trials in New Zealand. To estimate the nitrogen response, the model considers the time and amount of nitrogen applied, climate, topography, phosphorus fertility and land slope. In this application only the amount of nitrogen fertiliser applied varied between farming systems. Variability in inherent soil fertility is represented by the farm system specific Pslope (α) parameter as discussed previously**.**

5.5.1 Optimisation using Differential Evolution (DE)

Optimisation of the WFM has been performed with a specific evolutionary algorithm, known as differential evolution, a variant of a more common genetic algorithm developed by Storn & Price (1997). DE has been applied in a few agricultural bio-economic studies (Alfrod, Cacho, & Griffth, 2006; Mayer, Kinghorn, & Archer, 2005; Neal et al., 2005), and is implemented in the WFM in a similar way to Mayer, Kinghorn, & Archer. In the WFM the key feature of differential evolution is to generate a population of farms using features of biological evolution, such as reproduction, selection, mutation and recombination. The vector of individual farm characters is referred to as the genotype. Each parameter of the vector is described as an allele, and each allele in the genotype is represented by a real number. In the empirical application, alleles of the genotype are decision variables such as input use and management. A population of genotypes is randomly generated, with each genotype being characterised by its level of fitness, defined in terms objective function value dairy farm profit. Higher dairy farm profit indicates better fitness. Here the each genotype is simulated through the WFM to generate dairy farm profit. The process of differential evolution involves the iterative improvement of a set of solutions or genotypes based on a fitness function. The steps of differential evolution are illustrated in Figure 5.4. The details of the process based on Price, Storn, & Lampinen (2005) are described below.

Figure 5.4 Process of optimisation



Source: Adapted from Price, Storn, & Lampinen (2005)

Initialization

The population is initialized by specifying upper and lower boundaries on parameter values of interest. In empirical application, parameter values of interest are stocking rate (number of cows per ha) and fertiliser applied. Once initialization bounds have been specified, a random number generator assigns each parameter of every genotype a value from within the prescribed range (Equation 4).

$$x_{i,j,0} = rand_j(0,1).(b_{j,U} - b_{j,L}) + b_{j,L} \tag{4}$$

X_i –Genotype, $_{J}$- *a* parameter vector of interest

Once the initial population is established new population members are created by randomly selecting three members of the original population and subjecting them to a process of mutation and cross over. The 3 randomly chosen genotypes are labeled as x_1, x_2 and x_3.

Mutation

Mutated genotypes v_i are created by adding the scaled vector of differences between two genotypes with the third genotype. Equation 5 shows how to combine three different, randomly chosen genotypes to create a mutated genotype.

$$v_i = x_3 + F.(x_1 - x_2) \tag{5}$$

The scale vector F is a positive real number between 0 and 1 which controls the rate at which the population evolves. F can be specified by the user.

Cross over

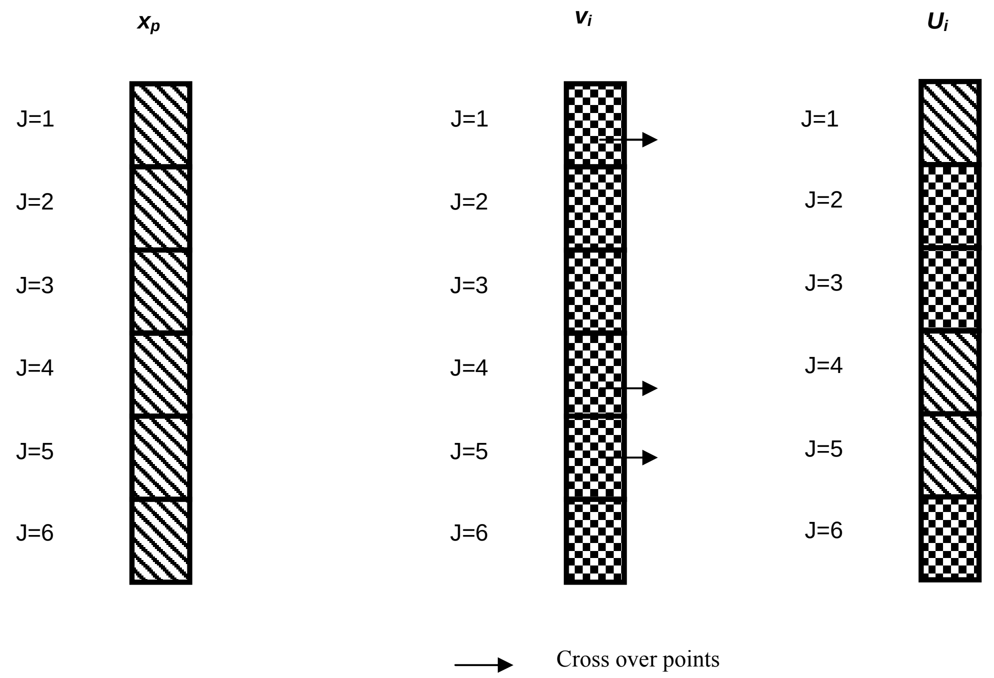
Crossing over of mutated genotype v_i with another member of the initial population x_p, known as the target genotype, creates a child genotype u. The cross over probability, CR, is a user defined value range $0 \leq CR \geq 1$ that controls the fraction of parameter values that are copied from the mutated genotype. To determine which source contributes a given parameter, CR is compared with the output of a random number generator, $rand_j\,(0,1)$. If the random number is less than or equal to CR, the child parameter is inherited from the mutant, v, otherwise the parameter is copied from the vector of genotype x_p. In addition to this, another operation of recombination is carried out by randomly selecting a parameter from the mutant vector into the child vector. This is to ensure that the child vector does not duplicate the parents. The process of crossing over is illustrated in Figure 5.5.

$$u_i = u_{ji} \leftarrow v_{ji} \text{ if } (\text{rand}_j\,(0,1) \leq CR \text{ or } j = j_{rand}) \tag{6}$$

Otherwise

$$u_i = u_{ji} \leftarrow x_{ji}$$

Figure 5.5 Crossing over



Selection

Selection is based on the value of the objective function known as fitness. In case of maximization problems, the child vector (*u*) replaces the target vector x_p, provided that the child vector has an equal or higher objective value than that of its target vector. Otherwise the target vector retains its place at least for one more generation. The above mentioned procedures are repeated until the optimum is located or until pre-specified termination criteria (number of generations) are satisfied.

Constraints

Constraints are introduced to restrict the nitrogen discharge per ha from farms. It require a solution to contain parameter values that satisfy the constraint. Most often, constraints are implemented as penalty functions (Price, Storn, & Lampinen, 2005) which decrease the objective function value when constraints are violated. Constraint violation is incorporated into the optimisation task by deducting the penalties for the constraint *p(x)* from the objective function.

$$f'(x) = f(x) - p(x) \tag{7}$$

The penalty is integrated into the objective value (Equation 7); individuals performing well will have a higher fitness and thus a greater chance of survival. The penalty is specified by attaching price tags to estimated nitrogen discharges. This penalty is then deducted from the objective function. The trade-offs between nitrogen discharges and economic farm surplus are derived from the results of optimisation, which yields a large range of alternatives. In this approach multiple constraints can be incorporated by introducing weights. In our previous work (Ramilan, Scrimgeour, Levy, & Romera, 2007) tax scenarios were implemented using penalty functions, but these have a number of draw backs (Price, Storn, & Lampinen, 2005): they require specification of appropriate weights to constraints in the presence of multiple constraint handling; inappropriate specification of weights leads to convergence in infeasible regions; there are issues around premature convergence and local minima. An additional problem associated with using the penalty function approach is the excessive time required to generate solutions. These drawbacks can all be avoided by adopting direct constraint handling.

Direct constraint handling

In direct constraint handling, each population vector or genotype is assigned a nitrogen discharge, estimated using the Metamodel. Trial vectors which fulfil the constraints are selected. If the objective function is unconstrained then the objective function values are simply compared in the same way as the differential evolution selection criteria. When neither vector is feasible the objective function values are not compared. Selection drives vectors in the direction of constraint violation decrease. Optimum farm responses for different nitrogen discharge levels are tailored by gradually tightening the constraints on nitrogen discharge.

5.6 Empirical analysis

5.6.1 Profit pollution frontier

In the WFM simulation, lower and upper bounds on the parameters of the decision variables (stocking rate and nitrogen application) are specified to initiate populations of farms or genotypes. Maximising Dairy farm profit is set as an optimisation target. The mutation rate is set at 0.4 and cross over rate is set at 0.5. These rates have been used in other differential evolution applications (Alfrod, Cacho, & Griffth, 2006; Neal, 2004). The initial population was set at 25 genotypes which were simulated through 40 generations, creating 1000 genotypes of various combinations of parameter vectors in each scenario. This process resulted in a matrix of thousands of farm activities in each farming system. The results are summarized in Table 5.3, which shows the maximum farm profit achieved at different levels of nitrogen discharge constraint.

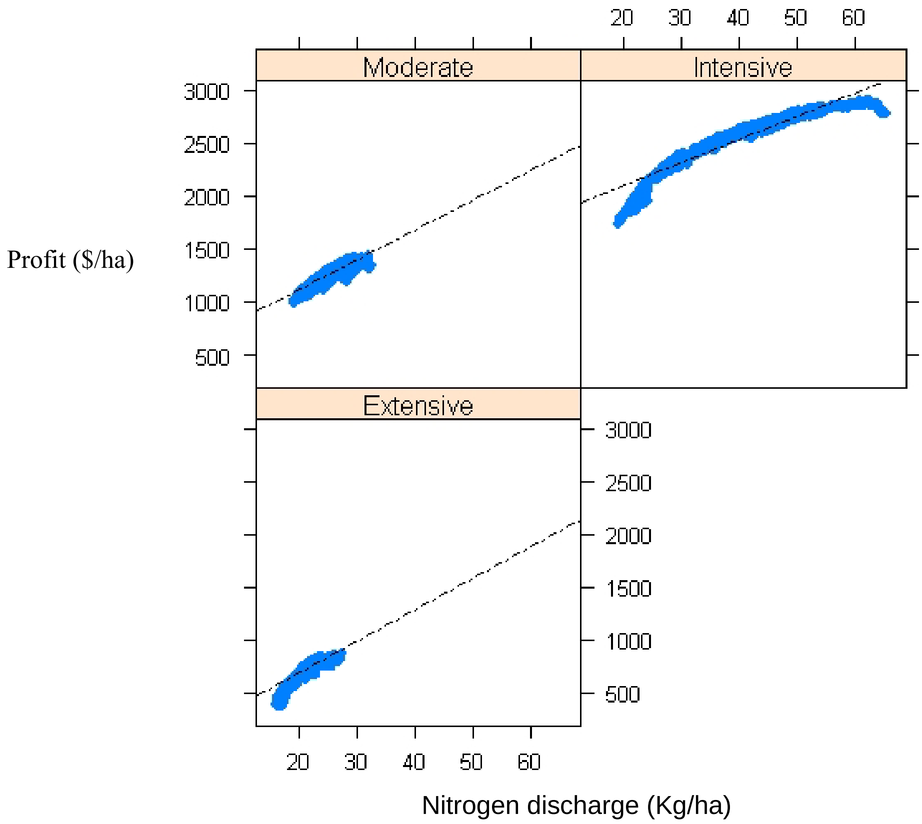
Optimisation through the DE algorithm took approximately 40 hours for scenarios in each farming system, using a Pentium IV computer with 3.2 GHz speed and 2048 MB RAM. Thus three farming system scenarios consumed a total of 120 hours of computation time. This high computational time is a problem in using DE, when it is needed to simulate many scenarios for different input and output prices and nitrogen discharge levels. In contrast, production function oriented activity analysis based on mathematical programming optimisation can produce results in a fraction of the time. However, the accuracy of the results from this approach depends on the functional form of the complicated multivariate equation and on the availability of accurate data. When policies are evaluated ex ante, the data on farm responses are not available. A further complication is that mathematical programming tends to find the local optimum rather than the global optimum. Compared with this, WFM's differential evolution approach does not require any complicated functional forms and is capable of evolving scenarios under varying constraints, but in activity analysis the optimum is selected from among the pre existing activities. For these reasons this study uses a simple and innovative hybrid solution that combines the merits of both DE and mathematical programming.

Table 5.3 WFM optimisation results

Economic Farm Surplus ($/ha)	Stocking Rate	Pasture Production (Kg/ha)	Milksolids Production (Kg MS/Ha)	Nitrogen Applied (Kg /ha)	Nitrogen Discharged (Kg/ha) Per Ha
			Moderate		
1429	2.7	12592	909	120	32
1418	2.4	12506	846	117	30
1406	2.3	12577	814	118	29
1348	2.1	12535	756	116	27
1315	1.9	12360	705	112	25
1250	1.9	12068	695	90	24
1197	1.7	12006	644	81	22
			Intensive		
2905	3.5	14993	1641	270	61
2843	3.2	14896	1512	258	53
2779	3.1	14644	1488	225	48
2671	2.8	14525	1350	210	42
2554	2.4	14380	1175	193	36
2409	2.5	13810	1194	121	30
2146	2.1	13167	1008	81	24
			Extensive		
880	2.3	9040	714	60	27
850	2.1	9131	671	60	26
837	1.7	9130	571	60	24
826	1.6	9184	553	60	23
796	1.5	9148	497	58	22
724	1.4	8823	476	42	21
653	1.2	8860	417	34	19

In the hybrid method, the matrix (results) generated in the DE optimisation for each farming system was used to build a profit pollution frontier. The profit pollution frontier for each farming system is illustrated in Figure 5.6 using R. R is an open source language and environment for statistical computing and graphics (R Development Core Team, 2007).

Figure 5.6 Profit-pollution frontiers



The profit pollution frontier portrays the relationship between nitrogen discharges and level of farm profit given constant technology. It is driven through subsequent reduction in stocking rate and nitrogen fertiliser application. This frontier was used to build econometrically specified non linear production functions for farming systems in terms of farm profit and nitrogen discharges. Production function have been specified by many functional forms, none of which is unanimously considered to be superior to the others (Goetz, Schmidt, & Lehmann, 2006). The polynomial formulations, in particular quadratic specifications, are generally used to specify agricultural production functions because of their suitable properties and ease of estimation. Quadratic forms are the most commonly used functional forms in empirical estimation of production and pollution functions (Wu & Babcock, 2001) and used in many studies to model yield responses to changes in nitrogen fertiliser use (Brady, 2003; Yiridoe & Weersink, 1998). Therefore the production function has been specified here as a quadratic function (Equation 8). A quadratic function tends to produce a concave surface similar to the visualised graphical patterns in Figure 5.6. The parameters of the functions are estimated using least squares regression. Estimated parameters are presented in Table 5.4. All the coefficients are highly significant with p values at 0.000. π is dairy farm operating profit and q nitrogen discharge per ha.

$$\pi(q) = \beta_0 + \beta_1 q + \beta_2 q^2 \tag{8}$$

Table 5.4 Parameter estimates of production function

Farming system	Moderate	Intensive	Extensive
Constant	-906.36 (32.37)	835.92 (154.96)	-3008.08 (82.35)
β_1	140.91 (65.14)	65.55 (237.2)	310.58 (94.15)
B_2	-2.16 (-52.26)	-0.54 (160.12)	-6.30(85.28)
Adj R^2	0.89	0.98	0.83

t statistics are given in the parenthesis

The profit- pollution frontier can be a useful tool from a policy implementation perspective as it establishes the relationship between production and pollution The implementation of environmental policies like emission charges is likely to meet with political resistance unless a clear relationship is established between the estimated nitrogen discharge and some other visible variable. In the Netherlands, for instance, surplus phosphorus from manure is charged in conjunction with individual farm quotas on livestock numbers. The indirect estimates of phosphorus surplus are based on the nutrient accounting system MINAS, introduced by the Dutch government in 1998. It taxes every farmer individually, based on nutrient surpluses of each farm (Ondersteijn, Beldman, Daatselaar, Giesen, & Huirne, 2002). Citing the Netherlands experience on mineral accounting, Weersink, Livernois, Shogren, & Shortle (1998) show that establishing a direct relationship between observable variables and estimated pollution discharge is imperative for the success of policies like emission charges. Further empirical simulations by Peterson & Boisvert (2004) suggest that monitoring corn yields could be substituted for potentially costly and intrusive monitoring of fertiliser use. Overall, a production function in terms of nitrogen emission for different farming systems can provide the information necessary for implementing environmental policies.

5.6.2 Trade-off analysis

The Profit-pollution function is subject to non linear optimisation under various policies to evaluate the trade off between farm profit and nitrogen discharges. Abatement costs associated with the various farming systems were derived through parametrically varying the nitrogen discharge constraint and repeatedly solving the optimisation problem. The optimstion model was written using the General Algebraic Modelling System (GAMS) (McCarl et al., 2007) and solved using the MINOS solver (GAMS/MINOS, 2001), recommended for problems with non linear objective function and linear constraints (McCarl & Spreen, 2004). The model was first solved without the nitrogen discharge constraint as a baseline. The baseline discharge level was then gradually decreased by 2 to 60 percent resulted in 30 discharge levels for each farming system. Abatement cost is the shadow price for water quality improvement. The abatement costs for different abatement levels are illustrated in Figure 5.7. The cost of abatement is much lower in the intensive farming systems. The abatement cost function , which shows the rise in abatement cost by increasing the discharge constraint by one unit, has been interpolated on resulting points on the abatement cost generated by the non linear optimisation, using the ordinary least squares regression (Figure 5.8). The functional form used is $C_i(A_N) = \beta A_N^2$. C_i is the cost of abatement and A_N is the nitrogen discharge. Estimated β coefficients for intensive, moderate and extensive farming systems are 0.54, 2.16, and 6.30 respectively. The abatement costs rise at an increasing rate with higher abatement levels. The curves are convex and positively sloping with the level of discharge reduction. It is comparatively cheap to abate initial amounts of discharge, but additional reductions require expensive forms of abatement. Reducing nitrogen discharges on intensive farms seems to be relatively cheaper than on extensive or moderate farms. Average cost of abatement for initial 10 kg of nitrogen is illustrated in Figure 5.9.

Figure 5.7 Abatement costs from optimisation

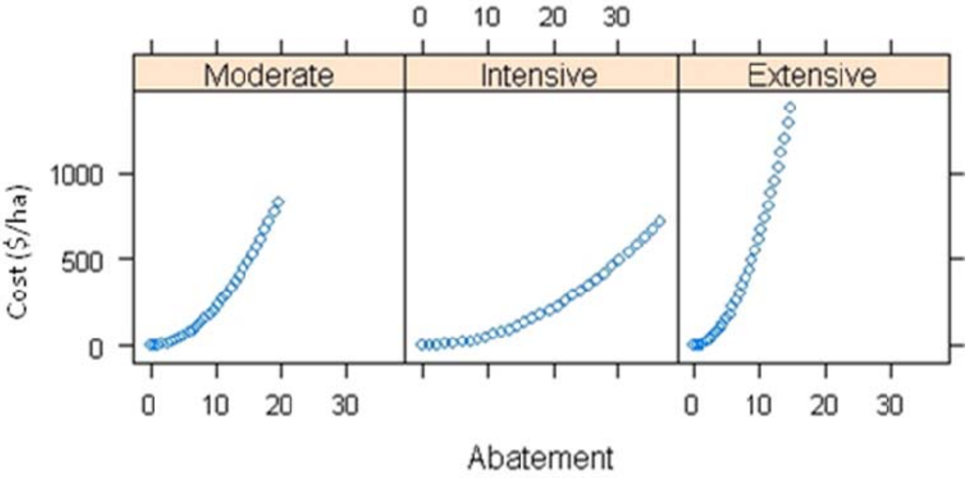


Figure 5.8 Interpolated abatement costs

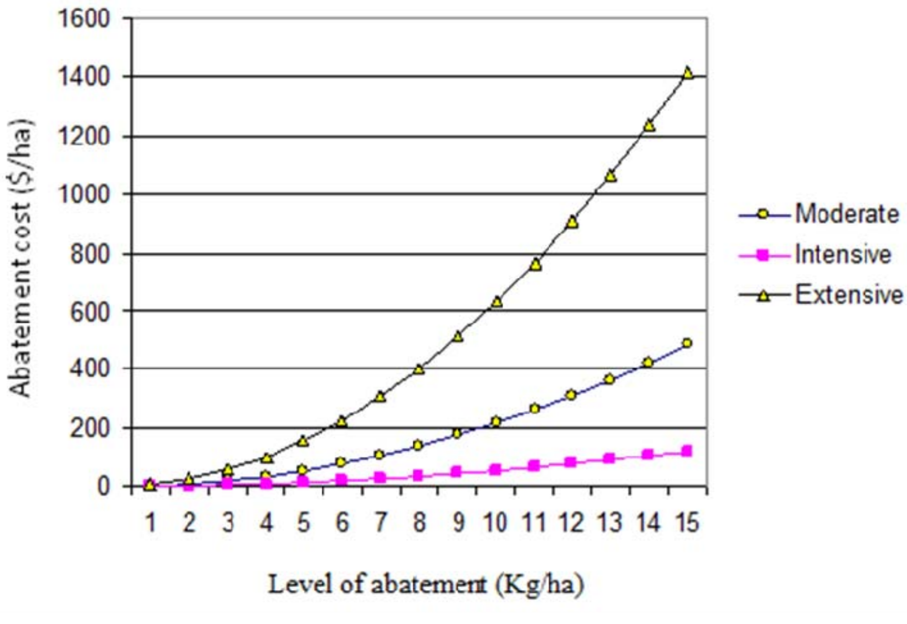


Figure 5.9 Average cost of abatement

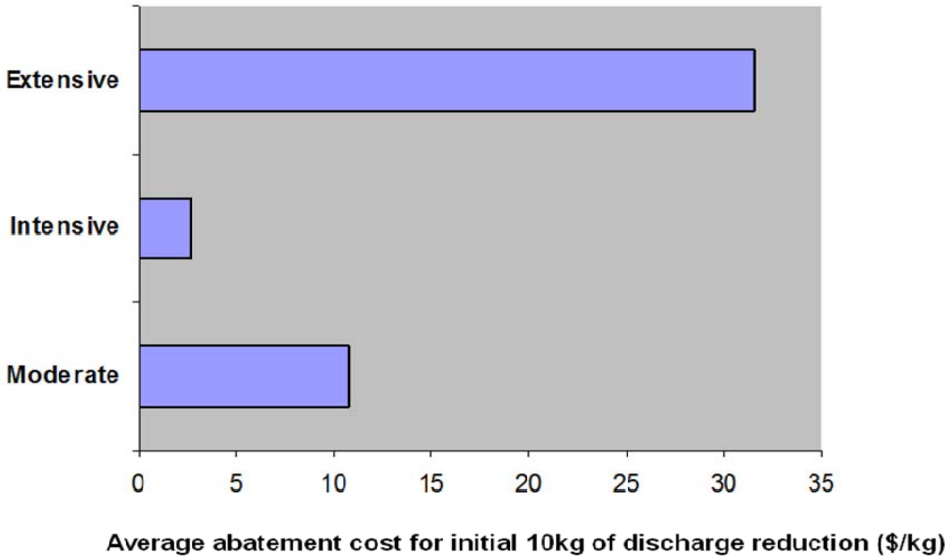
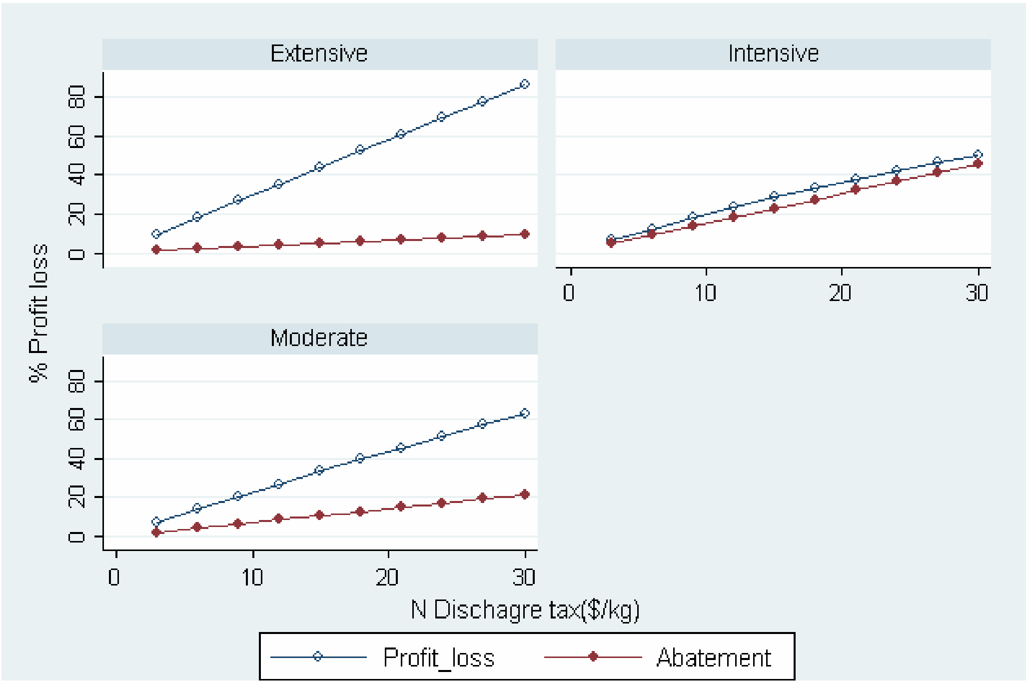


Figure 5.10 Economic impact of nitrogen discharge tax



Restricting nitrogen discharges may face strong political resistance, since standards are likely to be viewed as an infringement of the property rights of the farmers involved. Therefore, as an alternative, a tax regime on nitrogen discharges is simulated across the farming systems in the second scenario. In this scenario a unit of nitrogen discharge is taxed and deducted from the farm profit in the optimisation process.

Table 5.5 Cost of reducing nitrogen discharge in the taxation scenario

	Base	**$3**	**$6**	**$9**	**$12**
Moderate					
Profit	1391.75	1294.93	1200.20	1107.56	1017.00
Nitrogen discharge	32.62	31.92	31.23	30.54	29.84
Abatement cost($)		96.81	191.54	284.19	374.75
Abatement(kg N)		0.69	1.39	2.08	2.78
Abatement Cost($)-Standard		1.03	4.17	9.35	16.69
Intensive					
Profit	2825.18	2647.26	2477.68	2316.43	2163.51
Nitrogen discharge	60.69	57.92	55.14	52.36	49.58
Abatement cost($)		177.92	347.50	508.75	661.67
Abatement(kg N)		2.78	5.56	8.33	11.11
Abatement Cost($)-Standard		4.17	16.69	37.47	66.66
Extensive					
Profit	819.70	746.11	673.23	601.07	529.62
Nitrogen discharge	24.65	24.41	24.17	23.94	23.70
Abatement cost($)		73.59	146.47	218.63	290.08
Abatement(kg N)		0.24	0.48	0.71	0.95
Abatement Cost($)-Standard		0.36	1.45	3.18	5.69

The results shown in Table 5.5 indicate that the impact of farm incomes under varying taxes is much higher than under nitrogen restriction standards. The cost of a discharge tax to an intensive farm, for instance, is estimated to be $178, compared with just $4 for the same amount of reduction using a nitrogen restriction approach. A standard for nitrogen discharges results in more significant pollution reduction at much lower costs to farms. These results are consistent with the findings of other studies (Giraldez & Fox, 1995; Martinez & Albiac, 2006; Wu, Teague, Mapp, & Bernado, 1995). Relatively high tax rates would be required to induce farms to substantially reduce nitrogen discharge. The optimal level of tax increases with the marginal pollution costs of nitrogen discharge (Wu & Babcock, 2001). Extensive farming systems with higher cost of abatement would therefore require high levels of tax, but a higher level of tax is likely to threaten the economic viability of a farming operation. Doubts have also been cast over the political feasibility of introducing higher taxes (Hefland & House (1995). However, despite the risk, the social planner may wish to apply a tax on nitrogen discharges as it overcomes the political problems associated with setting limits on the discharges. Income losses could be lessened if mechanisms are devised to pay back the tax remittance to the farm in some other form. The comparative performance of tax and standards depends on relative slopes and correlations between marginal damage and abatement costs (Wu & Babcock, 2001). This study has not considered the damage costs of nitrogen discharges and transaction costs involved in policy implementation. This is an area that needs further exploration. Figure 5.10 shows that taxes are effective in reducing nitrogen discharges only on intensive farming systems. Thus farms with lower abatement costs are more sensitive to taxes.

In the third scenario a joint policy instrument is evaluated. A tax is charged on any nitrogen discharge surpluses over the nitrogen discharge restrictions and deducted from the farm profit in the optimisation process and the level of tax required to bring down the nitrogen discharges by 20% from the status quo is determined iteratively. The results demonstrate that the joint policy instrument is more cost efficient than the solo emission charges even though rate of taxes are higher (Table 5.6). For instance in a moderate farm system a 20 percent reduction of nitrogen discharge resulted in an approx. 7 percent loss in farm profit, when compared to 57 percent profit loss under scenario 2.

Higher abatement costs of extensive farming systems can be attributed to their already low levels of discharge, and efficient policy schemes for the reduction of nitrogen discharge should differentiate between farm types rather than using uniform measures. The effectiveness of targeted tax policies on farm nutrient discharge management is noted in other studies as well (Hopkins, Schnitkey, & Tweeten, 1996; Mapp, Bernado, Sabbagh, Geleta, & Watkins, 1994; Zilberman, Khanna, & Lipper, 1997). Targeting farms with high levels of nitrate emissions within the catchment has a greater potential for cost effective reduction of nitrogen discharges. However there is a trade-off between implementation difficulties associated with differentiated policies and the cost effectiveness achieved. This trade-off needs to be taken into consideration when designing policies.

Table 5.6 Results of joint policy instrument

	Moderate	Intensive	Extensive	
Tax	\$28	\$13	\$62	\$31
Profit (\$/ha)	1299.70	2745.67	666.62	780.05
Nitrogen discharge(Kg/ha)	26.09	48.66	19.73	22.19
Abatement (kg)	6.53	12.04	4.92	2.46
Cost of abatement (\$)	92.05	79.51	153.07	39.65
Reduction in profit (%)	6.61	2.81	18.67	4.84
Reduction in nitrogen discharge (%)	20.01	19.83	19.96	9.98

As indicated in the two joint policy scenarios for extensive farms, the social planner could design contracts of different types such as higher tax and lower percentage reduction of discharge or vice versa. This would have the effect of making farms self select a suitable policy, so intensive farms, for instance, would be likely to choose low tax and high abatement targets.

The differences in abatement costs between farming systems are significant. This can be attributed to variations such as input use intensity, and to geophysical factors such as differences in soil type and topography which influence nitrogen discharge. Abatement cost differences are likely to act as a spur for the trading of nitrogen discharge permits (Kampas & White, 2003), with low abatement cost farms opting to abate more and sell some of their permits, while high abatement cost farms may prefer to buy more permits and maintain their emission levels. Abatement costs help to readily identify the net buyers and net sellers, thus potentially reducing the transaction cost of environmental policy implementation.

Where taxing or restricting emissions is estimated on stocking rate and nitrogen fertiliser application there is a tendency to reduce use of those inputs but encourage the use of substitutes. For instance reduction of nitrogen fertiliser tends to boost the use of feed that is brought in. However, since stocking rate and feed brought in are likely to be highly correlated, a policy targeting the stocking rate tends to overcome this input substitution problem (nitrogen discharge coefficients of stocking rate are much higher than that of nitrogen fertiliser, see Figure 5.3).

Since the cost of abatement varies, a uniform policy has the effect of encouraging less efficient farms with higher marginal abatement costs to shut down or shift to alternative land uses, while intensive farms are forced to reduce their level of activity. Land use conversions in extensive farms could possibly be a good option for reducing problem nitrogen discharges from the social planner's perspective, but evaluating the effectiveness of policies on changes at the extensive margin is beyond the scope of this study. Nevertheless it is important to note that designing optimal input policies that affect only the intensive margin should not be done in isolation. Regulating the intensive margin has to be complemented by regulation of the extensive margin also (Goetz, Schmidt, & Lehmann, 2006).

5.7 Conclusion and implications for future research

This chapter has developed and applied an analytical framework for evaluating different water quality policies, in the presence of spatial production and pollution heterogeneity. The framework integrates the WFM, Metamodelling and mathematical programming. The WFM is used to simulate farm responses under different nitrogen discharge restrictions to build profit –pollution frontiers, which are used in a mathematical programming model. It is used to analyse the responses of different farming systems to alternative policies. The Metamodel for nitrogen discharge built and integrated with the WFM allowed the greatest possible flexibility in evaluating the effects of different policies, as it was not necessary to coordinate exogenously the nitrogen discharges from simulated farms. The model saved resources and time.

Information about abatement cost heterogeneity between catchment farm systems is valuable to implement environmental policies as it helps to identify low cost farms. A key factor affecting relative aggregate costs under alternative policy instruments is farm heterogeneity. It can be attributed to differences in production systems, soil type and topography. Targeted policies for taxing or restricting nitrogen discharges on certain production systems may be cost effective.

Compensating farms which do undertake measures to abate nitrogen discharges is supported by an increasing number of both farmers and environmentalists. According to section 3.9 of the Waikato Regional Plan, Environment Waikato would consider providing financial support to projects that contribute significantly to minimising the impacts of land use activities on water bodies (Environment Waikato, 2007c). The abatement cost information generated by the modelling framework would be useful in determining an appropriate level of compensation. The model is particularly relevant for cost sharing initiatives between farms and environmental agencies as it enables an agency to base any incentive packages to farms on accurate information about the abatement costs associated with a specific farming system (Yiridoe & Weersink, 1998).

The present study could be extended in a number of ways, but assessing the impact of price variations would be the highest priority, particularly in view of the recent upsurge in world dairy product prices. In the empirical analysis it is assumed that farmers can respond to nitrogen discharge restrictions or taxes only by changing the level of activity at intensive margins. However farmers do have several other options and environmental policies may provide incentives for farmers to adopt best management practices for reducing nitrogen discharges or land use changes. It may be of interest to extend this framework by incorporating best management practices.

The analysis is exploratory rather than a comprehensive assessment of a particular policy. The range of tax policies and standards chosen here are indicative of the range of instruments that can be evaluated in this way. This study considers general policy implications across a range of farming systems, but the results should be considered as preliminary. Given the heterogeneity of farms in the catchment a comprehensive policy analysis would require further work. The model proposed can be further refined by accommodating additional farm types. Abatement cost information derived from profit pollution frontier in this chapter is useful to design and implement effective policies. The usefulness of profit pollution frontier generated in policy implementation has been demonstrated in chapter 6.

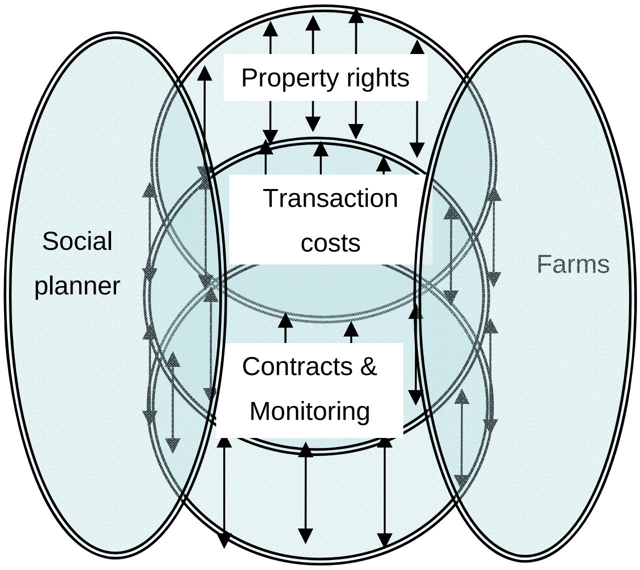
6. Challenges of environmental policy implementation

6.1 Introduction

Managing nonpoint pollution is a complex issue requiring appropriate institutions, sufficient information and incentives as it involves numerous agents (farms). To date, pastoral nonpoint sources have largely been free from regulation in New Zealand, however there is now increasing community pressure for better water quality (Brodnax, 2006). Water quality concerns have triggered scientific research and as a result, basic data is now available for environmental management and the development of best management practices. Best management practices convey the property rights in a specific manner (Stephenson, Norris, & Shabman, 1998). Bio-economic modelling is already playing a role in the design and development of environmental policies (Ramilan & Scrimgeour, 2006). However, barriers to implementation, high enforcement costs and imperfect compliance due to inappropriate institutions and imperfect information are likely to reduce the effectiveness of policies. Imperfect information affects compliance monitoring. Lack of compliance monitoring and enforcement are said to be causes of deteriorating water quality (Cullen, Hughey, & Kerr, 2006). Therefore it is important to reconsider existing institutions. Institutions in the environmental policy context mainly refer to the establishment and enforcement of property rights.

In New Zealand local authorities are empowered through the Resource Management Act (RMA) to monitor resource consents and complaints about environmental issues. Local authorities are generally referred to as the social planner in the literature - see Westra, Easter, & Olson (2002). This chapter views environmental policy implementation as a complex interaction between the evolution of property rights, transaction costs, monitoring and contract design, involving the social planner and farms (Figure 6.1). For instance in the presence of well defined property rights and a few numbers of farms it is easier to design and implement policies like contracts, thus incurring lower level of monitoring and transaction costs. It draws contributions from existing theoretical and empirical literature on economics and environmental law to analyse institutions, design of contracts, monitoring and transaction costs relevant to environmental policy implementation in the Karapiro catchment. Accompanying empirical analyses show how transaction costs and problems associated with imperfect information can be minimised.

Figure 6.1 Interaction of rights, transaction costs and contracts



Section 2 of this chapter analyses environmental property rights and their present status in New Zealand. Section 3 discusses transaction costs. Section 4 analyses the implications of information asymmetry, monitoring and contract design. Section 5 outlines the way in which the conceptual framework is adapted to encompass non compliance and monitoring in policy implementation. Section 6 provides an empirical analysis, mainly using the results of the simulations performed in the previous chapter. Section 7 consists of discussion based on the analysis.

6.2 Property rights and its challenges

Demetz (1967) defined property rights as the capacity to use or to control the use of an asset or resource, while Allen & Lueck (2002) defined them as the ability to freely exercise choices over the asset in question. Comprehensively, property rights are described as a bundle of entitlements defining an owner's rights and privileges relating to the use of resources, and the limitations on that use. Property rights include the customs, law and regulations governing the right of individuals and farms to have access to and use the environment, and the obligations that go with this.

Environmental problems basically arise when property rights are ill defined (Tietenberg, 2006). Property rights evolve through common law courts, legislatures, voluntary associations and government institutions (Anderson, 2004). Efficient solutions to environmental problems can involve private negotiation, judicial remedies and regulation by the legislative and executive branches of government. According to Stiglitz (2000), internalising the cost of pollution is accompanied by a number of problems. The dispersed nature of nonpoint pollution and the difficulties in measuring it increase the transaction costs of internalising externality through voluntary negotiations among individuals. Achieving a solution through the judicial system is far from ideal as the individual contribution of one farm to the pollution is small compared to the cost of the judicial process, and uncertainties associated with quantifying the impact of pollution may lead to unwarranted outcome through a litigation process. Adequately defined property rights are useful in establishing clear lines of responsibility for the implementation and enforcement of policies. Experiences drawn from elsewhere on the issue of property rights and pollution are also worth noting. Cole & Grossman (2002) cited a court ruling in which the defendants claimed a "right to pollute" groundwater partly by virtue of the fact that they had been doing so for a long time without penalty. The court ruled that regardless of when the polluting acts occurred, and regardless of society's changing views on the propriety of polluting the environment, the defendants had never had a right to pollute the groundwater. The overall benefits of applying this or similar rulings in the Waikato context would need to be carefully weighed as dairy farming is a major economic activity and is linked to many other industries.

Nonpoint pollution control in the Waikato region at present depends largely on moral suasion and on voluntary measures such as the establishment of riparian margins. A lack of property rights and an absence of scientific evidence on the impact of farm nitrogen discharges have meant that other available policy measures have not been implemented. Farms have enjoyed an historical right to affect water quality and have taken advantage of this privilege, as it is clearly easier for a farm to discharge nutrients than to improve water quality by controlling discharge. This practice has not been perceived as infringing the rights of others, and farmers have not seen themselves as polluters. Under the present property rights regime farms are not required to pay the full social cost of the nitrogen pollution they generate. In the absence of explicitly presented environmental and human health costs, farms have no incentive to take these costs into consideration in the decision making process. Ignorance of the impact of external damage leads farms to select management practices that result in greater than the socially optimal levels of nitrogen discharge. This violation of exclusiveness is referred to as externality, and it leads to divergence between the private cost and the social cost of damage caused by pollution.

There is now a clear public desire for improved water quality and an associated questioning of any implicit farm property rights that allow discharge of pollutants. So the desire for improved water quality is in direct conflict with any right to discharge nitrogen. Conflicting interests on nonpoint discharges into water have led to the presentation of contentious evidence from various members of the community to the Environment Court[1]. Evidence presented with regard to the proposed Waikato Regional Plan rules for nonpoint discharges and livestock access to water bodies, show the size of the problem. Draft rules for the Regional Plan proposed by environmental groups were opposed by various land use groups, especially forestry owners, as the rules impose significant costs on their current and future operations. There is also a debate on the spatial dimension of policy implementation such as identifying nutrient sensitive zones for implementing policies[2].
Recognising the need to ensure environmental sustainability, the dairy industry has recently taken the initiative of mapping out a strategy for sustainable environmental management (Dairy Environment Review Group, 2006). The industry, along with

---

[1] In June 2005, the Environment Court convened to hear appeals on provisions in the Waikato Regional Plan dealing with water pollution. .

[2] Identification of sensitive water bodies to have approved farm plans to address nutrient discharges.

Environment Waikato and DairyNZ, has been undertaking an education campaign about the rules relating to dairy effluent. Under the Dairying and Clean Streams Accord some voluntary measures have been taken to exclude livestock from waterways by establishing riparian margins and to practice proper effluent disposal (Ministry for the Environment, 2003). In the Waikato many farmers have adopted land based effluent management systems despite the high capital cost. Incentives have been limited to savings on consent application fees and reduced fertilizer input (Parminter, 1999).

If property rights are well defined, the majority of environmental problems are resolved by markets. Ronald Coase claims that if property rights are adequately defined and the cost of using the market to reallocate property rights is minimal, the allocation of resources will be independent of the initial distribution of rights. This statement is known as the Coase Theorem (Coase, 1960). In addition, Coase is skeptical about the value of using regulations to tax polluters as this ignores the reciprocity of the problem. Demsetz (1967) maintains that clearly defined and enforced property rights are required for any form of human co-operation to be workable, especially a form involving agreement. However, it is often difficult to define property rights adequately in the context of a nonpoint source. According to Depres, Grolleau, & Mzoughi (2007) government can play a positive role by reducing the cost of defining, enforcing and trading property rights. Economic and legal institutions are important when transaction costs are not zero and property rights are not well defined (Allen & Lueck, 1999a). The options available according to Coase (1974) are to do nothing, to force farms to change their practices through legal means, to buy all the sensitive land in the catchment, or to come to a contractual arrangement with farms. Most of these alternatives would be hard to implement. Regulating farms through enforcement of existing laws is difficult due to the problem of proving liability, and dairy farms in the catchment are very productive so any purchase price is likely to be prohibitive. However, in the case of Lake Taupo, there is a proposal that land be bought from willing land owners in the catchment and either retired, or on-sold with a nitrogen covenant attached (so it could be used in a less nitrogen-leaching way) (Ministry of Agriculture and Forestry, 2005b).

Rules are being proposed to control nitrogen discharges in terms of stocking standards and best management practices and the Regional Council is exploring the feasibility of controlling the nutrient problem through tradable emission permits (Environment Waikato, 2005b). This approach privatises the right to access the resource to a pre-specified level, imposing limits on nitrogen discharge to achieve water quality targets. Establishing a market for nitrogen emissions requires the establishment and allocation of rights to discharge. All current privileges would need to be surrendered, and the rights to nitrogen emissions would then be allocated. Ill-defined property rights, together with transaction costs are reported to be the major barriers to the smooth functioning of a tradable permit market in nonpoint sources (Collentine, 2006).

In redefining property rights, it is important to know the level of improvement that is needed and the changes that are necessary. Physically measuring and monitoring diffuse discharges is impractical and expensive, so deriving technically perfect property rules based on real measurement is not a realistic option. The relationship between changes of farming practice in terms of nitrogen reduction, and economic effects is complex and likely to be nonlinear. Moreover, results are observable only over a long time span. Even farmers lack the necessary knowledge to determine the changes that must be achieved and the implications of these changes. However estimating nitrogen discharges and its trade-off with farm income are a basic requirement to design and implement any environmental policy. Therefore nitrogen discharges and associated costs involved in restricting them need to be estimated as an interdependent surrogates for property rule formulation. This necessitates the use of simulation models, which can express nitrogen discharges as a function of production, profit, input use, management practices and location specific environmental attributes. The estimates from simulation models show clearly the impact of various environmental policies. They do need, however, to be precise and reliable if they are to stand up to legal challenge and gain political legitimacy, as legal action would adversely affect the use of these models for emission based policy enforcement purposes. The legal validity of using simulated discharges has been questioned, with arguments centring around the accuracy of the estimates due to stochastic influences outside the farmer's control (Weersink, Livernois, Shogren, &

Shortle, 1998). Nitrogen discharges, for instance, can increase with precipitation intensity. Since the model results are generally not acceptable as evidence, they can be challenged through the legal system, leading to costly litigation. Model-based nitrogen discharges must be scientifically robust if they are to be included in contracts where signatories agree to waive their right to challenge model results through the court system (Romstad, 2002). To help ensure legal validation, it is important to carry out monitoring and measurement of stream water quality and to verify that the model estimates closely match the results taken from the real world. Grouping rights into subsets can improve the efficiency of transactions, as it gives agents the ability to contract on necessary rights only (Depres, Grolleau, & Mzoughi, 2007).

6.2.1 Legislative structure for the environment

Environmental legislation in New Zealand centres on the Resource Management Act of 1991 (RMA) which promotes voluntary and regulatory approaches to control non-point source pollution. Most of the policies and rules that influence diffuse source pollution are managed by regional or local governing bodies. The Act confers primary powers on local authorities and the Environment Court[3], a specialist institution within the New Zealand court system. In general, policies are decided locally and are interpreted by the Court.

Section 32 of the RMA states that local authorities must consider alternatives; assess the benefits and costs of objectives, policies, rules and other methods for environmental improvement. The other methods mean the provision of information, services or incentives, levying charges including rates. However the act by itself has reportedly not empowered the councils to implement environmental policies (Denne, 2006). The RMA does not provide clear tools for managing the environment. Even though section 24 gives the Minister for the Environment a role in investigating the use of economic instruments including charges levies, and other fiscal measures and incentives in order to achieve the purposes of the Resource Management Act, this is not accompanied by specific powers that would enable the Ministry to use such

[3] Further details about Environment court is available at http://www.justice.govt.nz/environment/home.asp.

economic instruments. The amended version of Section 32 removes the explicit reference to charges and incentives, and states only that local government should consider whether any specific objective is the most appropriate way of achieving the purposes of the Act. Sections 9 and 15 of the RMA give regional councils statutory responsibility for preventing, remedying and mitigating adverse effects on water quality in their regions. Requirements for water quality enhancement could be more specific in the RMA. For instance in Oregon, legislation enables the development and application of best management practices to protect water quality (Boyd, 2000).

6.2.2 Role of the environmental agency as social planner

Given current challenges there has been increasing reliance on public remedies to redefine property rights, and it can be argued that government intervention is required to regulate nutrient discharge into water. This could be either in the form of redesigning the current property rights structure to allow for a private market solution, or in implementing policy instruments that will convey the desired property rights structure. Solutions to the pollution problem through public initiative are broadly categorised into direct regulation and market based solutions.

Provided the adequate legal authority, the environmental agency can play a significant role in defining and assigning property rights to facilitate bargaining solutions. In the absence of well defined property rights, the regulator can play a vital role in assigning initial entitlements (Richards, 2000) and set environmental quality targets, choose instruments to accomplish goals, monitor compliance, and initiate actions to enforce rules. These actions are likely to reduce the transaction costs. Gangadharan (2000) suggests that regulator designed programmes could facilitate the evolution of markets that encourage participation.

Ruhl et al (2003) offers a model institutional structure for a catchment management act. They argue legislation must empower the local agency with the authority and responsibility for managing surface and ground water quality and quantity issues, and the relevant agency must be capable of establishing a democratically based legitimacy at regional and local levels. The institutional structure must have the capacity to carry out scientific, economic and social analysis, as well as having responsibility for making policy and regulatory decisions through public, transparent procedures based on best available evidence. The authors also stress the need for local authorities to play a stronger partnership role, with a shift from acting as regulator/ advisor to being facilitator/partner.

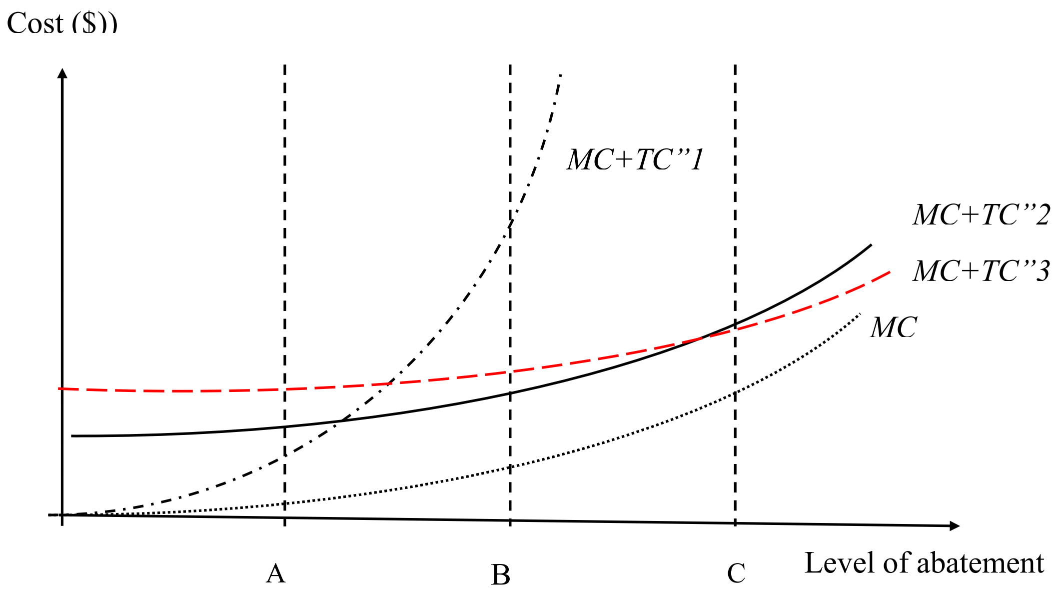
6.3 Transaction costs

Costs involved in defining property rights and implementing policies are referred to as transaction costs. Coase (1937) first described the concept of transaction costs in his article on the nature of the firm. Williamson (1985) defined transaction costs as the cost of organizing and transacting exchanges - including search and information costs, bargaining and decision costs, policing and enforcement costs. It has also been defined as the cost of information gathering, contracting and enforcing property rights (Allen, 1991; Allen & Lueck, 1999a; Bromley, 1989). McCann, Colby, Easter, Kasterine, & Kuperan (2005) defined it as the resources used to define, establish, maintain, and transfer property rights.

The size of transaction costs involved in eliminating externalities is affected by the number and range of agents, by technology, policy under consideration, level of uncertainty, asset specificity, institutional environment and the cost of abatement or amount of abatement (McCann & Easter, 2000). In addition the diffuse nature of nitrogen discharges and the difficulties associated with measuring them, the intensity or level of monitoring, and the choice of policy instruments tend to influence the transaction costs. Carefully designed contracts can minimise the transaction costs (Depres, Grolleau, & Mzoughi, 2007). Defining property rights based on surrogate measures minimise transaction costs (Griffin, 1991).

In the presence of transaction costs, the assignment of property rights plays a crucial role in identifying efficient solutions. Incorporating transaction costs in policy analysis leads to the design of policies and institutional arrangements with lower transaction costs (McCann & Easter (1999). The authors also state that including transaction costs in environmental policy analysis would lead to lower levels of optimal abatement because the total cost of achieving a given level of abatement is much higher.

Figure 6.2 Increasing, constant and decreasing transaction costs



Source: Adapted from McCann & Easter (1999)

The relationship between the transaction costs of various policies and the level of abatement is illustrated in Figure 6.2. *MC* stands for marginal cost of abatement. *TC* is transaction cost. In the absence of a marginal damage function, the optimal policy will differ depending on the politically determined target level of abatement. If the target is set at A, the optimal policy is 1. This could, for instance, represent a situation where persuasive argument or a perceived threat of further restriction could achieve a small amount of abatement. However the cost of achieving further abatement is significantly higher or ineffective by adopting these measures. If the target level is B then a mixture of policies 1 and 2 is optimal. Policy 2 could be taxes or standards. The choice of optimal strategy at level C depends on the total cost of abatement, since each policy incurs high initial transaction costs. Policy 3 could be tradable emission permits, where the initial transaction costs are higher. If the marginal abatement costs are constant for these policies, the policy with the lowest marginal cost will always be optimal regardless of the level of abatement.

6.3.1 Components of transaction cost

Transaction costs generally include information, contracting and enforcement costs. Information costs are the cost of targeting farms, and finding the links between the discharges and farming practices. Contracting costs are the administrative and staffing costs involved in designing and finalising contracts with targeted farms. Enforcement costs are the costs of monitoring to audit the adoption of nitrogen discharge measures, estimating discharge levels and administering incentives.

McCann, Colby, Easter, Kasterine, & Kuperan (2005) and (McCann & Easter, 1999) describe transaction costs involved with an environmental policy as the sum of a range of costs involved in research information gathering and analysis associated with defining the problem:

- enactment of enabling legislation, including lobbying and the cost of public participation, or alternatively, the cost of changing the law through the courts or modifying existing regulations
- design and implementation of policy, which may include costs of regulatory delay;
- support and administration of the ongoing programme;
- contracting costs, which may include additional information costs, bargaining costs, and decision costs, which involve when markets are established for pollution;
- monitoring and detection, which may include both the monitoring of the environmental outcomes or level of compliance with the regulation, tax subsidy scheme, or private contract, as well as the development of monitoring technologies;
- prosecution/ inducement/ conflict resolution costs incurred in cases of non-compliance.

In the case of nonpoint pollution, transaction costs are mainly incurred by the social planner. Farms too, have to spend time in adopting some policy options such as preparation of nutrient budgets, operating effluent irrigators and talking to environmental authorities. There have been few empirical studies of who bears the different shares of transaction costs. Table 6.1 presents one set of possibilities, adapted from the work of McCann et al.

Table 6.1 Composition of transaction costs

Type of transaction cost	Incurred by		
	Legislature/ Courts	Agencies	Farms
Research and information	+	++	+
Enactment or litigation	++	+	++
Design and implementation		++	+
Support and administration		++	+
Contracting		+	++
Monitoring and Detection		++	+
Prosecution and enforcement	+	++	+

In selecting policy, abatement costs and transaction costs need to be considered together. McCann et al (2005) and Kampas & White (2004) state that a policy should not be rejected on the grounds of high transaction costs alone, since there may be a tradeoff between transaction costs and other types of costs. Some policies with high transaction costs are likely to have low abatement costs. The components of a policy can vary depending on the policy itself, and existing institutions. Implementation of tradable emission permits, for instance, requires changes in the institutional environment and legal system to define and establish pollution rights. A government or environmental agency has to be involved in the allocation of property rights (in this case discharge allowances among various land use groups). Further on, the environmental agency has to record, monitor and enforce the trading of permits. Detailed analysis of transaction costs associated with the various environmental policy instruments is beyond the scope of this study.

6.3.2 Measurement of transaction cost

Choice of least cost policy requires ex ante evaluation of transaction costs. One option to quantify transaction costs is to find similar programmes that have been implemented elsewhere and examine the costs of these programmes. However this approach has the following problems (McCann & Easter, 1999); the programmes may not be representative of institutions involved in the local case; similar programmes may have not been implemented before and there is likely to be a piece meal approach when values are pulled from different programmes. As an alternative they suggested ex ante evaluation of transaction costs. This involves in-depth interviews with selected key individuals of environmental agency using measurement techniques developed for nonmarket valuation. For instance the Contingent Valuation Method (CVM)[4] was used to derive informed opinions for a Minnesota River pollution problem in the USA. This strategy has also been adopted by Fang, Easter, & Brezonik (2005). McCann, Colby, Easter, Kasterine, & Kuperan (2005) suggested the use of Choice Modelling to separate out the various components of transaction costs at measurement. They also suggested that collecting and reporting transaction cost data should become a routine part of environmental agency activities.

6.4 Information asymmetry

In economics, information asymmetry occurs in transactions where one party has more or better information than the other. Most commonly, information asymmetries are studied in the context of principal-agent frame work in pollution problems (Moxey, White, & Ozanne, 1999; Ozanne & White, 2007).

Once property rights are defined, successful implementation of policies depends on information acquisition. Catchment based approaches for water quality improvement requires much more information about nitrogen discharges. Farm nitrogen discharge is a complex problem involving multiple polluters, spatial factors, complex fate and transport processes, time lag and stochastic environmental factors, which affects assigning pollution to particular entity difficult. As a result farms tend to have more information than a social planner, who implements policies. In discussing optimal environmental policy, the traditional approach has been to assume simply that polluters comply with policy. However, compliance by farms cannot be guaranteed in any situation where information asymmetry is present.

---

[4] CVM is a non market valuation strategy

Information asymmetry between farms and the social planner in environmental contracts is classified into two types: adverse selection and moral hazard. Adverse selection occurs ex ante to a farm's decision to enter an environmental scheme, where the social planner is unable to accurately quantify the nitrogen discharges and farming profit losses incurred by participating farms. Moral hazard arises ex post to a farm's decision to enter an environmental scheme, where the social planner cannot verify perfectly and without cost that all farms are fully compliant with their obligations. This provides farms with an incentive to renege on their obligation, if they can avoid detection by the social planner. In the presence of information asymmetry, therefore, it may not be possible to achieve the optimal level of abatement.

Differences in farm abatement cost contribute to information asymmetry (Hart & Latacz-Lohmann, 2005). Farms with high compliance costs are likely to violate the policy requirements as their pay off to cheating (abatement cost saved) is larger than that of other farmers. Farms may mislead the social planner by reporting lower than actual levels of discharge. This provides them with an incentive to ignore policy regulations or avoid penalties. This situation becomes possible when monitoring is imperfect, costly and where there are difficulties in observing some activities related to discharge. Therefore carefully designed contracts and effective monitoring are imperative in alleviating any information asymmetry.

6.4.1 Contract design

Contracts between the social planner and farmers can be used to incorporate changes and extensions to property rights and act as a vehicle for implementing environmental policies. In designing environmental contracts, the wealth of experience drawn from agricultural production contracts could be used to design environmental contracts. Principle agent models played an important role in designing agricultural production contracts to address information asymmetry (Allen & Lueck, 2002). For instance, in an environmental context they have been used to address the problem of adverse selection (Moxey, White, & Ozanne, 1999).

Effective policy implementation is facilitated by having as many dispute free contracts as possible. The Vittel case (Depres, Grolleau, & Mzoughi, 2007) showed the value of using a competent research team to define what actions are to be permitted or forbidden in order to achieve the desired standard of water quality. The research team can also play a mediating role in ensuring a mutual understanding of the varying interests of the social planner and the farmer. Šauer, Dvořák, Lisa, & Fiala (2003) suggest that a special institution could be created to negotiate and act as a mediator in negotiations over surface water pollution. In the Waikato, DairyNZ could play this research team role since it is seen by farmers in a positive light as helping them to keep the environment clean (Tagg, 2007). In contrast, Environment Waikato, the social planner for the region, has sometimes been perceived as bureaucratic, imposing rules and limits on dairy farming (Stringleman, 2007). Having a good research team as a mediator can help to resolve disagreements over technical and environmental questions and therefore facilitate the implementation of policies.

Information asymmetry is intensified when policies are based on surrogate measures such as estimated nitrogen discharges. Social planners do not possess perfect information regarding the practices of the farms with whom they enter into a contract. There are two forms of information advantage that farms may enjoy over the social planner. When signing a contract to reduce nitrogen discharge, the farm always has an incentive to declare an inflated discharge level, enabling it to minimize the subsequent actions it needs to take. In addition to this, after signing a contract there is an incentive for farmers to renege on contracts if they can do so and avoid detection by the regulator. In order to minimize this sort of cheating, contracts should include clauses enabling free access to farm accounts and visual inspection of farms. Contracts may specify input levels and best management practices to control nitrogen discharge levels, or they can place a quota on the amount of nitrogen discharge on a farm. Violating the contract could be subject to penalties. The challenge is to devise contracts that function effectively in the presence of information asymmetry. The empirical analysis section of this chapter shows potential solutions to this problem.

Restriction of nitrogen discharges is a particularly sensitive issue, as nitrogen is one of the most important factors determining agricultural farm production. The proposed environmental policies have the potential to curtail pre existing property rights of farms, and may even threaten their viability (Hardaker, Humie, Anderson, & Lien, 2004). Contracts that include compensation in exchange for restriction on nitrogen discharges are useful in overcoming information asymmetry. For instance Moxey, White, & Ozanne (1999) developed an optimal truth-revealing mechanism for environmental policies by using transfer payments coupled with input quotas. Therefore, if appropriate, contracts need to include provisions to compensate farmers for the profit they forgo by adhering to the contract. Information asymmetries can be minimised by assigning a higher discharge quota to a farm on productive land, in conjunction with reduced compensation to reflect the lower profits forgone. This has the effect of removing the incentive for the farm to select as inappropriate contract. How to source the funds to pay compensation is beyond the scope of this chapter, However according to section 3.9 of the Waikato Regional Plan, Environment Waikato would consider financial support of projects which minimize the impacts of land use activities on water bodies (Environment Waikato, 2007c). But using public funds to reduce discharges has been criticised on the grounds that it is a victim-pays regime (Environmental Defence Society, 2007). This society stresses that the social planner should take the view that land owners were not authorized to pollute the water and did not have a presumptive right. Burrows (1979) stated that "polluter subsidy perversely moves industry output (pollution) away from the Pareto optimum even though individual firms respond by cutting their output (below) the pareto optimum level". Palmquist (1990) also stated that subsidies could result in a substantial pollution in the long run as a result of increasing number of operators. As an alternative Macho-Stadler & Perez-Castrillo (2006) suggest non monetary penalties such as publicising anti-social behaviour or even imprisonment.

The ratchet effect should also be taken into consideration when designing contracts. It occurs when the principal (social planner) and agent (farm) engage in a series of contracts over time and it shows as an increase in nitrogen discharge standards in the light of past performance (Allen & Lueck, 1999b). If the principal does not commit to constant contractual terms, it will use information collected on past performance to set new standards of agent behaviour in order to improve water quality. This problem may be solved by signing long term contracts. However when there is an ex ante uncertainty around the environmental impact of a policy and the damage function is not fully understood, property rights established on surrogate measures may need to be updated or revised. The time frame for contracts, therefore, has to be carefully designed considering the lifespan of machinery or technology adopted currently.

The efficient working of environmental contracts is affected where there is a large number of heterogeneous participants (Depres, Grolleau, & Mzoughi, 2007). Targeting farms with lower abatement costs tends to reduce the moral hazard problem as they have a smaller payoff for violating a contract. This is likely to reduce monitoring efforts required. Farms with lower abatement costs can also be assisted to install best management practices with a greater degree of detail. Targeting these farms will incur higher selection costs but may improve subsequent policy performance (Moxey, White, & Ozanne(1999).

6.4.2 Monitoring

Compliance monitoring is important for agri- environmental policy as policy effectiveness depends largely on the ability of the social planner to monitor and enforce any contract (Weersink, Livernois, Shogren, & Shortle, 1998). The diffuse nature of nitrogen discharges from agricultural sources poses serious challenges for monitoring and enforcement. Since farms have more information than the social planner about their emissions, effective methods need to be devised to ensure that monitoring is efficient. Moral hazard tends to rise when the cost of monitoring depends largely on monitoring effort (larger variable cost component to fixed costs) (Ozanne, Hogan, & Colman, 2001). The problem of moral hazard can be minimised by increasing the level of monitoring, and thus the probability of detection.

Monitoring and adoption of best management practices at the extensive margin are relatively easy to achieve, when compared to monitoring changes at the intensive margin. For instance, monitoring adoption of riparian margins is straight-forward and because it does not require repeated effort, it costs less. Monitoring nitrogen discharges as a result of changes in the intensive margin, however, is difficult and costly. Nitrogen discharge can be reduced to a certain extent by adopting best management practices, but the rest of the reduction has to come from management of nitrogen fertilizer, stocking rate and supplementary feeding.

Animal density (stocking rate) is increasingly being used as a monitoring indicator and standard for nutrient management policies (Ribaudo et al., 2003) as it provides a straightforward, relatively easy-to-calculate indicator of a farm's nitrogen discharge potential. However, application of this approach in dairy farming systems is fraught with its assumption on feeding practices and fertiliser application and geophysical farm variations. It may be possible to increase non compliance, keeping the stocking rate constant. A study based on Waikato dairy farm trials shows the impact of varying the level of nitrogen fertiliser application while keeping the stocking rate constant (Table 6.2). It indicates that there is an incentive to increase the level of nitrogen fertiliser. Woodford (2006) also notes that constraints on nitrogen fertiliser application, particularly on dairy farms, are likely to be environmental rather than economic. In addition, the impact of animals on nitrogen discharge depends on the type of animal and such characteristics as body weight and geo spatial factors such as soil type, topography and hydrological flows.

Table 6.2 Nitrogen fertiliser and stocking rate

Variables	**Trial 1**	**Trial 2**
Stocking rate ((Cows/ha)	3.34	3.34
Nitrogen fertilizer application (Kg/ha)	0	200
Profit ($/ha)	1957	2244
Nitrogen discharged (Kg/ha)	30	65

Source: Ledgard & Thorrold (2003)

However biophysical simulation models developed are capable of capturing the above factors, when estimating nitrogen discharges. The use of a calibrated and validated catchment specific simulation model enhances the transparency of estimation. Given political acceptability and legal validity, these estimates can play a role in defining a relevant subset of rights that need to be considered in implementing specific policies.

In the Netherlands, nutrient discharges from farming are calculated using a nutrient (or mineral) accounting system introduced by the government in 1998 (Ondersteijn, Beldman, Daatselaar, Giesen, & Huirne, 2002). This system requires farmers to self report estimated pollution as well as production, and additional information is acquired from off-farm sources. Feed and fertilizer suppliers, for instance, are required to supply farm-specific sales details to the social planner. A New Zealand example of such a model used for nitrogen discharge is the Overseer nutrient budget model (Ledgard & Power, 2006). Nitrogen discharge estimates from the Overseer model have been validated using direct measurements across New Zealand (Appendix 4.1).

Since it is difficult to monitor inputs with pollution potential such as fertiliser and supplementary feeds, it may be useful therefore to devise methods which enable output-based monitoring and shifting the focus from monitoring input to output is likely reduce the monitoring burden. Weersink, Livernois, Shogren, & Shortle (1998) showed that establishing a direct relationship between an observable variable and estimated pollution discharge is essential for the success of policies like those covering emission charges. In this respect the profit pollution frontier produced in chapter 5 has immense value. Peterson & Boisvert (2004) show that monitoring corn yields could be substituted for the potentially costly and intrusive monitoring of fertiliser use. Bontems, Rotillon, & Turpin (2005) also use the relationship between the pollution and production of a particular farm type to design incentive-based environmental policy. However, output-based monitoring needs to be carefully planned and be sufficiently flexible to accommodate the impact of weather and productivity growth.

The periodic use of complex models on a farm may be demanding, particularly in terms of data collection and specification. So Geographic Information System (GIS) can be used to provide the information needed to design environmental policy (Cook & Norman, 1996; Horst, 2005), including farm size, stocking rate, pattern of production and pollution-related geophysical attributes. These spatial tools can then be used to define the most relevant geographical target areas for policy intervention. Once farm information such as dairy cows, fertiliser application, adoption of best management practices and milksolids production is established in GIS, the internet can be used by the social planner to access farm related information. Since most farmers have internet access this approach enhances transparency and reduces the transaction cost of overall policy procedures. Gleeson & Carruthers (2006) also showed that web-based software tools hold substantial and largely untapped potential for developing cost effective farm-based environmental management.

Since the measurement of water quality is costly in spatial and temporal terms, researchers can employ remote sensing techniques to retrieve required information. Boyd (2000) describes monitoring strategies that include proposals for remote-sensing via satellite to check compliance with such land management and construction requirements as buffers and cover crops. Remote sensing techniques are currently used for lake water quality information retrieval (Sudheer, Chaubey, & Garg 2006), showing that precision of monitoring will increase with the development of technology. Choe & Frase (1999) also revealed that mean monitoring cost can be reduced by effective use of appropriate technology (Choe & Fraser, 1999).

6.5 Model setup

This section develops a conceptual model for monitoring a farm which is subject to an agri-environmental policy that requires an income-reducing action on the part of the farm. It proposes a penalty and incentive scheme to alleviate information asymmetry and to reduce transaction costs. There is a social planner (regulator) and n agents (farms) in the catchment. The model is based on a number of assumptions. Farms in the catchment are heterogeneous[5] in terms of abatement costs. Heterogeneity of farms is denoted by i, $i \in \Theta$. The social planner knows of the existence of different types of farms, their compliance costs and the proportion of each type in the catchment. However, the social planner cannot exactly tell which type each farm is.

---

[5] Causes of heterogeneity are soil type, land slope, distance from the main stem of the river, input use etc.

The production and pollution relationship is denoted as a function of nitrogen discharge as follows:

$$y^i = f_i(x_n) \qquad (1)$$

y^i -yield per ha for a farm, x_n nitrogen discharge per ha. This function is assumed to be a twice differentiable and concave in terms of nitrogen discharge. Nitrogen discharge indirectly represents the input use and management choice of farm.

In the absence of contractual arrangements between farms and the social planner, farms always tend to discharge nitrogen that maximises profit π_0^{*i}. g_i is farm profit:

$$\pi_i^0 = g_i(x_n^{*i}) \qquad (2)$$

The optimal discharge level x_n^{*i} is determined solving the first order condition.
As discussed in the previous chapter, when farms are contracted under a joint environmental policy instrument they tend to maximise the profit as follows:

$$\pi_i^R = g_i(x_n) - \tau(x_n - x^R) \qquad (3)$$
$$x \leq x^R$$

The abatement cost, C, is defined as the difference between the unrestricted maximum level of profit π_0^{*i} and the restricted level of profit π_R^{*i}. Therefore expected farm profit can be modelled as

$$C = \pi_0^{*i} - \pi_i^R \qquad (4)$$

The abatement cost function can be defined as follows: farm *i* can produce an abatement A_N with a cost of abatement C_i

$$C_i(A_N) = \beta A_N^2. \qquad (5)$$

The social planner objective function (π^s) is to maximise the welfare by minimising cost of abatement and monitoring to achieve the exogenous environmental target of reducing nitrogen discharges to a certain level (z). λ_i is the percentage of farm land belonging to each farm type. r_i is the level of total nitrogen discharge reduction from each farm type.

$$\pi^s = Max(\pi_i^R - m(e)) \qquad (6)$$

$$z = \lambda_i r_i$$

m (e) -monitoring cost of the social planner. Monitoring costs depend on the level of monitoring effort employed (e)**.** The level of effort may be represented, for instance, by labour hours or the frequency of visits and effort put into scrutinizing farm activities.

The specification for monitoring cost is based on Falconer & Whiteby (Falconer & Whiteby, 1999), who suggest that the monitoring process comprises both fixed and variable cost components.

$$M(e) = m_1 + m_2 e + m_3 e^2 \qquad (7)$$

m_1 is the fixed cost include contracting and updating costs. m_2, m_3 are variable cost parameters depending on the level of effort.

In the absence of perfect observation of nitrogen discharges, the monitoring strategy of the social planner and reporting strategy of the farm become strategic decisions. Level of monitoring a farm can be denoted as α depends on the level of effort $\alpha = f(e)$. Level monitoring is measured in terms of time spent to monitor a farm. The probability of detection depends on the level of monitoring. The following equation captures the relationship between the level of monitoring and probability of detection.

$$P = f(\alpha) \quad (8)$$

P is the probability of detection, α is the level of monitoring. In the following model setup probability of detection is, for the sake of simplicity, treated as equal to the level of monitoring. i.e $P = \alpha$.

A farm may choose to report q rather than the true emission level x_n but a farm never reports a higher level of emission than its true level $(q \leq x_n)$. If a farm is caught reporting discharge levels below the true level, then a penalty is imposed on the farm, in addition to the evaded tax. This penalty takes a form of function $\theta(x_n - q)$. It can be an increasing function of violation (v_i). $v_i = x_n - q$. Violations are penalised according to a quadratic penalty function[6]:

$$f(v_i) = \phi v_i + \sigma v_i^2 / 2 \quad (9)$$

where ø and ó are >0. In empirical application, the constant marginal penalty schedule is implemented where ó is set to 0.

$$\theta(0) = 0, \theta^{'}(v) \geq 0, \theta^{''}(v) \geq 0, x \geq 0$$

Under the strategic behaviour a farm's expected income I_s is

$$I_s = E\pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau(q - x^R) - \alpha\tau(x_n - q) - \alpha\theta(x_n - q) \quad (10)$$

I_s can be equal or less than the unregulated profit π_0^{*i}

If the optimum amount a farm should discharge and report to maximise income depends on level of monitoring, then penalty and rate of tax can be denoted by following first order conditions given interior solutions.

$$\frac{\partial E\pi_i^R}{\partial X_n} = g_i(x_n)^{'} - \alpha\tau - \alpha\theta^{'}(x_n - q) = 0 \quad (11)$$

---

[6] The functional form for the penalty function is adopted from Stranlund (2007)

$$\frac{\partial E\pi_i^R}{\partial q} = -\tau + \alpha\tau + \alpha\theta^{'}(x_n - q) = 0 \qquad (12)$$

Profit if not caught

$$I_s = \pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau(q - x^R) \qquad (13)$$

Profit if caught

$$I_s = E\pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau(q - x^R) - \tau(x_n - q) - \theta(x_n - q) \qquad (14)$$

Profit for truthful behaviour is I_t

$$I_t = \pi_i^R = g_i(x_n) - \tau(x_n - x^R) \qquad (15)$$

The degree of violation depends on two factors, the first being the relative merits of I_t and I_s. When the difference is larger, then the likelihood of strategic behaviour is higher. Second is the abatement cost: when C is larger, then payoff for strategic behaviour is higher. If the abatement costs of farming systems are known to the social planner, targeting farms with the lower C, will reduce the level of strategic behaviour. Macho-Stadler & Perez-Castrillo (2006) also show that when there is a wide range of farm types and responses, it is sensible to focus on farms that place a lower value on reducing pollution. While targeting and taxing farms specifically can be politically unpalatable, it may be possible if the tax is replaced with transfer payments and coupled with penalties. In this case there is a trade off between the cost of monitoring and the cost of incentive payments (Choe & Fraser, 1999). Providing increased incentives and appropriate targeting of farms tends to reduce the cost of monitoring. Targeting farms also has the effect of concentrating monitoring effort, so that there is a smaller difference between true income and the income that would be expected from behaving strategically.

6.5.1 Selective/Target monitoring

Targeting involves separating farms into larger non-target and smaller targeted groups, and increasing the probability of detection of those in the target group. This should have the effect of reducing the abatement and monitoring costs. Spatial targeting lowers the cost of administering contracts when implementing environmental policy (Wu & Babcock, 2001). Monitoring resources are used more efficiently and the moral hazard problem is reduced (Fraser, 2004) because with fewer farms being monitored, the level of monitoring on those farms is increased. Targeting the farms reportedly increases information efficiency while lessening the data collection effort (Farzin & Kaplan, 2004). Spatial targeting farms reduced transactions costs by 75% in reducing runoff from dairy farms in the US (Carpentier, Bosch, & Batie, 1998).

Fraser (2004) used a monitoring/penalty system to target farms. He classified targeting into three types: "Resource neutral", "Non resource neutral" and "Resource neutrality". In selective monitoring a sub group of farms with lower abatement costs is targeted implying higher probability of detection than farms outside the sub group.

Resource neutral

Monitoring resources are not changed with the introduction of targeting, so the probability of being detected increases for one group but decreases for the other groups. The higher level of detection (α_h) is achieved for the target group by shifting monitoring resources away from those in the non target group, where the level of being detected is low (α_l). Under this scheme the expected farm income is:

Targeted group

$$I_s = E\pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau(q - x^R) - \alpha_h \tau(x_n - q) - \alpha_h \theta(x_n - q) \qquad (16)$$

Non-targeted group

$$I_s = E\pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau(q - x^R) - \alpha_l \tau(x_n - q) - \alpha_l \theta(x_n - q) \qquad (17)$$

$$\alpha_l < \alpha < \alpha_h$$

Even though this option is likely to minimise strategic behaviour in the targeted group, it is also likely to encourage the non target group to violate the policy more as the advantages of violating reflected in the level of relative differences of income that result from strategic behaviour and truthful behaviour.

Non-resource neutral

In the "non resource neutral" option the chance of one farm group being detected increases without affecting the detection level of the other groups because additional resources are provided to monitor the target group. The detection level in the non target group ($\alpha_l=\alpha$) is unchanged. Under this scheme the expected farm income is: .

Targeted group

$$I_s = E\pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau(q - x^R) - \alpha_h \tau(x_n - q) - \alpha_h \theta(x_n - q) \qquad (18)$$

Non- targeted group

$$I_s = E\pi_i^R(\alpha, x_n, q) = p_i f^i(x_n) - \tau(q - x^R) - \alpha\tau(x_n - q) - \alpha\theta(x_n - q) \qquad (19)$$

$$\alpha < \alpha_h$$

In this scheme strategic behaviour will be reduced, but farms in the non target group still have an incentive to violate. In addition, the social planner has to dedicate more resources to monitoring, increasing the *m (e)*. In selecting an appropriate targeting level there is a trade off between minimising strategic behaviour, and the cost of monitoring.

Resource neutrality

The third option, "resource neutrality", adjusts the level of monitoring, the tax rate, and the size of the penalty. Farm groups are differentiated in terms of level of monitoring and penalty parameters. The objective here is to minimise strategic behaviour without increasing the level of monitoring. Non target group farmers are subject to a higher level of penalty ɵ. Increased monitoring levels for the target group are resourced by decreasing the level of monitoring in the non target group. The expected income resulting from strategic behaviour is minimised in the target group by increasing the chance of being detected. In the non target group, it has been achieved by increasing the penalty. Under this scheme expected income of farms is:

Targeted group

$$I_s = E\pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau(q - x^R) - \alpha_h \tau(x_n - q) - \alpha_h \theta(x_n - q) \qquad (20)$$

Non-targeted group

$$I_s = E\pi_i^R(\alpha, x_n, q) = g_i(x_n) - \tau_h(q - x^R) - \alpha_l \tau_h(x_n - q) - \alpha_l \theta_h(x_n - q) \qquad (21)$$

$\alpha_h < \alpha < \alpha_h$

$\theta < \theta_h$

$\tau < \tau_h$

These schemes could be used in the environmentally sensitive sub regions of the catchment. However, monitoring schemes themselves can never completely eliminate strategic behaviour, and information asymmetry may still remain. Logically, by increasing the penalty to colossal levels, strategic behaviour could be reduced with less monitoring effort, but this approach does not reflect judicial reality, which will not permit high higher levels of penalty (Ozanne, Hogan, & Colman, 2001).

6.6 Empirical application and discussion

Data for the empirical application of equations described in the model setup in section 6.5, particularly farm types and their responses to parametric nitrogen discharge reductions, are taken from the results of the work reported in the chapter 5.

Monitoring

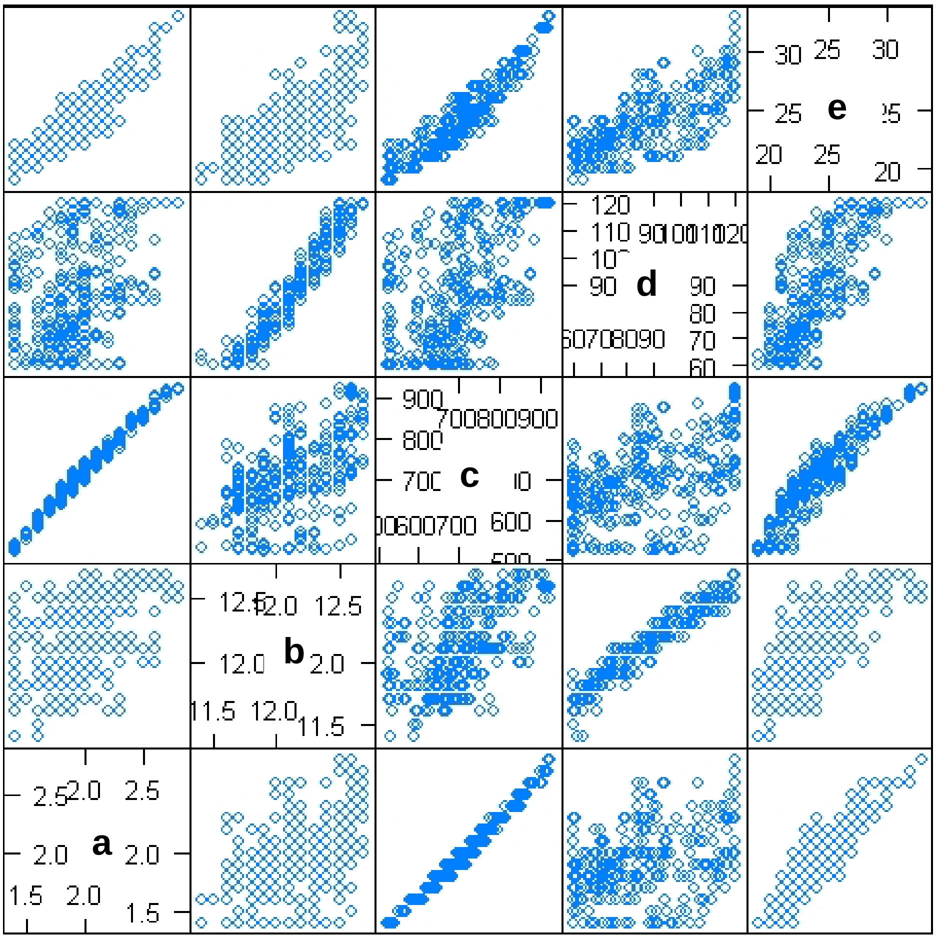
In establishing a relationship between production variables and nitrogen discharge the Whole Farm Model simulation results for a moderate farming system from the previous chapter is subject multivariate exploratory data analysis, using scatter plot matrices (Figure 6.3). Scatter plot matrices are generated using the Lattice package (Sarkar, 2008) in an R programming environment (R Development Core Team, 2007).

.

Figure 6.3 shows a clear relationship between the key variables: milksolids production and nitrogen discharges; pasture production and nitrogen fertiliser applied; stocking rate and milksolids production. These relationships are useful for monitoring purposes as they provide *a priori* notions about the relationship between nitrogen discharges and other observable variables.

A farm is required to declare only the level of production, estimated level of emission, input use and farm size. The social planner can confine monitoring effort to those farms which report conflicting patterns. An unusually low level of reported emissions, for instance, would be a good reason for putting monitoring processes in place. Pasture production can be useful as an indicator of intensification, and pasture growth and cover can be monitored with a high degree of accuracy using satellite technology (DairyInsight, 2006). The accuracy of model simulations can be improved by combining satellite data with the simulation model to reflect spatial variability in pasture growth at a farm scale (Hill, Donald, Vickery, Moore, & Donnelly, 1999). All this reduces information cost while minimising the sort of inadequate monitoring which would allow farms to behave strategically on agreements.

Figure 6.3 Scatter plot matrix



a- Stocking rate (Cows/ha)

b- Pasture production (000'kg /ha)

c- Milksolids production (kg/ha)

d- Nitrogen fertiliser applied (Kg/ha)

e- Nitrogen discharge (kg/ha)

Once these relationships are established for different types of farms, and contracts are signed on required level of nitrogen discharge reduction, the social planner has to monitor compliance with contractual obligations.

Table 6.3 Simulated farm responses to nitrogen level restriction

	Baseline	10%	20%	30%	40%
Moderate					
Profit($/ha)	1429.00	1406.00	1315.00	1197.00	985.00
Stocking rate(Cows/ha)	2.68	2.29	1.91	1.74	1.40
Pasture (Kg/ha)	12592.09	12576.85	12360.04	12006.31	11878.32
Milksoilds (K/ha)	908.74	813.76	705.47	644.03	522.12
N-fertiliser(Kg/ha)	120.00	118.00	112.00	81.00	60.00
N- discharge (Kg/ha)	31.94	28.61	25.47	22.31	19.24
Intensive					
Profit($/ha)	2825.18	2843.00	2775.00	2663.00	2536.00
Stocking rate(Cows/ha)	3.53	3.19	3.21	2.91	2.40
Pasture (Kg/ha)	14993.24	14896.15	14566.26	14415.38	14281.77
Milksoilds (K/ha)	1640.91	1512.08	1514.47	1388.65	1177.88
N-fertiliser(Kg/ha)	270.00	258.00	220.00	199.00	189.00
N- discharge (Kg/ha)	60.79	53.75	48.58	42.27	35.93
Extensive					
Profit($/ha)	880.00	829.00	788.00	651.00	461.00
Stocking rate(Cows/ha)	2.27	1.81	1.46	1.20	0.95
Pasture (Kg/ha)	9039.52	9120.00	9288.73	8961.44	8349.27
Milksoilds (K/ha)	714.32	601.33	495.53	413.74	329.93
N-fertiliser(Kg/ha)	60.00	60.00	58.00	36.00	4.00
N- discharge (Kg/ha)	27.41	24.19	21.87	19.16	16.36

In order to find discrete quantities of output, input and nitrogen discharge for various levels of nitrogen discharge reduction, selective activities from the WFM simulation results reported in table 5.3 are used to form a simple model using linear programming. Farm level linear programming models have been used in many studies to model environmental and economic policy (Bartolini, Gallerani, Raggi, & Viaggi, 2007; Taylor, 1992). The linear programming model is solved using Premium Solver Platform Version 8 (Frontline Systems, 2007). This is a spreadsheet Solver, an extension of that bundled with Microsoft Excel. The results are tabulated in Table 6.3. Scatter plot matrices and linear programming output are helpful to trace the relationship between variables to reduce information asymmetry. Farms may be asked to complete an online form and the information given in these can be useful in confirming farm type. Farm abatement cost curves similar to those derived in the Chapter 5, can be used to target farms which place less value on reducing pollution. Abatement cost information generated by the modelling framework is also useful in making decisions on the size of transfer payments. It is particularly relevant for cost sharing initiatives between farms and the social planner as it enables the social planner to base any incentive packages to farms on accurate information about the marginal abatement costs associated with a specific farming system (Yiridoe & Weersink, 1998). If the social planner is unable to observe the farm type, there is an incentive for the low abatement cost farm to declare itself as high abatement cost farm, or vice versa, depending on the policy instrument. Having more information about farm types reduces the cost of negotiation in environmental contracts (Moxey, White, & Ozanne, 1999). However given the heterogeneity of farms and the complexity of input transformation into a final product, this kind of relationship should be treated only as a rough guide. Further these relationships need to be altered to accommodate best management practices and productivity growth by adopting innovative eco-friendly technologies. Variations in the production and potential nitrogen discharges due to weather can be accommodated by adjusting the functional relationship based on annual yield data derived from surveys and the regional monitoring farms.

Imperfect compliance

There are few empirical analyses of imperfect compliance in the literature. A simple two farm model is presented here, with some assumptions regarding functional specifications and parameter values. Farms are considered to be profit maximisers and risk neutral. The abatement cost function (derived in the Chapter 5) of each farm type is known to the social planner. Farm types are denoted as j. $j=(1,2)$, where 1 is intensive in production and 2 is moderate in production. Since locally estimated transaction/monitoring cost values are not available, values relating to transaction costs were obtained from elsewhere. In practice, very few studies have attempted to quantify transaction costs. Establishing farm nitrogen standards initially involved information costs such as determining farm physical and production characteristics. Simulation models for nitrogen discharges were calibrated at 72 hours per farm, and annual monitoring and updating was estimated at 12 hours per farm (Carpentier, Bosch, & Batie, 1998). Travel costs were also included, as were the different hourly rates for professionals such as technicians, an agronomic expert and a lawyer. Given the availability of GIS databases, mobile computers, online communication and simulation models with guided user interfaces, it is reasonable to assume that the hours required per farm could be less than those estimated by Carpentier at al (1998). Initial information acquisition and preparation of contracts would cost 40 hrs per farm, and this represents the fixed component of the monitoring cost (m_1). Variable components include hours spent on periodic monitoring. The total monitoring function is considered as linear ($m_1+m_2(e)$). Effort (e) is assumed to be 5 hours per farm on average. This assumption is reasonable, given the complex nature of input-output relationships, and the 2.5 -4.0 hours [7] that is required for compliance monitoring of dairy farm effluent disposal. A monotonic relationship is assumed between the level of effort and the probability of detection. i.e 1 hour of monitoring results in a 0.1 probability of detection. Monitoring effort is independent of intensity and farm size, and the cost of monitoring is calculated at $80 per hour. This is on a par with the estimated values for monitoring dairy farms by the Tasman District Council. The parameter values and functional forms chosen are presented in Table 6.4. These are merely illustrative as the purpose is to show relative non compliance rather than absolute magnitudes. Choosing appropriate tax and penalty rates to reach the targeted discharge level are not addressed here.

The simulation problem is set up in a spreadsheet. An abatement cost function is used to derive the cost of reducing nitrogen discharge from the baseline to 20%. Tax on excess nitrogen discharges above the target is based on a figure of $5 per kg. The true discharge levels and discharge levels for reporting under strategic behaviour are found under different policies using Excel Solver. Results are presented in Table 6.5.

---

[7] Tasman District Council (http://www.tdc.govt.nz/)

Table 6.4 Simulation parameters for monitoring

Variables	Intensive system	Moderate system
Baseline Scenario		
Profit($/ha)	2825.00	1392.00
Nitrogen discharges	60.00	33.00
Policy Scenario		
Required level of reduction (Kg/ha)	12	6.6
Abatement cost function($/ha) *	$0.54 * A^2$	$2.16 * A^2$
Penalty function (θ)	ϕv_i	
Monitoring cost function	*m1+m2(e)*	
Tax on excess discharge (τ)	5.00 ($/kg)	
Probability of monitoring (α)	0.4 -0.6	
Level of penalty for false reporting (θ)	5.00-8.00 ($/kg)	

* Abatement cost coefficients are estimated in the Chapter 5.

Table 6.5 Simulation results for monitoring

α	τ	θ	Expected income	Nitrogen discharge (Kg/ha)		Cost of monitoring ($/farm)
				True	Reported	
				Intensive system		
0.4	5	5	2,791	55	48	3,520
0.6	5	5	2,777	55	55	3,680
0.4	5	8	2,777	55	55	3,520
				Moderate system		
0.4	5	5	1,367	32	26	3,520
0.6	5	5	1,362	32	32	3,680
0.4	5	8	1,362	32	32	3,520

The results show that the effectiveness of policy performance depends on compliance monitoring, reporting and penalties for rule violations (Table 6.5). Farms can be urged to report truthfully. Achieving equality between true and reported discharge levels has been possible by adjusting the penalty, level of monitoring and tax on excess discharges. However, in practice level of reporting can be smaller than that of simulated reporting levels. This can be attributed to monitoring limitations (Ozanne, Hogan, & Colman, 2001)

Determining which type of target is appropriate is dependent on the social planner's monitoring budget, and their ability to use different levels of penalty for different farming systems. However, targeting intensive farms is cost effective in terms of abatement cost and there is a higher expectation of truthful behaviour. Macho-Stadler & Perez-Castrillo (2006) also suggest that it is optimal to devote resources to those farms that place a smaller value on reducing discharges. McCann & Easter (2000) also provide empirical evidence of a positive relationship between abatement cost (*AC*) and transaction cost (*TC*). Their estimated relationship is $Ln\ TC = 0.010 + 1.464AC$. The intercept term indicates the fixed component of the transaction costs. Targeting farms with lower abatement costs is likely to reduce the cost of transactions.

Even though the level of monitoring is assumed to be exogenous to the farm's decision in the empirical analysis, in reality the perceived probability is influenced by the farm's own actions. It would be reasonable to think that the probability of monitoring is negatively related to degree of truthful reporting. Thus it is likely that frequent inspections would diminish with the history of truthful behavior.

In fact the impact of targeted monitoring on non compliance can be larger at a catchment scale where many more farms are involved. The empirical analysis that focused on two types of farms can be extended to a greater number of farm types. It is likely that farms with similar geo physical characteristics and proximity to water body are geographically clustered together, and targeting them as a group can save time and monitoring costs.

The existence of different attitudes to risk should not be ignored when designing environmental policy. Moral hazard is likely to be more significant where risk aversion is low. However, modelling farm behaviour to take into account policy risk and production risk increases the complications. For instance, a farmer can be risk averse in responding to a policy contract but at the same time be a risk taker in the production environment, willing to increase the input use that results in more pollution.

6.7 Conclusion

This chapter has addressed the challenges of implementing environmental policy and has suggested the ways of handling it. The implementation of environmental policy is facilitated by exclusive property rights, which are enforceable and transferable among individual farms. Developing scientifically and legally defensible data collection methods and design of surrogate measures for discharges are critical in defining property rights. Empirical applications show how to deter moral hazard arising from hidden actions. Model simulations illustrate that targeting leads to more efficient budget allocation and hence greater reduction in pollution.

Using a production-nitrogen discharge function has the potential to reduce information asymmetry, thus the cost of monitoring. Targeting farms for monitoring improve the abatement and reduce transaction costs. Information about individual farms with high abatement costs is valuable as it helps to preclude high cost farms to reduces the potential problem of adverse selection and moral hazard. Therefore an environmental agency can chose a low level of monitoring and even higher level of financial and technological support. The analytical framework of this chapter can be useful for emission trading schemes to set permit prices based on right penalties determined through iterative process.

This study assumes nitrogen discharges are deterministic. In most circumstances, discharge levels and consequent damages depend on stochastic environmental factors. In particular, variability in rainfall over time can have a profound effect on discharges as well as on the consequent pollution transport. However, the water quality consequences of increased nonpoint source discharges after high rainfall may be diluted by the very weather events that created the enhanced loads (Stephenson, Norris, & Shabman, 1998). Given this, it is justifiable to adopt a deterministic approach based on nitrogen discharges calculated on average rainfall.

One alternative to targeting farms is to develop a menu of contracts with differentiated tax and abatement levels designed to elicit truthful responses from farmers. With *a priori* information on the profit pollution function for different farming systems and derived abatement costs, a social planner can devise contracts with varying levels of abatement, tax and penalty that would make polluters self select.

It is technically possible to target farms according to their pollution potential, but in political terms the use of different penalties is a contentious issue. Since farming has several public benefits such as employment and income generation, it is desirable to investigate ways of compensating farmers who reduce farming intensity or refrain from expanding their activities. It may be practicable to use financial incentives in conjunction with penalty schemes to offset production losses. Further compliance monitoring can be complemented through technological subsidies such as extended extension services to estimate nitrogen discharges through nutrient budgets.

Given major institutional changes a probable analysis should carefully consider distributional issues as politicians will place considerable emphasis on distributional issues. The evolution of property rights is driven by an ongoing search for ways to internalise externalities. Institutional arrangements need to evolve to empower people to take abatement measures for nitrogen discharges. The diversity of risk preferences should not be ignored in designing environmental policy.

7. Environmental and economic efficiency of dairy farms

7.1 Introduction

Increasing agricultural productivity has been a policy objective in New Zealand, but higher productivity has been accompanied by higher input use, creating negative externalities are described in chapter 3. Therefore measuring the environmental performance of dairy farms and integrating this information into farm productivity calculations is important for informed policy decisions which promote sustainable development. Incorporating farm nitrogen discharges into farm production measures helps to identify farms which are efficient both economically and environmentally. In this chapter such measures are described as environmental-economic efficiency measures. Efficient farms can be used to benchmark progress and help in the design of policy that promotes farm efficiency. These measures can also be used for monitoring and evaluating farms. To date, analysis of dairy farm performance in New Zealand has ignored undesirable effects on the environment, and it is difficult to source the data required to measure the environmental-economic efficiency of farms. Relevant data would need to cover farm management, economic information, spatial characteristics and nitrogen discharges. Given this, the spatially micro-simulated virtual population data generated in chapter 4 is used for analysis.

This chapter focuses on the environmental and economic performance of farms in relation to the best performance in the industry. These measures are used to identify the reasons for differences in performance and to provide options for improvement. The methodology used in this study is a two stage process which first involves solving a data envelopment analysis (DEA) problem. In the second stage, the efficiency scores from the first stage are regressed on other explanatory variables.

Section 2 of this chapter describes the methods used to measure efficiency. Section 3 looks at DEA. Section 4 reviews conventional efficiency measures in the DEA framework. Section 5 reviews in detail ways to define environmental efficiency. Section 6 discusses the challenges in measuring environmental efficiency in the New

Zealand dairy farming context and proposes modifications needed for empirical application. Section 7 discusses the analysis of variations in environmental efficiency. Section 8 describes the data, and Section 9 presents a discussion of the results, along with a summary of the findings.

7.2 Methods of efficiency measurement

Efficiency measurement requires the construction of a frontier. The two popular methods of doing this employ econometrics or mathematical programming. Generally used econometric and mathematical programming approaches are stochastic frontier analysis (SFA) and data envelopment analysis (DEA) respectively. SFA requires the selection of an appropriate functional form and choice of a distribution for the inefficiency scores. It is time consuming and suited only for single-output technologies. A multi-output case can be studied only if the various outputs can be aggregated into a single output (Coelli, Rao, O'Donnell, & Battese, 2005). However, some environmental performance analysis involves the handling of multiple outputs. Asmild & Hougaard (2006) state that DEA's ability to handle multiple inputs and outputs in different units, is ideal for measuring environmental performance. The DEA approach does not require the assumption of functional form to specify the relationship between inputs and outputs and the distributional assumption of the inefficiency term. This avoids unnecessary restrictions about functional form, which are likely to distort efficiency measures (Coelli, 1995; Fraser & Cordina, 1999). DEA also allows comparison of one farm with another in terms of a performance index. The DEA approach can, however, be criticised for not accounting for the possible influence of measurement error and other noise in the data (Coelli, Rao, O'Donnell, & Battese, 2005). The main strength of the stochastic frontier approach is that it deals with stochastic noise. Simar & Wilson (2000) also provided statistical foundations for DEA estimates using bootstrap methods. Since a virtual population of farms is used to construct the frontier in this study, it is not necessary to consider sampling variability - the data can be considered to be noise free. In fact, in this study efficiency is measured rather than estimated. DEA is used in this chapter because of its flexibility and helpful features.

DEA has been used in many studies to analyse environment oriented efficiencies (Coelli, Lauwers, & Van Huylenbroeck, 2007; Fare, Grosskopf, & Pasurka Jr, 2007; Tyteca, 1996; Wossink & Denaux, 2006). It has also been used to analyse agricultural productivity in New Zealand (Jaforullah & Premachandra, 2003; Jaforullah & Whiteman, 1999; Neal, 2004). None of these studies, however, has attempted to measure environmental performance. Jaforullah & Whiteman (1999) used a sample of New Zealand-wide data to estimate efficiency, but given the large variations in weather and soil across the country, the reliability of their estimates is open to question (Fraser & Cordina, (1999). The modelling approach of this chapter incorporates geophysical variables when measuring efficiencies.

7.3 Data Envelopment Analysis

DEA involves mathematical linear programming. It constructs a non parametric piecewise surface or frontier over the data, by enveloping the data points of the observed best practice activities. Efficiency measures are then derived from the distance of each individual observation to that frontier. The measure provides a score for each farm from 0 (worst performance) to 1 (best performance). The DEA model developed by Charnes, Cooper, & Rhodes in (1978) was input oriented and assumed constant returns to scale (CRS). There have been numerous subsequent developments such as allowing for variable returns to scale (VRS) (Banker, Charnes, & Cooper, 1984) In the DEA literature individuals are often referred to as decision making units (DMU). In the context of this chapter, the farm is used to denote DMUs. The following subsection briefly discusses popular DEA models and specifications which have been widely employed in applied analysis. Details of DEA model specification are discussed in Coelli, Rao, O'Donnell, & Battese (2005) and Cooper, Seiford, & Tone (2000).

7.3.1 DEA specifications

Solving DEA requires the specification of orientation, returns to scale and weights. Weights are endogenously determined by the algorithm. This section provides a specification of DEA for measuring efficiency.

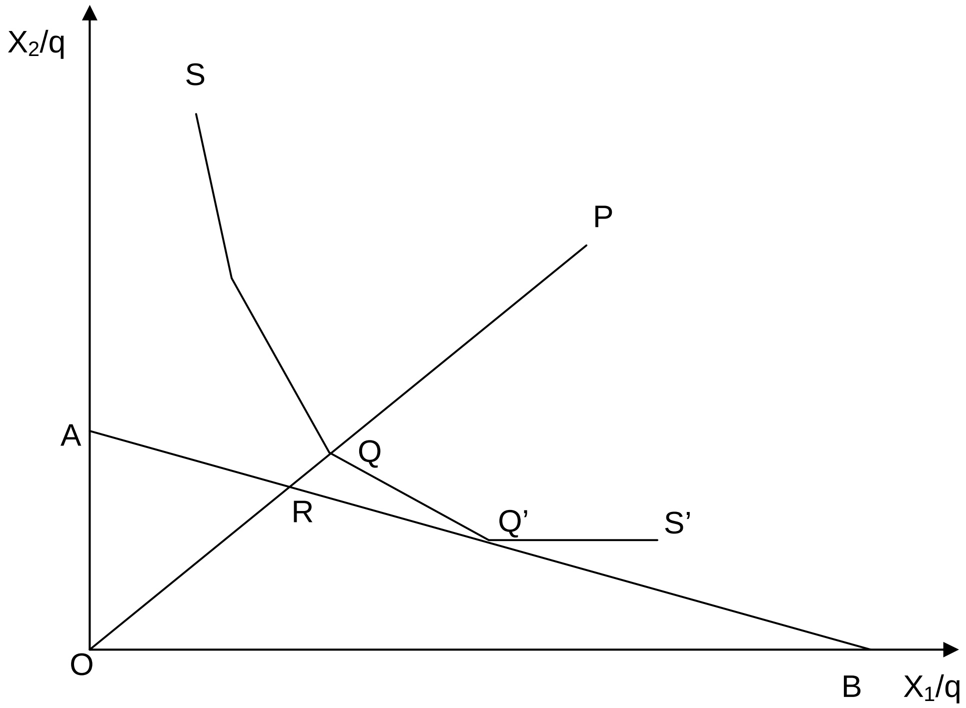
Input and output orientations are adopted in DEA to measure efficiency. Input orientation focuses on the extent to which inputs can be reduced while maintaining the existing level of output. Output orientation focuses on the extent to which output can be increased with a given level of input. In an agricultural context either input or output orientation can be used, depending on the application. For instance, if output production is restricted through quotas, then input orientation is preferred. On the other hand, output orientation is applicable in situations where it is possible to enhance production using available resources. Output oriented models are very much in the spirit of neo-classical production functions defined as the maximum achievable output given input quantities (Fare, Grosskopf, & Lowell, 1994).

Efficiency estimates are insensitive to orientation under CRS. However under VRS these estimates are not similar (Coeli et al, 2005), so it is important to choose the appropriate orientation. CRS assumes that all farms operate at an optimal scale, but financial, land and other constraints mean that in reality they will operate at different scales. VRS specification allows a calculation of efficiency devoid of these scale effects. The VRS model may be more relevant for measuring environmental efficiency because CRS assumes that a doubling of all inputs would result in doubling of all outputs, which is obviously not the case with nitrogen discharge into water. Since the convexity assumption of VRS about production possibility set is weak, it results in very cautious estimates of improvement potential. A study by Jaforullah & Devlin (1996) states that dairy farms are characterised by constant returns to scale, and results from Neal (2004) suggest that the majority of farmers are operating in areas of increasing returns to scale or constant returns to scale. Under CRS technology, the derived efficiency scores are either less than or equal to the VRS technology efficiency scores. The details of these returns to scale are discussed in the following section.

7.4 Measuring efficiency

The basic approach to measuring farm level efficiency is to estimate a frontier that envelops all the input and output data, with those observations lying on the frontier being described as efficient. Any farm that deviates from the frontier is considered to be inefficient. Inefficient farms can either reduce their input use while maintaining output level, or use the same level of input to increase the output. The concept of technical efficiency is the basis for deriving other types of efficiencies.

Figure 7.1 Technical and allocative efficiencies



7.4.1 Technical efficiency (*TE*)

Technical efficiency is defined as the ability of a farm to produce either the maximum possible output from a given bundle of inputs and a given technology, or to produce a given level of output from the minimum amount of inputs for a given technology. Technical inefficiency would require a farm to reduce inputs without reducing outputs or to increase output from same amount of input.

Figure 7.1 explains the concept of *TE* of input orientation. SS' is the frontier, which indicates an efficient combination of inputs for a certain amount of output. A farm which uses P amount of inputs is said to be inefficient, because all inputs could be reduced proportionally without any reduction in output. *TE* =0Q/0P. *TE* is equal to one for technically efficient farms.

The constant returns to scale input orientation

A constant return to scale input oriented model is used to explain the concept of technical efficiency. The following notations are used to describe the model. Each farm is assumed to have *K* inputs and *M* outputs. The total number of farms is *N*. Outputs of each (j^{th}) farm are described by a column vector of outputs (q_i). Inputs of each farm are described by column vector (x_i). The entire data set consists of $K * N$ input matrix of X and $M * N$ output matrix of *Q*. The feasible production set is defined as

$$q \in R_{+}^{M}\text{ , } x \in R_{+}^{K}\text{ ,}$$
$$T = \{(q,x) \in R_{+}^{M+K} \mid x \text{ Produces } q\} \qquad (1)$$

where the production technology is assumed to be convex and non-increasing in input, non decreasing in outputs and exhibits strong disposability in inputs and outputs. If the observed input vector is not technically efficient (i.e. not on the isoquant), technical efficiency can be reached by proportionally shrinking the input combinations until it is projected onto the boundary of the technology set. The boundary of the technology set is denoted by the isoquant. The purpose of DEA is to obtain a measure of efficiency as a ratio of a weighted sum of outputs divided by a weighted sum of inputs. Using the above notation results in:

$$\frac{u' q_i}{v' x_i} \qquad (2)$$

where *u* is a $M * I$ vector of output weights and *v* is $K * I$ vector of input weights. The optimal weights (*u* & *v*) are obtained by solving the mathematical programming model, which maximises the efficiency measure of the j^{th} farm.

Max $\frac{u^{'}q_i}{v^{'}x_i}$

Subject to

$u^{'}q_j / v^{'}x_j \leq 1$ *j=1,2,.....N* (3)

$u, v \geq 0$

u and *v* are solved so that the measure of efficiency for the *i*th farm is maximized, subject to the constraint that all efficiency measures must be less than or equal to one. However, the above formulation yields infinite solutions, so the problem is reformulated as follows:

$Max_{\mu,v}(\mu^{'}q_i)$

Subject to

$\zeta^{'}x_i = 1$

$\mu^{'}q_j - \zeta^{'}x_j \leq 0$ $j = 1,2,.....N$ (4)

$\mu, \zeta \geq 0$

Here the notation has been changed from u and v to μ and ζ.

Using duality theory, an envelopment form of the previous expression is derived.

$Min_{\theta,\lambda}\theta_j$

subject to

$-q_i + Q\lambda \geq 0$

$\theta x_i - X\lambda \geq 0$

$\lambda \geq 0$ (5)

* $Q\lambda$ can be elaborated as $\sum_{j=1}^{N} q_{ij}\lambda_j$ and $X\lambda$ can be elaborated as $\sum_{j=1}^{N} x_{ij}\lambda_j$

θ_j, is a scalar and λ is a *N * 1* vector of constants. The estimated value of θ is the efficiency score for each of *N* farms. The estimate will satisfy the restriction θ<=1 with the value θ_j =1 indicating the efficient farms. In other words, it is possible to reduce the input use of farms by (1- θ_j). To derive a set of N efficiency scores, the problem needs to be solved *N* times, once for each farm. The problem specified above takes the j^{th} farm and then seeks to radially contract input vector x_i as much as possible, while still remaining within the feasible input set. The inner boundary of this set is a piecewise linear isoquant determined by observed data points. The radial contraction of the input vector x_i produces a projected point (Xλ, Qλ) on the surface of the technology. These projected points are a linear combination of the observed data points. The constraints ensure that the projected points cannot lie outside the feasible set. This envelopment form is often used in empirical estimation because fewer constraints are involved than in the multiplier form.

The variable return to scale input orientation model

The mathematical programming for VRS is as follows:

$Min_{\theta,\lambda}\theta_j$

subject to

$-q_i + Q\lambda \geq 0$

$\theta x_i - X\lambda \geq 0$

$\lambda \geq 0$

$N1'\lambda = 1$ (6)

The convexity constraint introduced in equation $N1'\lambda = 1$ ensures that farms are benchmarked against farms of similar size. This approach forms a convex hull of intersecting planes that envelop the data points more tightly than with the CRS conical hull. Therefore efficiency scores derived under VRS formulation are greater than or equal to those obtained under CRS. By allowing for VRS, the measure of technical efficiency can be decomposed into pure technical efficiency and scale efficiency (*SE*) (Fare, Grosskopf , & Lovell, 1985).

$$SE = TE_{CRS} / TE_{VRS} \tag{7}$$

$SE = 1$ indicates scale efficiency. $SE < 1$ indicates scale inefficiency. Scale inefficiency can be due to increasing or decreasing returns to scale, and it can be determined by looking at the sum of weights. If the sum is less than 1 then returns to scale are increasing. A sum greater than 1 indicates decreasing returns to scale.

The variable returns to scale output orientation model

The output oriented model specification is very similar to the input specification. The formal mathematical representation is as follows.

$$Max_{\phi,\lambda}\phi_j$$

subject to

$$-\phi q_i + Q\lambda \geq 0$$

$$x_i - X\lambda \geq 0$$

$$\lambda \geq 0$$

$$N1'\lambda = 1 \tag{8}$$

where $1 \leq \phi < \infty \, and \, \phi - 1$ measures the proportional increase in outputs that can be achieved by the j^{th} farm, with input held constant. The reciprocal of this measure $(1/\phi)$ yields an estimate that take values between 0 and 1 (technical efficiency). Significantly, the output and input orientations yield exactly the same set of efficient farms, although efficiency estimates differ.

7.4.2 Economic efficiency and allocative efficiency

The input or output orientation discussed above can easily be adopted to derive economic efficiency by using appropriate data (price and cost information). Economic efficiency can be interpreted in terms of cost or revenue or profit efficiency. In Figure 7.1, let c_i represent a $K * 1$ vector of input prices for the j^{th} farm and let x represent the observed vector of inputs used associated with point P. Let x' and x* represent the input vector associated with the technically efficient point Q and cost minimizing input vector at Q' respectively. Then the cost efficiency of j^{th} farm can be defined as the ratio of input costs associated with input vectors x and x*, associated with points, P and Q'. i.e CE=0R/0P. Thus cost efficiency is defined as the ratio of the minimum cost to observed cost. If $CE_j = c_i^{'} x_i^{*} / c_i^{'} x_i$ If $CE_j = 1$, then farm *j* is considered to be economically efficient. Economic efficiency comprises technical efficiency and allocative efficiency. Allocative efficiency on a farm is maximized when marginal value product equals cost. Allocative efficiency (*AE*) is computed as

$$AE = CE / TE \tag{9}$$

It is equal to 0R/0Q.

TE is technical efficiency, calculated by the input oriented measure of DEA mentioned in Equation 6**.**

The following mathematical formulation describes the concept of economic efficiency in terms of cost minimization:.

$$Min_{\lambda, x_i^*} c_i x_i^*$$

subject to

$$-q_i + Q\lambda \geq 0 \,;$$

$$x_i^* - X\lambda \geq 0$$

$$\lambda \geq 0$$

$$N1^{'} \lambda = 1 \tag{10}$$

The economic efficiency concept can be applied to calculate revenue efficiency and profit efficiency in the output oriented framework. To calculate revenue efficiency, revenue is maximized for given output prices and input levels. The total revenue efficiency (*RE*) of the j^{th} farm is calculated as the observed revenue to maximum revenue.

Profit efficiency is calculated by maximizing the difference between revenue and cost (i.e minimum cost and maximum revenue). Profit efficiency (*PE*) is measured as the ratio of observed profit over maximum potential profit. All these efficiency measures (*CE, RE* and associated allocative and technical efficiencies) can take values ranging from 0 to 1, where the value of 1 indicates perfect efficiency. However, profit efficiency measures need not be bounded by 0 and 1. These could be negative if profits are negative, or it could be undefined if maximum profit is 0. Technical efficiency is usually measured with output orientation in order to derive allocative efficiency in profit maximization and revenue maximization scenarios.

7.5 Measuring environmental performance

The current emphasis on environmental issues has led dairy farms to consider improvements in environmental performance as well as in productivity, and policy makers are interested in any actions that can be taken to achieve this. The process around measuring performance is consequently receiving greater attention.

The incorporation of environmental impact information into production process analysis provides an opportunity to measure environmental performance and changes in performance under environmental constraints (Tyteca, 1997). A generally used technique is to look at one area of environmental efficiency, such as achieving the highest feasible ratio of milk solids to nitrogen discharge. Different environmental performance measures which are used in DEA, are discussed at greater detail below.

There are difficulties associated with incorporating environmental impacts into the analysis of environmental performance. In the literature environmental effects are brought into the model as either undesirable outputs or undesirable inputs. In recent literature two novel approaches have been adopted. One approach uses the concept of nutrient budgeting to derive environmental efficiency in agricultural applications. In the second approach, enhancing the nutrient content of the output is modelled as a means of minimizing nutrient into environment. These environmental efficiency measures are often reported along with the conventional efficiency measures discussed above. Like conventional efficiency measures, environmental efficiency also has different orientations. Environmental efficiency is modelled as an input or output, or both input and output, or by using the concept of material balance. The following subsections review various environmental efficiency measures.

7.5.1 Environmental impact as an input or output

Some studies model environmental impact as an undesirable output. Tyteca (1996) claims that an environmental performance indicator can be calculated as the ratio between the overall productivity measure (using both desirable and undesirable outputs), and the gross productivity index where undesirable outputs are ignored. It involves comparison of technical efficiency measures, with and without environmental effects. Fare, Grosskopf, & Tyteca (1996) estimate environmental efficiency by running two input oriented DEA models for each decision making unit. In the first run, technical efficiency is estimated, allowing for the conventional proportional contraction of all inputs. In the second run the environmental impact is incorporated as a weakly disposable output[1]. The environmental efficiency is then estimated as the ratio of the efficiency score obtained with the first run over the score obtained in the second run. The indicator takes a value less than or equal to one. Fare, Grosskopf, & Pasurka (2007) model environmental impact as a bad output in a joint production approach using DEA on U.S coal fired electric plants, in which good and bad outputs are assumed to be jointly and weakly disposable. This study models traditional productivity, with and without regulating the production of bad outputs, to establish a relationship between traditional productivity and environmental regulation.

Environmental impact has been modelled as an input in some studies (De Koeijer, Wossink, Struik, & Renkema, 2002; Ondersteijn, Lansink, Giesen, & Huirne, 2002; Reinhard, Lovell, & Thijssen, 1999; Wossink & Denaux, 2006). There are a number of ways to denote input oriented technical and environmental efficiency under

---

[1] Strong disposability implies that there is no charge for disposing of unwanted input or output. Weak disposability concept means costly disposal of nitrogen surplus. Further details are available in Coelli et al (2005) pages 195-197.

conditions of variable returns to scale. Treating environmental impact as an input provides separate estimates of technical efficiency and environmental efficiency. In the agricultural context Reinhard, Lovell, & Thijssen (2000) take intensive farms in the Netherlands to model nitrogen surplus as an environmentally detrimental input to calculate technical efficiency and environmental efficiency using the DEA framework. They specify nitrogen surplus as a proxy for pollution and modelled as an additional input variable.

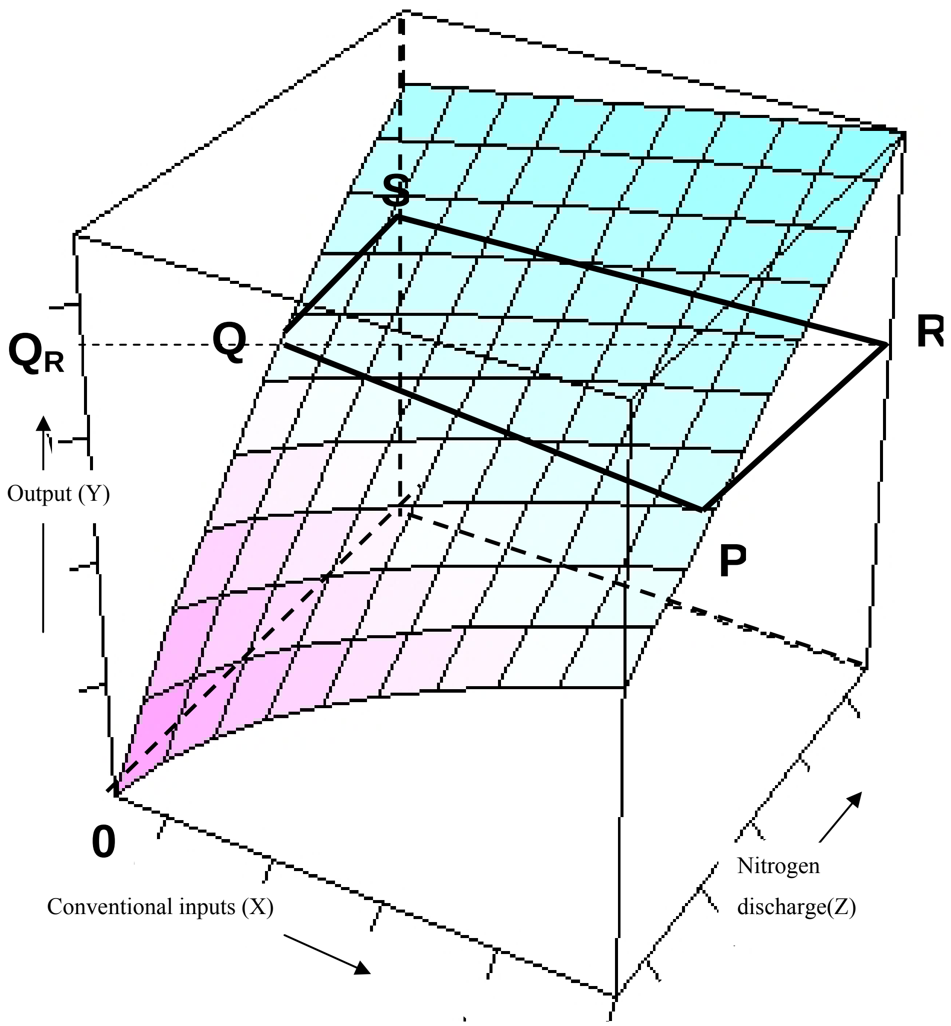
Figure 7.2 presents the best practice production frontier *f(.)* with output Q, conventional input *X* and nitrogen discharge *Z*, Q<=*f(X, Z*). The frontier *f(.)* is assumed to be increasing and concave in surface. *PQRS* is a surface with identical quantities of output for various combinations of conventional inputs and nitrogen discharges.

The output oriented measure is expressed as the ratio of maximum feasible output to the observed level of output, conditional on specific levels of conventional input and pollution variables. The output oriented technical efficiency in environmental models is illustrated on Figure 7.3. The output oriented model allows for radial expansion of the output while holding conventional and pollution inputs fixed. Output *Q* is measured on the vertical axis and the combination of conventional input (X) and pollution variable (Z) are shown on the horizontal axis. The production frontier shows the maximum feasible output achieved from a given level of combined input/ discharge. Looking at point R, the frontier (point C) can be reached without any increase in input use or nitrogen discharge. The output oriented environmental efficiency EE^o is measured by the equation 11.

$$EE^o = \max\{\Phi : \Phi Q_R \leq F(X_A, Z_A)\}^{-1} = |DQ_A| / |DQ_C| \qquad (11)$$

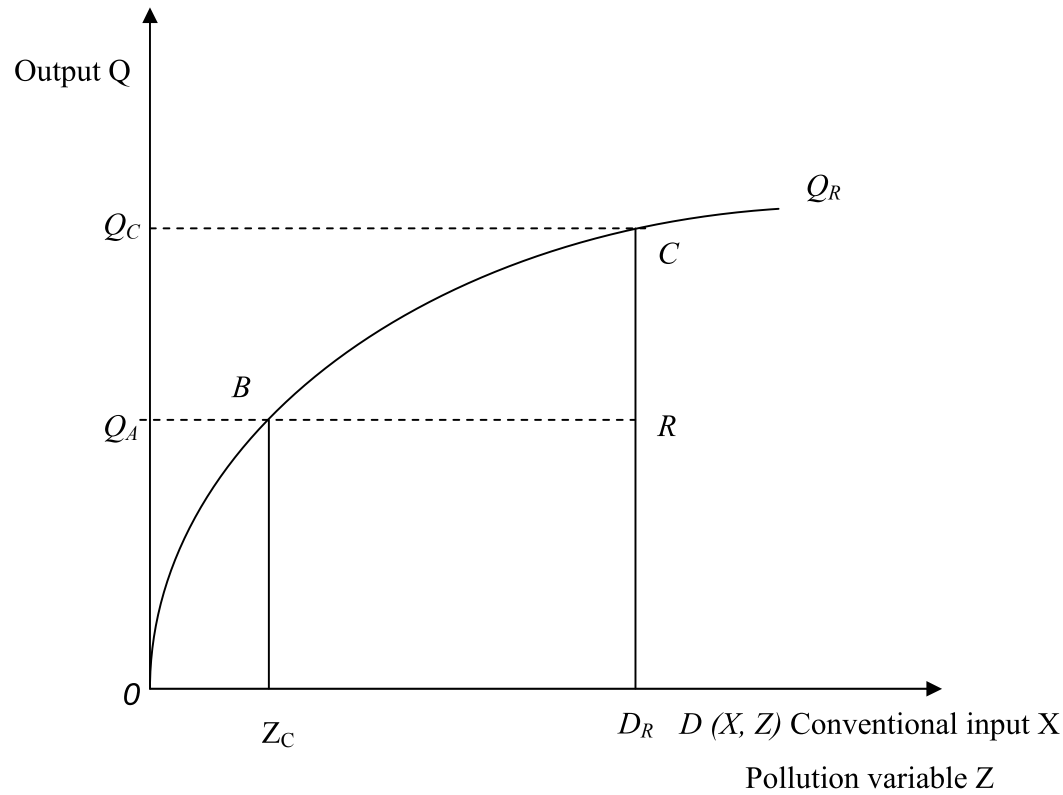
Q_R is the maximum feasible output given both conventional input and nitrogen discharge.

Figure 7.2 Environmental-production frontier surface



Reinhard et al (2000) defined environmental efficiency with input orientation as "the ratio of minimum feasible to observed use of an environmentally detrimental input, conditional on observed levels of the desirable output and the conventional inputs". Input oriented efficiency involves a radial contraction of inputs inclusive of pollution, while holding output constant. Figure 7.4 is used to illustrate an input oriented production frontier. Conventional input *X* is measured by the vertical axis. Nitrogen discharge *(Z)* is measured on the horizontal axis. The isoquant of the fixed level of production *(Q_R)* shows various combinations of conventional input and nitrogen discharge. Environmental efficiency is provided by the radial input oriented measures. The point *R* is the observed combination of conventional (OX_R) and nitrogen discharge (OZ_R). The optimal combination of conventional input and nitrogen discharge occurs at point *B*. The optimal adjustment occurs along the ray *OR* from *R* to *O*. The movement along the ray *OR* involves an equiproportionate reduction of conventional input and nitrogen discharge. The radial input oriented measure of environmental efficiency is defined in the equation 12:

Figure 7.3 Output oriented production frontier



$$EE_R = \min\{\theta : F(\theta X_R, \theta Z_R) \geq Q_R\} = |OB| / |OR| \qquad (12)$$

X_B and Z_B are the minimum feasible conventional input and nitrogen discharge, given the prevailing production technology and the observed value of the output *YR*.

Ondersteijn, Lansink, Giesen, & Huirne (2002) model technical efficiency as an overall efficiency by incorporating nutrient surpluses as a weakly disposable input.

Weak disposability is incorporated by introducing an equality constraint and a scaling parameter δ (Equation 13).

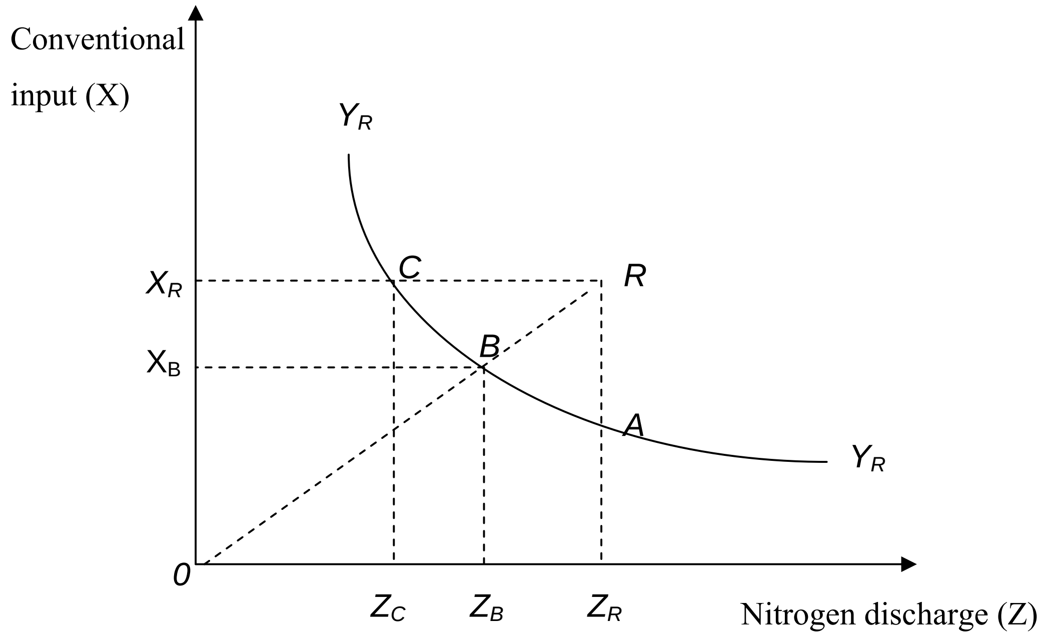
$$
\begin{aligned}
&Min_{\theta,\lambda}\beta \\
&\text{subject to} \\
&-q_i + Q\lambda \geq 0 \\
&\beta x_i - X\lambda \geq 0 \\
&\beta\delta z_i - Z\lambda = 0 \\
&\lambda \geq 0 \\
&N1'\lambda = 1 \qquad (13)
\end{aligned}
$$

They define environmental efficiency in a similar way to Renihard et al (2000), but incorporate the weak disposability concept for nutrient surpluses.

Wossink & Denaux (2006) estimate technical and environmental efficiencies using the conventional framework for input orientation (Equation 6). The model used to assess environmental efficiency is similar to the model for technical efficiency, the only difference being that the expected environmental impact of each input category is used instead of the amount of observed conventional inputs.

It is important to make efficient use of inputs which cause nitrogen discharge. When farms improve the technical efficiency of their use of polluting inputs, economic and environmental efficiency will also be achieved (De Koeijer, Wossink, Struik, & Renkema, 2002).

Figure 7.4 Input oriented production frontier



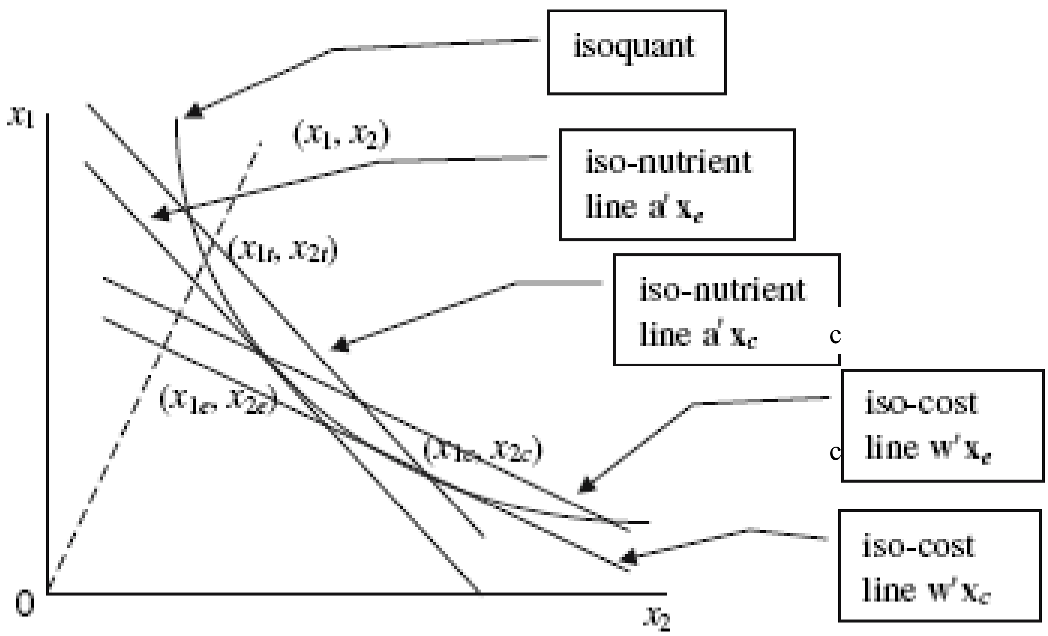
7.5.2 Environmental impact as an input and output

Asmild & Hougaard (2006) measure the environmental performance of Danish pig farms by modelling the estimated nutrient content of inputs and the nutrient content of outputs, together with a number of other economic variables. In this approach environmental efficiency is measured as the ability to minimize nutrient surplus by maximizing nutrient removal. The nutrient surplus is always assumed to be positive, because a deficit leads to depletion of soil nutrient status over time. Asmild & Hougaard use a two step sub-vector DEA approach to quantify efficiency measures. First the combined economic and environmental improvement potential is calculated by incorporating the economic output variable along with the nutrient content of the output (the environmental variable) in the output matrix. Secondly, the economic improvement potential is determined by incorporating only the economic output variables in the output matrix. Thirdly, the environmental improvement potential is calculated by including only environmental variables (nitrogen content of the milk) in the output matrix. Fourthly, the efficiency of economic improvements followed by environmental improvement is reckoned. Finally, the efficiency of environmental improvements followed by economic improvement is calculated.

7.5.3 Material balance concept

The concept of material balance has been applied to measure the environmental efficiency of Belgian pig finishing farms by Coelli, Lauwers, & Van Huylenbroeck (2007). This section discusses their framework in detail as it is both novel and versatile. The authors quantify environmental efficiency by determining the combination of inputs that result in the lowest level of nutrient surplus to produce a specified amount of output. Nutrient surplus (*z*) is simply calculated as a linear function of input and output using the material balance concept (z=*a`x-b`q*). a and b are the nutrient content of inputs and outputs respectively. When output is fixed, nutrient surplus is minimized by decreasing the nutrient content in the inputs. The input vector which involves minimum nutrient is denoted by x_e. The minimum nutrient quantity is $a`\, x_e$. The nutrient quantity of observed input is $a`x$. Figure 7.5 illustrates the concept of nutrient and cost minimization.

Figure 7.5 Cost and nutrient minimization



Source : Coelli et al (2007)

The nutrient content of each input is reflected in the slopes of isonutrient lines denoted by Equation 14. *N* denotes the total nutrient content.

$$N = a_1 x_1 + a_2 x_2 \tag{14}$$

The isonutrient line which passes through the nutrient minimizing points (x_{1e}, x_{2e}) has the lowest intercept, implying that it has the lowest amount of total nutrient (*N*). The environmental efficiency of a farm is defined as the ratio of minimum nutrients over observed nutrients.

$$EE = a`x_e / a`x \tag{15}$$

The environmental efficiency is decomposed into two components: technical efficiency (*TE*) and environmental allocative efficiency (*EAE*). As noted previously, *TE* is measured as production of a given level of output from the minimum amount of inputs. *TE* is indicated by the ratio of the minimum level of inputs to produce to observed levels of input to produce the same. *TE* scores were calculated using either Equation 5 or 6, depending on the orientation.

$$TE = a`x_t / a`x \tag{16}$$

$$EAE = a`x_e / a`x_t \tag{17}$$

All three efficiency measures take values between 0 and 1, with a value of 1 indicating perfect efficiency. The equation 18 denotes the relationship between these three efficiency measures:

$$EE = TE * EAE \tag{18}$$

Cost efficiency can be found given the input prices (c). It can also be decomposed into technical and allocative efficiency. Using the price information, it is possible to establish the cost of the nutrient minimizing input combination (c`$X_e$) and the nutrient content of the cost minimizing input combination (*a`Xc)*. These measures can be used to find the cost associated with moving from a cost minimizing point to a nutrient minimizing point. This could be interpreted as the shadow cost of nutrient reduction. A convergence of the cost and nutrient minimization points can be achieved by using appropriate policy incentives. Given the damage cost of nutrients, a new comprehensive social cost minimizing point can also be found.

7.6 Challenges to measure dairy farm environmental efficiency

The practical applications of environmental efficiency measures described above rely on simple input and output flow. However, the environmental impact of New Zealand dairy farms on water quality is a complex process which depends on climate variability, pasture and cow physiology and geophysical variability. In addition to this, the outdoor, pastoral nature of New Zealand farming means that it is difficult to control input and output flows, particularly of nitrogen.

The approach adopted by Coelli (2007) is elegant with regard to policy analysis, but its real world application would be a challenge for dairy farming in New Zealand. Coelli et al's model was applied to an intensive pig farming system, which is an indoor production activity where nutrient inflows and outflows are highly manageable and there are no uncontrollable environmental effects. In an intensive farming system of monogastric animals like pigs, calculating nutrient surpluses is straightforward, determined by subtracting the nutrients removed with the harvested crops from nutrient input through manure and fertiliser. The nature of extensive dairy farming means that it is not possible to estimate nitrogen surpluses directly, as part of nitrogen input and removal can be attributed to natural processes such as atmospheric nitrogen fixation and denitrification.

In New Zealand dairy farming the clover/atmospheric nitrogen contribution is difficult to control. The contribution of clover nitrogen is dependent on factors that affect the clover growth and persistence of dairy pastures, including climate, soil nitrogen levels, nitrogen fertilizer use, soil fertility, companion species, choice of cultivar, pasture establishment, grazing management, and pests and diseases. As a result, the clover content of pasture changes on a cyclical pattern, and also a low level of nitrogen fertilizer application tends to boost clover nitrogen fixation. Biologically fixed nitrogen in clover plants is converted into various forms and excreted into the soil. Excreted nitrogen is converted to nitrates through ammonification and nitrification. According to reported farmlet trial results (Tillman, 2008) there is an inverse relationship between nitrogen fixation and the addition of nitrogen fertilizer. In addition to this, nutrient surpluses alone do not fully represent water quality damage from farming systems as there are other influences at work as well. These include soil type, topography, animal productivity, climate and winter management (Thomas, Ledgard, & Francis, 2005). Coelli et al (2007) and Asmild & Hougaard (2006) modelled nutrient surpluses rather than environmental impact, and Asmild & Hougaard (2006) added constant atmospheric nitrogen deposit. In Ashmild's approach the combined economic and environmental efficiency likely to suffer due to the dimensionality problem due to additional variable.

In this study farm nitrogen discharges are modelled with the following function:

$$z = f(Fertiliser\ N, Stocking\ Rate, Feed, Soil, Topography) \quad (19)$$

where z- indicates the nitrogen discharge per ha. To estimate nitrogen discharges, the Overseer[2] nutrient budget software is used. In calculating nitrogen discharges, winter management and effluent disposal practices are assumed to be on a par with industry recommendations, and an average rainfall of 1100 mm for the Waikato is used.

The measurement of environmental efficiency in this chapter combines the merits of the efficiency measures described by Renihard et al (2000), Ashmild (2006) and Coelli (2007) relevant to the real world.

Input oriented approaches are useful in situations where the environmental focus is on reducing pollution while maintaining production (Wossink & Denaux, 2006).

---

[2] The Overseer used in this chapter is an improved version (5.3.1). Version 5.2.6 is used in previous chapters.

The mathematical formulation for input oriented *technical efficiency* under variable returns to scale is similar to Equation 8, except that z is the vector nitrogen discharge. This formulation computes input oriented technical efficiency as the ability of a producer to reduce input, including nitrogen discharges, for a given level of output.

$$Min_{\theta,\lambda}\theta$$

subject to

$$-q_j + Q\lambda \geq 0$$

$$\theta x_j - X\lambda \geq 0$$

$$\theta z_j - Z\lambda = 0$$

$$\lambda \geq 0 \qquad (20)$$

Economic efficiency is formulated as the ability to minimize farm expenses (x^*) for a given level of other variables. The mathematical formulation is similar to Equation 9. It is measured as the ratio of minimum cost to observed cost.

$$Min_{\lambda,x_i^*} c_i x_i^*$$

subject to

$$-q_j + Q\lambda \geq 0;$$

$$x_j^* - X\lambda \geq 0$$

$$z_j - Z\lambda = 0$$

$$\lambda \geq 0 \qquad (21)$$

Environmental efficiency is defined as the ratio of minimum nitrogen discharge to observed nitrogen discharge, conditional on observed levels of the desirable output and the conventional inputs. This is achieved by minimizing the nitrogen discharge for a given level of output and other conventional inputs.

$$Min_{\theta,\lambda}\theta$$

subject to

$$-q_j + Q\lambda \geq 0$$

$$x_j - X\lambda \geq 0$$

$$\theta z_j - Z\lambda = 0$$

$$\lambda \geq 0 \qquad (22)$$

Environmental –economic efficiency is modelled as minimizing nitrogen discharge and farm expenses simultaneously, given output level and other inputs. This overcomes the dimensionality problem in Ashmild's approach. Ashmild measured economic and environmental efficiency by incorporating economic output variables along with the nutrient content of the output (environmental variable) in the output matrix. Since the nutrient content of the output is an extra variable, it is likely to suffer from the dimensionality problem as increasing the number of variables inflates the efficiency.

$$Min_{\theta,\lambda}\theta$$

subject to

$$-q_i + Q\lambda \geq 0$$

$$\theta x - X\lambda \geq 0$$

$$\theta z_j - Z\lambda = 0$$

$$\lambda \geq 0 \qquad (23)$$

A two stage process is adopted to model economic improvements and then environmental improvement. In the first stage, economic improvement potential is calculated by maximizing the farm income for a given level of other inputs including nitrogen discharges. Farm income is derived by multiplying milksolids produced by the payout received. The output orientation is used as it is easy to get the estimates for the subsequent stage, where economic efficiency is followed by environmental efficiency.

$$
\begin{aligned}
&Max\ \theta \\
&\text{subject to} \\
&-\theta(p*q_i)+Q\lambda \geq 0 \\
&x - X\lambda \geq 0 \\
&z_i - Z\lambda = 0 \\
&\lambda \geq 0 \qquad (24)
\end{aligned}
$$

Farms are first made economically efficient through multiplying economic output (farm income) by economic efficiency scores. Then in the second step the environmental efficiency is derived using economically efficient output, similar to that specified in Equation 22. Finally, two the step analysis carried out perform environmental improvements followed by economic improvement here farms are first made environmentally efficient by using the environmental efficiency scores. Then in the second step economic efficiency is derived using adjusted environmental output.

The above DEA efficiency measures are calculated using an open source software package, FEAR (Version 1.1) by Wilson (2008). It is implemented on R, which is a language and environment for statistical computing and graphics. The routines included in FEAR 1.1 allow computation of DEA estimates of technical, allocative and overall efficiency while assuming either variable, non increasing, or constant returns to scale.

7.7 Analysis of environmental efficiency variation

Environmental efficiency is affected by many factors such as management, input use, topography, and soil type. Impacts can be captured by various techniques: dividing the farms into distinct groups in terms of natural factors and solving separate DEAs; incorporating variables directly into the DEA formulation as non discretionary variables; and the two stage approach, where the first stage determines the efficiency by DEA and the second stage tries to explain the differences in efficiency between farms by regressing environmental variables on the efficiency estimates (Coelli, Rao, O'Donnell, & Battese, 2005). Direct incorporation of categorical variables into DEA is difficult and requires more complicated mixed integer linear programming models

(Coelli, Rao, O'Donnell, & Battese, 2005). The two stage approach was preferred for a number of reasons: its ability to accommodate multiple continuous and categorical variables; the requirement of no prior assumptions regarding the direction of influence of environmental variable and statistical inference on the influence upon efficiencies; computational convenience and transparency. This approach has been used in many studies (Dhungana, Nuthall, & Nartea, 2004; Hansson & Ohlmer, 2008; Reinhard, Lovell, & Thijssen, 2002; Wossink & Denaux, 2006).

In order to investigate the factors that explain environmental efficiency, the model for DEA efficicency results is calculated in a second stage using a regression. Since DEA efficiency scores are censored at 0 and 1.0, OLS estimation yields inconsistent estimates. Instead of OLS, Tobit regression using the maximum likelihood approach is used.

The explanatory model can then be written as Equation 25:

$$Y^{*} = X\ \beta + \mu \tag{25}$$

where Y is a DEA efficiency score, rescaled between 0 and 100, and used as a dependent variable. X is a vector of independent variables related to farm specific attributes. β is the unknown parameter vector associated with the farm specific attributes, and μ_i is an independently distributed error term assumed to be normally distributed with 0 mean and constant variance, σ^2. Tobit regression is implemented in Stata 10 (StataCorp., 2007). Tobit regression results may be biased in the presence of high correlation between the variables used in the first stage and second stage (Coelli, Rao, O'Donnell, & Battese, 2005; Simar & Wilson, 2007). Simar & Wilson (2007) proposed a bootstrapping technique to overcome this problem within the two stage DEA. However this is not a problem in this study as second stage explanatory variables are not highly correlated with the variables of the first stage analysis.

7.8 Data description

The data used in this study was compiled from the virtual farm population created in chapter 4. It incorporates 210 farms, which represent homogeneous farm types in the whole catchment. Physical and financial farm variables and estimated nitrogen discharges are used for analysis. In the Waikato 90% of farm revenue on average is derived from the sale of milksolids, according to the DairyNZ's Economic Farm Survey for 2003/04 and 2004/05, It is reasonable therefore to treat milksolids as the sole economic output of the farms.

Given the virtual nature of the data, particular care was taken in the selection and definition of variables. Land, building and plant and machinery variables were avoided as they may not be representative of the farms in the catchment. Land prices in particular are influenced by location as well as economic productivity, and variations in plant and machinery are affected by the particular type in use. The economic farm surplus variable was not used, as depreciation, labour, runoff and stock may not be applicable to the virtual population.

The choice of variables has to be limited to avoid the problems of dimensionality that can affect DEA analysis. Due to the nature of the technique the number of model variables may affect DEA results. DEA efficiency rating depends on the number of farms and the number of inputs and outputs specified (Ondersteijn, Lansink, Giesen, & Huirne, 2002). Adding more model variables for a given sample size can yield higher efficiency scores for units in the sample. Defining ten or more inputs result in almost all DMUs being efficient. In the same way, fewer units in a sample for a given number of model variables may lead to higher efficiency scores. Chambers et al (1998) suggested as a rule of thumb that the model specification should include at least three times as many data observations (farms) as variables. However, omitting necessary input or output may lead to misspecification of the production model. Therefore various inputs belonging to the same category and measured in the same physical units have been aggregated. Major types of supplementary feeds were aggregated using the energy content of the major ingredient in terms of Megajoules[3]. Farm expenses are specified by aggregating variable[4] and fixed costs. Farm expenses defined here are on average less than 20 percent of the average farm expenses reported in the Economic Farm Survey of Dairy Farms. This is due to the exclusion of some variables[5] which would have been difficult to assign to farms in a virtual population. For the same reason dairy farm income also excludes other dairy income and net stock income.

The backward elimination procedure described by Chatterjee & Hadi (2006) was implemented in Stata 10 through step wise regression to select variables. Table 7.1 presents summary statistics of the variables for farms used in the efficiency analysis.

Table 7.1 Descriptive statistics of the data used in the efficiency analysis

Variable	Units	Mean	Stdev	Minimum	Maximum
Milksolids	Kg	97,870	52,699	30,891	350,957
Farm size	ha	107	63	26	570
Milking cows	No	284	167	99	1200
Nitrogen discharge*	Kg	4133	2606	836	21090
Farm expenses	$	260,560	141,851	82,607	855,459
Farm income	$	434,541	233,982	137,155	1,558,249

* Nitrogen discharge was estimated using Overseer

In estimating efficiency variation the stockingrate is used. However this takes no account of variations in the breed type[6]. Several variables are hypothesised to affect farm efficiency in the second stage tobit regression. Table 7.2 lists the variables used

[3] Metabolisble energy content of average quality grass silage is 9.5 (MJ/Kg DM). Maize silage 10.3 (MJ/Kg DM)

[4] Variable cost=animal health cost+ farm dairy+ electricity+ supplement costs + fertiliser costs +labour (paid & unpaid); Detail description of farm economic components are provided in Appendix 3.1.

[5] Grazingoff, crop regrassing, run off lease and standing charges of variable costs and other expenses & standing charges of fixed costs.

[6] Comparative stocking rate involves cow live weight as a right measure, which potentially accounts for breed differences.

in the regression analysis. Level of intensification is covered by incorporating information on stocking rate, quantity of maize silage feed per cow and nitrogen fertiliser per ha. Given the relatively small size of the catchment, weather variables across the farms are considered to be constant.

The geophysical environment which is likely to affect the nitrogen discharges is represented by dummies for soil type and topography. These dummy variables categories were merged into larger groups when there were only a small number of observations in a category, and they were similar in terms of nitrogen discharge potential. The market value of cows was used as a proxy for genetic merit and resultant feed conversion efficiency. It was assumed that the market value of stock included only the milking cows.

Table 7.2 Explanatory variables used in Tobit regression

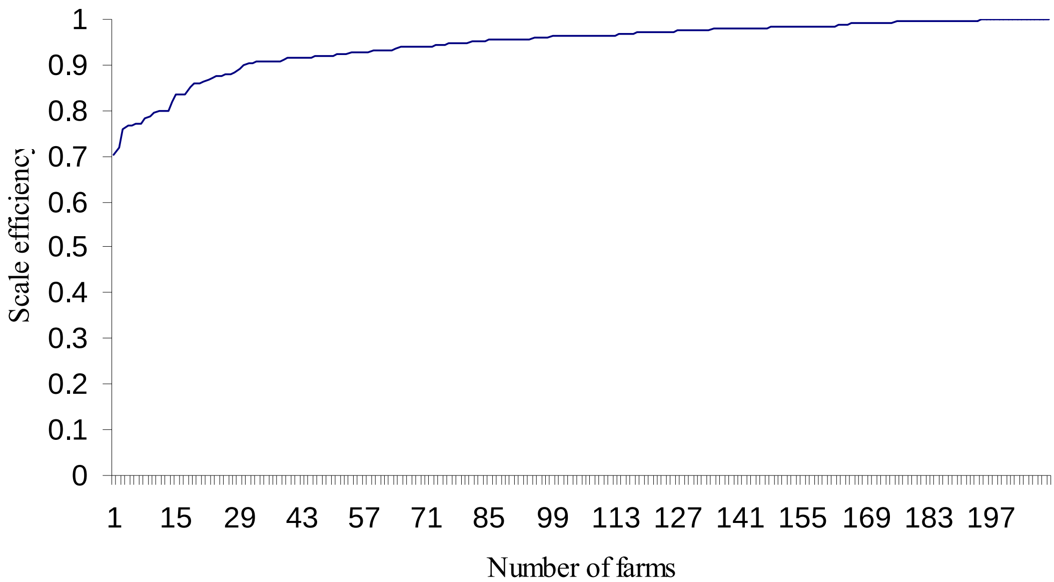
Variable	Units	Mean	Stdev	Minimum	Maximum
Production environment					
Maize silage/cow	tones	0.21	0.29	0.00	1.33
Market value/cow	$	989.00	158.50	491.00	1224.49
Milksolids per cow	No	351.07	45.46	246.37	464.49
Stocking rate	Kg	2.72	0.48	1.80	4.51
Fertiliser nitrogen	$	135.22	64.00	20.00	290.00
Geo physical environment					
Major soil & topographic dummies					
Podzol- rolling					
Volcanic-easy					
Volcanic-rolling					
Pumice_rolling					
Pumice_easy					

7.9 Results and discussion

7.9.1 Efficiency measures

Scale efficiency of farms was examined in terms of technical efficiency. The mean scale efficiency was 0.96, so farms are considered to face constant returns to scale. New Zealand dairy farms are characterised by constant returns to scale in other studies as well (Jaforullah & Devlin, 1996; Jaforullah & Whiteman, 1999; Neal, 2004). Figure 7.6 shows the distribution of scale efficiency of individual farms, with more than 95% of farms being scale efficient. Since CRS measures are insensitive to orientation, it is used to measure subsequent efficiencies. The reciprocal of the output-oriented measure $(1/\phi)$ yields an estimate that takes values between 0 and 1 and equivalent to the input oriented measure. Significantly, the output and input orientations yield exactly the same set of efficient farms, although the efficiency estimates differ. Therefore constant returns to scale are assumed in estimating the final model specification.

Figure 7.6 Distribution of scale efficiency



The distribution of efficiency scores for each specification is shown in Figure 7.7. With respect to technical efficiency, a larger number of farms are in the range of 0.7 to 0.9. In the case of economic efficiency, a larger number of farms are in the range of 0.6 to 0.7, and looking at environmental efficiency a larger number of farms is found to be in the range of 0.4 to 0.7. Figure 7.8 shows the cumulative frequency distributions of the different efficiency measures. Approximately 80 percent of farms achieved less than 80 percent environmental efficiency. In contrast, more than 60 percent of farms achieved more than 80 percent technical efficiency. Environmental-economic efficiency seems to be similar to technical efficiency. Farms are performing better than environmental efficiency in terms of balancing environmental and economic objectives.

Table 7.3 DEA efficiency scores

Efficiency measure	**Efficient farms**	**Mean**	**Stdev**	**Min**	**Max**
Technical efficiency	16	0.82	0.09	0.57	1.00
Economic efficiency	13	0.72	0.13	0.49	1.00
Environmental efficiency	3	0.64	0.12	0.42	1.00
Environmental-economic efficiency	12	0.80	0.11	0.55	1.00
Economic efficiency followed by environmental efficiency	19	0.75	0.10	0.57	1.00
Environmental efficiency followed by economic efficiency	10	0.78	0.08	0.67	1.00
Allocative efficiency	5	0.89	0.19	0.41	1.00

The distribution of efficiency scores for each specification is shown in Figure 7.7. With respect to technical efficiency, a larger number of farms are in the range of 0.7 to 0.9. In the case of economic efficiency, a larger number of farms are in the range of 0.6 to 0.7, and looking at environmental efficiency a larger number of farms is found to be in the range of 0.4 to 0.7. Figure 7.8 shows the cumulative frequency distributions of the different efficiency measures. Approximately 80 percent of farms achieved less than 80 percent environmental efficiency. In contrast, more than 60

percent of farms achieved more than 80 percent technical efficiency. Environmental-economic efficiency seems to be similar to technical efficiency. Farms are performing better than environmental efficiency in terms of balancing environmental and economic objectives.

Figure 7.7 Efficiency estimate histograms

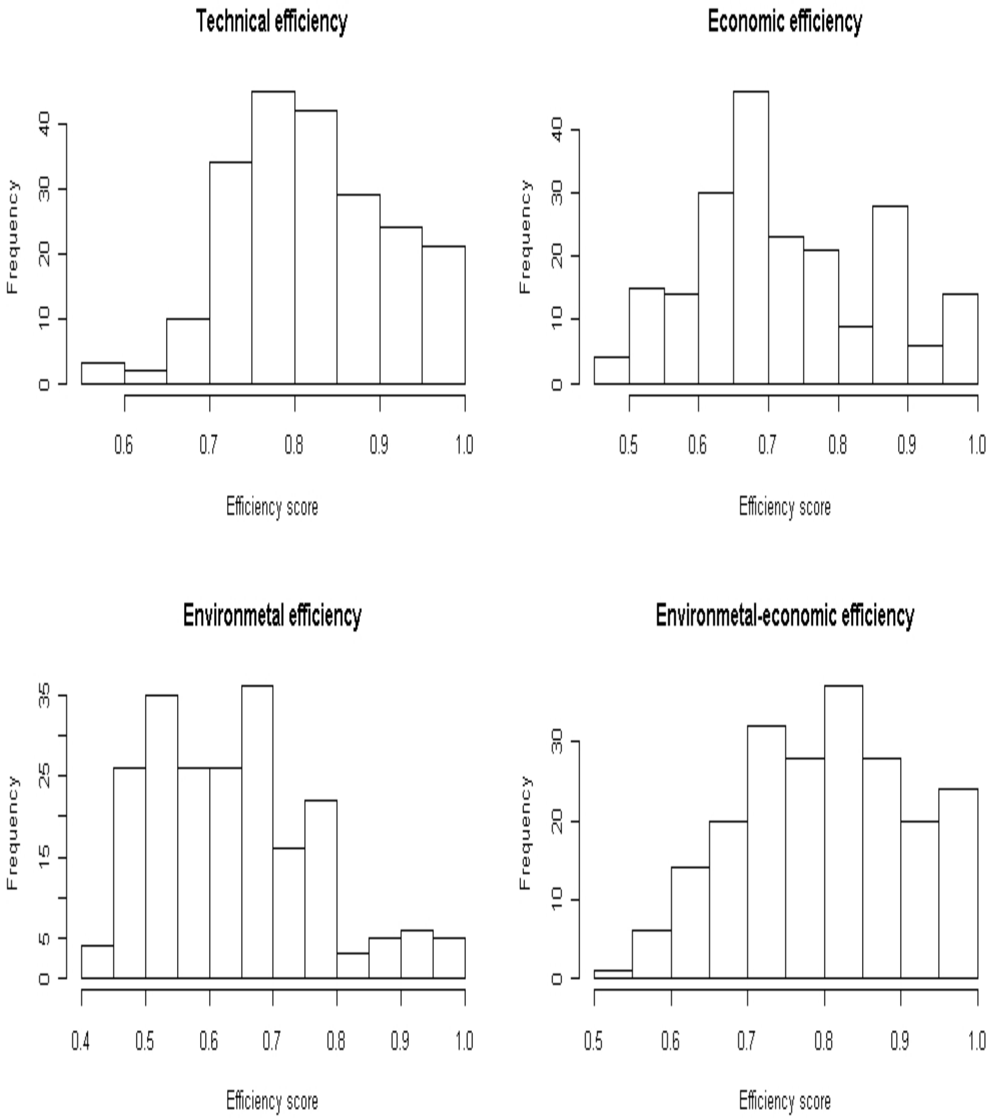
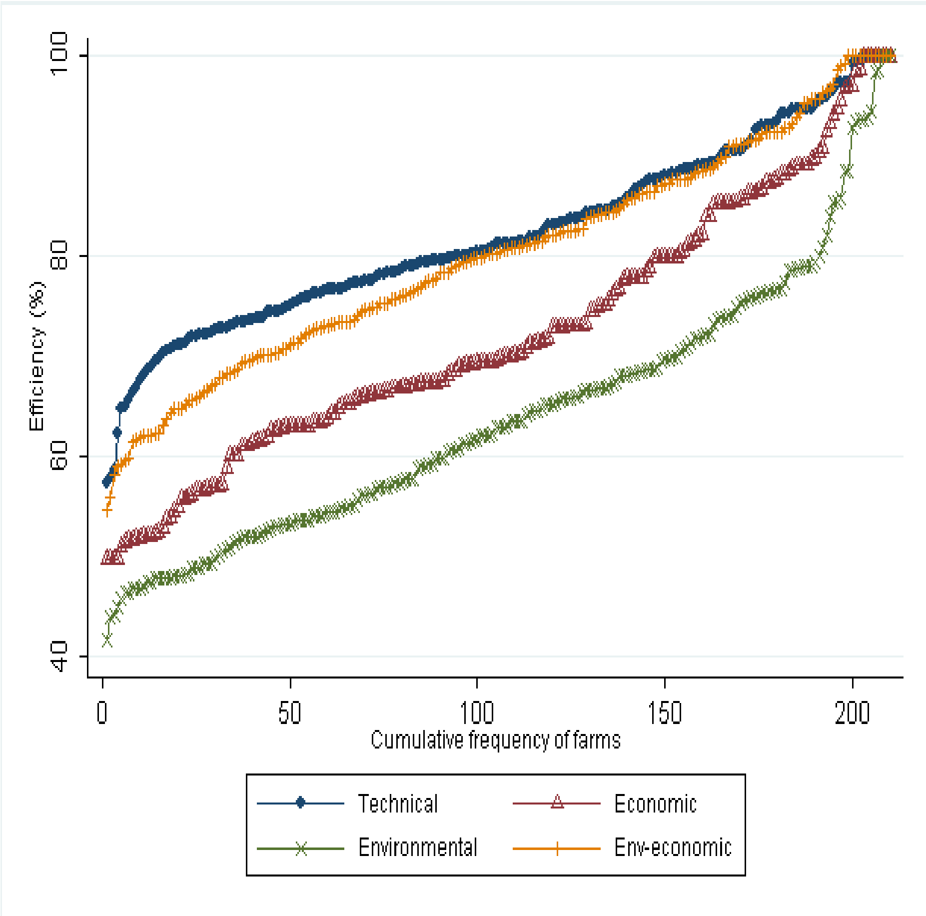


Figure 7.8 Cumulative distribution of efficiency



Efficiency measures are computed according to DEA models specified in Equations 20 to 24. The results are summarized in Table 7.3. Substantial differences are found in efficiencies among farms. The average level of technical efficiency of 0.82 means that in principle the farms can reduce their input use by 1- 0.82 (18 percent) and still maintain the existing level of output. In effect, the level of output can be enhanced by keeping the level of inputs constant. However, the perceptions of risk and the skill level of farmers might have an impact on their ability and desire to achieve this sort of efficiency. The measure of technical efficiency found here is similar to the technical efficiency of dairy farms (0.83) estimated by Jaforullah & Whiteman in 1999.

Mean economic efficiency of 0.72 suggests that the average farm could reduce costs by 28 percent and still produce the same output. This economic efficiency is largely the result of technical inefficiency. The mean allocative efficiency is quite high, at 0.89. This suggests that most farms are using an input mix that approximates the cost minimizing the input mix. The high mean allocative efficiency scores are most likely due to the production technology, which is well known and adopted by farms (Coelli et al, 2007).

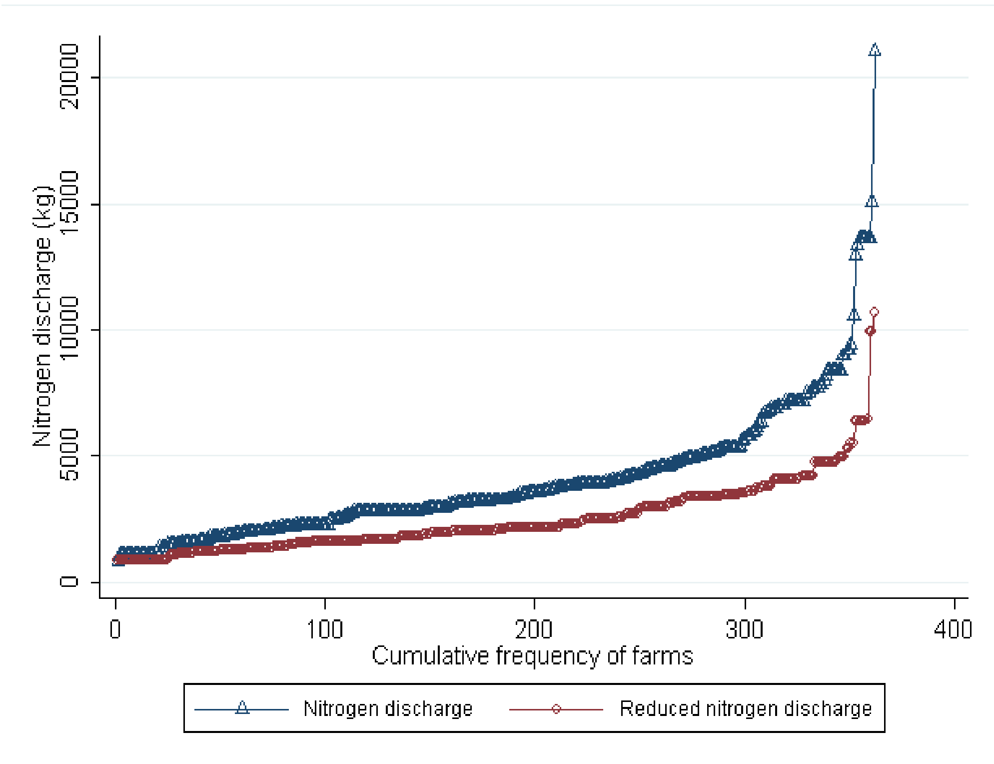
The mean environmental efficiency of 0.64 indicates that the average farm may be able to produce their current level of output with 36 percent less nitrogen discharge. Extrapolating from this across the catchment would suggest that 552, 962 kg of nitrogen discharged per year could be avoided if all farms achieve environmental efficiency. However, natural geophysical factors such as soil type and topography are likely to make this difficult to achieve. Agri-environmental policies need to consider differences in the inherent efficiency of farms. Figure 7.9 compares the cumulative nitrogen discharge levels between the status quo and the environmentally efficient scenario. This indicates the potential for very significant nitrogen discharge reduction in dairy farming, without any need to find extra and expensive new technologies for pollution reduction. However, there is a cost associated with operating at the emission minimizing point. Table 7.4 shows average nitrogen discharge and expenditure in relation to economic and environmental efficiency. Achieving environmental efficiency costs on average $757 per ha. Moving from an economically efficient nitrogen discharge level to an environmentally efficient discharge level reduces the mean nitrogen discharge by 38 percent. This information can be used to determine the shadow cost, which is (2534-1777)/(39-24)= $50.50 per kg for this nitrogen discharge reduction. Appropriate environmental policies may be required in order to move farms towards an environmentally efficient point. Whether to do this or not depends on the relative costs of alternative best management practices. Some potential best management practices are illustrated in the Appendices 5.1 and 5.2.

Farms are performing relatively well in achieving both economic and environmental goals, but there is room for improvement. Caution should also be exercised in interpreting the combined environmental-economic efficiency scores as further testing with different datasets should be undertaken. Reaching economic efficiency enables the enhancement of environmental efficiency to some extent as demonstrated by the mean efficiency. This is true in case of environmental efficiency followed by economic efficiency as well.

Table 7.4 Average nitrogen discharge and expenditure for economic and environmental efficiency

	Economic efficiency	Environmental efficiency
Nitrogen discharge (kg/ha)	39	24
Farm expenses ($/ha)	1777	25340

Figure 7.9 Comparison of nitrogen discharges



7.9.2 Environmental efficiency variation

Factors affecting environmental efficiency are shown in Table 7.5. The pseudo R^2 of 0.068 reported may not be the best measure of fit, so R^2 is based on predicted and observed efficiency values. The calculated value is 0.44, which is similar to OLS R^2. The model, therefore, explains 44 percent of the variation. Given the cross sectional nature of the data, the fit can be considered reasonable. As might be expected, stocking rate has a negative and significant effect on environmental efficiency, indicating that lowering the stocking rate has the potential to significantly improve environmental efficiency. The effect of the production potential of each dairy cow is positive but not significant, which may simply reflect that there is little variation in production potential. However Ondersteijn et al (2002) found a higher milk production per cow is concomitant with fewer cows and increased efficiency in terms of conversion of feed into milk. The market value of cows has been used as a proxy for breed quality and seems to have a slight positive effect on efficiency. Reinhard, Lovell, & Thijssen (2002) also showed that a more productive breed of cows could contribute to environmental efficiency by reducing the stocking rate and increasing the feed conversion efficiency. Maize silage has a positive effect on efficiency but it is not significant, which may be due to low levels of usage (on average 0.2 tons per head). According to farm trials feeding maize silage tends to reduce nitrogen discharge by 10 percent because of a higher conversion of nitrogen to milk in low protein supplementary feed (Ledgard, Penno, & Sporsen, 1999). There are concerns over feeding maize silage, however, because feed cost is higher and there are additional nitrogen discharges from growing the extra maize.

The Podzol soil group is used as the base to interpret the coefficients on the dummy variables. It is represented by the regression intercept. The estimates on the three dummy variables thus measure the proportionate difference in environmental efficiency in relation to Podzols. The effect of pumice soil on environmental efficiency is significant and negative, since pumice soils are prone to nitrogen leaching. However, the negative impact of volcanic and Podzol soils is less pronounced than with pumice soils, showing the importance of considering geophysical variations when designing policies for water quality improvement.

Table 7.5 Parameter estimates for environmental efficiency

Variables	Estimate	Standard error	t-value	p-value
Intercept	85.45	8.57	9.97	0.000
Production environment				
Maize silage/cow	0.61	2.74	0.22	0.82
Market value/cow	0.02	0.01	3.34	0.01
Milksolids per cow	-0.01	0.01	-0.77	0.44
Stocking rate	-4.39	1.51	-2.92	0.00
Physical environment				
Dummy variables				
Volcanic-easy*(0.24)+	-17.10	2.77	-6.18	0.00
Volcanic-rolling*(0.18)+	-19.16	2.89	-6.63	0.00
Pumice_rolling*(0.33)+	-25.39	2.69	-9.37	0.00
Pumice_easy* (0.14)+	-25.19	2.97	-8.54	0.00
ó	10.05	0.50		
Pseudo R^2	0.07			
Log-likelihood	-772.12			
Number of observations	210			

* *Podzol- rolling* is used as a base and captured by the intercept term

+ The values in parenthesis behind the dummy variables indicate the percentage of the total observations that are described by each dummy variable.

7.10 Conclusion

This chapter has developed an analytical framework to measure environmental and economic efficiency. The second stage parameter estimates reflect the impact of variables that can guide policy to improve environmental efficiency.

The farms studied are shown to be technically efficient producers, but there is still significant room for improvement in terms of environmental efficiency.

In order to realize the environmental improvement potential, it would be useful to identify the characteristics of those farms that are environmentally efficient. While there is a natural incentive for improving economic efficiency, some reluctance can be expected with environmentally motivated changes in production practices. Economic efficiency can be viewed as a private good for farms. Environmental efficiency, on the other hand, is a public good, important from a social point of view. It may, therefore, be necessary to provide further incentives through regulatory initiatives (Asmild & Hougaard, 2006).

There is an increasing need for objective measurement of farm environmental performance, and environmental efficiency scores can be good indicators of the stress farms place on the environment. Environmental efficiency measures can also provide useful guidelines for policy makers (Tyteca, 1996). They enable impact analysis of various environmental policies such as use of pollution standards, taxes or tradable emission permits. Picazo-Tadeo & Reig-Martinez (2007), for instance, used DEA environmental efficiency scores to assess farm performance under nitrogen taxes and nitrogen use permits. Given adequate data, this approach can be extended to analyse other environmental issues such as greenhouse gas emissions and energy efficiency. There is a potential to develop an environmental performance index by incorporating information on the damage costs of environmental impacts into DEA framework (Munksgaard, Christoffersen, Keiding, Pedersen, & Jensen, 2007). Once the damage costs of nitrogen discharges are determined, the social cost efficiency of farms can be established. DEA scores have been used to develop sustainability indicators as well (De Koeijer, Wossink, Struik, & Renkema, 2002).

In the efficiency measurements, it is assumed that farms do not adopt any best management practices. A range of such options are proposed, such as limiting external nitrogen input, increasing nitrogen use efficiency via lower protein feed resources, reducing farm dairy effluent losses, avoiding direct deposition of excreta to land in autumn/winter by using grazing off or feed pad systems or herd homes and nitrification inhibitors. However, these best management practices may need additional inputs such as extra capital for building feed pads or herd homes. In order to model the impact of best management practices in a mathematically feasible way, Coelli et al (2007) suggest including the amount of pollution abatement as an output variable. This enables the measurement of changes in various efficiency measures. For instance in the farming context, the amount of nitrogen abated as a result of herd homes can be included as a good output, while additional costs can be included with conventional inputs.

Further research on the socio-economic characteristics of farms is needed to identify the reasons for variations in environmental efficiency. Farmer characteristics such as education, experience and ownership structure can play a major role in farm efficiency. For instance, the management ability of the farmer is affected by personal characteristics (Nuthall, 2001), and owner-operated farms are reported to be more efficient (Wossink & Denaux, 2006). The virtual population data used for the analysis does not include such information, and this is an area for further research. Given the virtual nature of the data, the efficiency measurements and associated results discussed in this study need to be treated cautiously.

Finally, farm level environmental-economic efficiency scores should not be directly interpreted as representing the amount of environmental harm caused by farms, since the location of farms in relation to a water body may influence the damage to the water body. In addition, some farms could be taking measures to abate pollution through the adoption of best management practices such as using nitrification inhibitors and winter pads. Therefore an extension of the model to incorporate these could be of interest, given ready availability of data on abatement activity.

8. Modelling interception technology and potential land use changes with respect to nitrogen pollution

8.1 Introduction

Two major strategies used to manage nonpoint pollution are source reduction and interception (Ribaudo, Heimlich, Claassen, & Peters, 2001). Source reduction strategies induce changes in the way nutrients are managed on the farm. Interception strategies involve filtering out nutrient flow and transforming nutrients from surface and sub-surface farm discharges before they reach surface waters. Interception strategies are used for nitrogen abatement in many studies (Ribaudo, Heimlich, Claassen, & Peters, 2001; Tanner, Nguyen, & Sukias, 2005). Common types of interception strategies described in the literature are wetlands, riparian buffers strips and the creation of forest land. Riparian buffers generally encompass vegetative strips of land that extend alongside the streams, rivers and bank of lakes (Parkyn, 2004).

Riparian buffers have become an important abatement tool in the recent past. The use of it is widely promoted as a mitigation measure for the effects of sediment and nutrient runoff from land use intensification in New Zealand (Collier, Davies-Colley, Rutherford, Smith, & Williamson, 1995). The ‘Clean Streams’ project of the Ministry for the Environment (2003) encourages and supports farmers in their efforts to reduce the impacts of farming on waterways by rendering advice and financial assistance up to 35 percent of costs for fencing and planting waterway. The establishment of riparian buffers are costly and funding is limited. However not all land affect water quality goals in the same way. A social planner must therefore identify effective ways to allocate their scarce funds among heterogeneous farms. To facilitate the decision making process, a stylised model for a riparian buffer is developed for the Karapiro catchment, followed by an empirical analysis using a virtual population of farms derived in the chapter 4 to consider the impact of land use conversions on water quality.

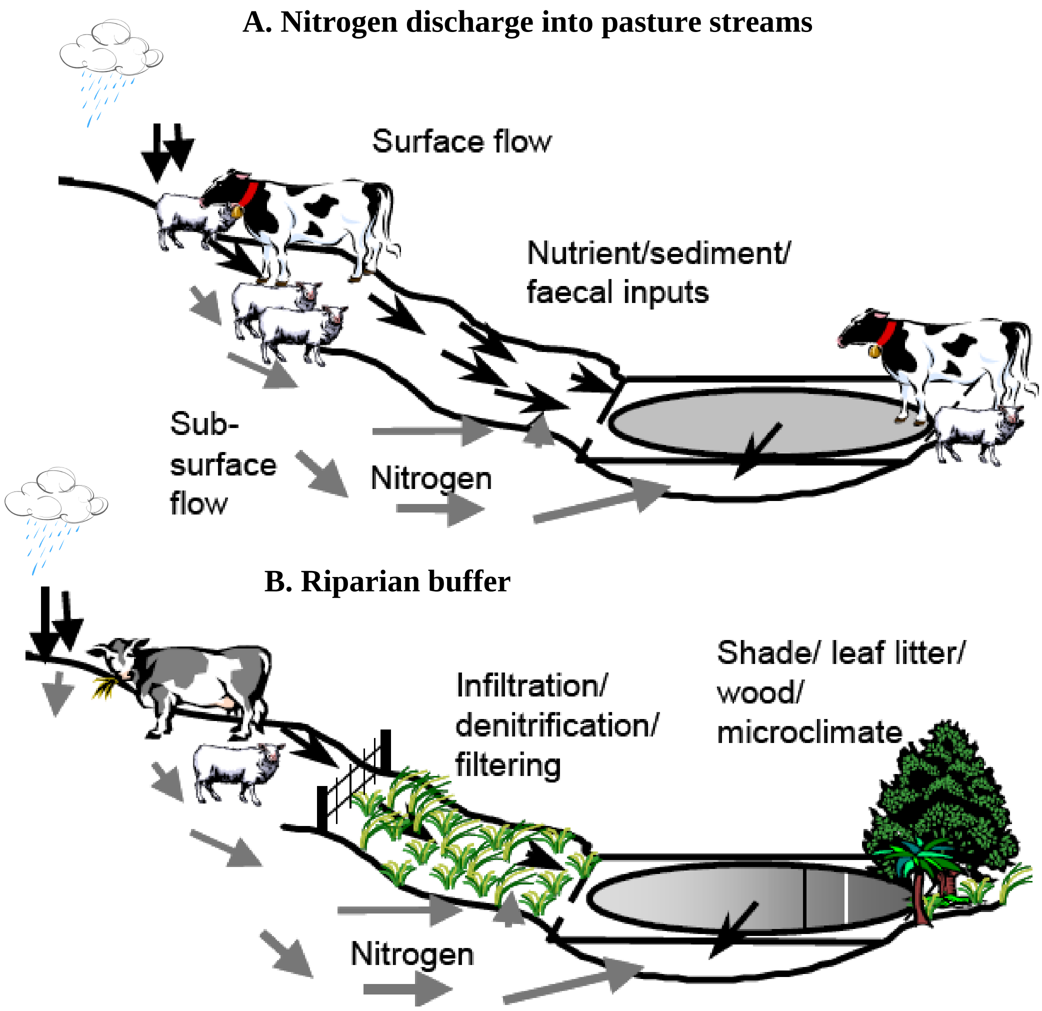
Large-scale deforestation of planted forests (land use change from forest to principally grazing) has a major potential to increase nitrogen discharges. In 2005 the area deforested in New Zealand was estimated to have been approximately 18% of the area harvested (Smith & Horgan, 2006). A survey (Manley, 2005) indicated that, without policy intervention or a significant change to the projected profitability of various land uses around 170,000 ha of the existing plantation estate would be deforested over the period from 2005 to 2020. Forest area in the southern part of the catchment has been identified as one of the potential areas for deforestation. Therefore this chapter also looks into the implications of potential land use changes on nitrogen discharge.

8.2 Farm nitrogen and riparian efficiency

The estimation of farm nitrogen delivery to water bodies is a complicated issue for variety of reasons. First, it requires estimation of nitrogen discharge, which depends on land use and geophysical properties. The term nitrogen discharge reflects the nitrogen lost from the farm through leaching and runoff. In pastoral systems nitrogen discharge is calculated by the amount of nitrogen applied in fertiliser, farm dairy effluent, urine and dung by grazing animals depending on soil type and porosity, topography and rainfall. Disaggregating nitrogen discharge into runoff and leaching is problematic as very little information is available to distinguish surface and subsurface flows (Thomas, Ledgard, & Francis, 2005). Thus the Overseer model does not differentiate between leached nitrogen and runoff nitrogen. Nitrogen delivered into a water body requires estimation of nitrogen transport that depends on the distance, hydrology and terrain features of flow pathway.

The effectiveness of a buffer will depend upon its ability to intercept nitrogen in its various forms travelling along surface and subsurface pathways. Riparian buffers contribute to nitrogen removal by stock exclusion; filtering the surface runoff; vegetative uptake and biological denitrification (Martin, Kaushik, Trevors, & Whitely, 1999). Biological denitrification was the main mechanism for nitrate removal. Figure 8.1 illustrates the riparian approach to nitrogen removal.

Figure 8.1 Nitrogen flow and riparian margin



Adapted from: (Parkyn, 2004)

According to the scientific literature nitrogen removal efficiency of riparian buffers can be quite variable. Philippe & Hill (2006) cited many US studies in which nitrate removal efficiency from sub surface flow varies from 90 to 44 percent. Gilliam, Parsons, & Mikkelsen (1997) reported that buffer zones are capable of removing 50-90% sediment associated nitrogen from surface run off and subsurface flows depending on the hydrology. Parkyn (2004) cited Fennessy & Cronk (1997) and Gilliam (1994), who claimed greater than 90% reduction in sub surface nitrogen by forested riparian buffers. This may be due to nitrogen uptake by deep tree roots and denitrification. A study by Williamson, Smith, & Cooper (1996) revealed that the riparian margins were capable of reducing particulate nitrogen by 26 percent. A recent report prepared for the "Water programme of action" by Agribusiness group et al (2007) stated that buffer strips are capable of removing 7% of total nitrogen discharge (4% by filtering and 3% by stock exclusion). Bedard-Haughn, Tate, & Kessel (2004) reported buffer effectiveness for nitrogen removal as follows; 8 meter buffer 28% and 16 meter buffer by 42%.

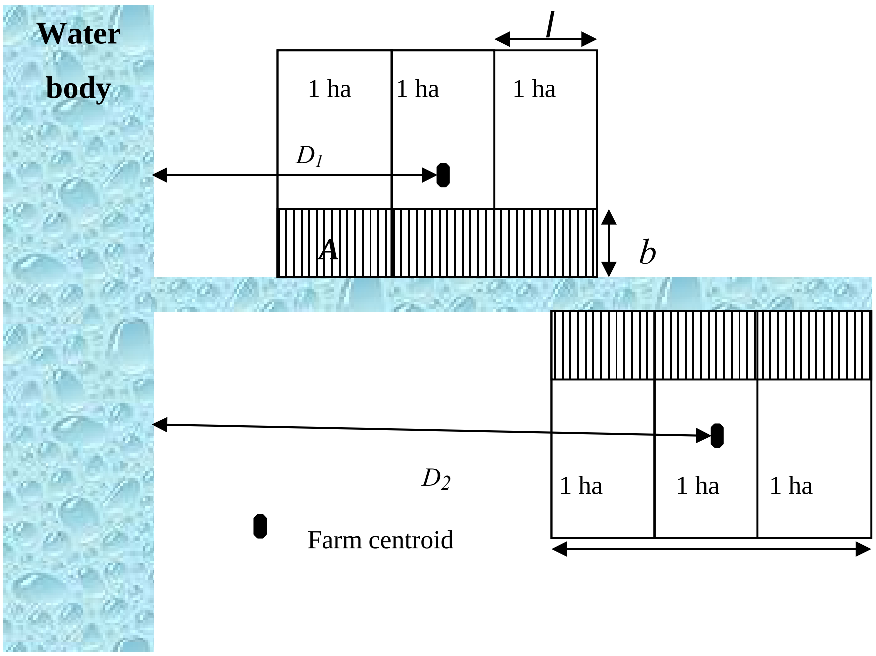
Lowrance et al (1997) reported that from surface flow 73% of nitrogen is removed by a 9.1 m buffer strip and 54% removed by a 4.6 m buffer strip of dense vegetation. In general, the steeper and longer the slope that feeds into the waterway, the wider the riparian buffer needs to be. Collier et al (1995) recommended 1-3 meters width for gently rolling slope and 5-10 meters width for steeper slopes. Stace & Fulton (2003) considered 5-10 meters width riparian margin along the lake margin for riparian protection works in the Rotorua catchment. Fencing of riparian margins reportedly has the potential of removing 90% of nitrogen from surface flow by means of enhancing the microbial action (Environment Waikato, 2004). Some claim lower efficiency as nitrate rich ground water tends to flow under the riparian zones and discharge directly to streams. For instance the key nitrate pathway with porous pumice soils in the central North Island of New Zealand is vertical, down to groundwater. So the nitrate predominantly bypasses riparian vegetation (Howard-Williams & Pickmere, 1999). Wilcock et al (2006) reported that intercepting surface or subsurface nitrogen flow was inadequate as most of the nitrogen loss is through drainage. Therefore determining overall buffer effectiveness requires an understanding of the attenuation efficiency with respect to nutrients washed into the buffer and quantification of the nutrient load that bypasses the buffer (Parkyn, 2004). Generalisation of these cited performances are used later in empirical analysis.

8.3 The model

Farm nitrogen discharge is assumed to depend on stocking rate, nitrogen fertiliser application, soil type, topography and length of riparian margin (equation 1). Where, S_i is the vector of stocking rate, N_i is the vector of synthetic nitrogen application. θ_i is the vector of geo physical parameters such as soil type and topography.

$$Z_i = f(S_i, N_i, \theta_i) \qquad (1)$$

Figure 8.2 Hypothetical farm



A hypothetical farm is divided into segments of one hectare (Figure 8.2) and a model is built for the segment. Let *A* be the share of that land in which riparian buffers have been established. The extent of riparian buffer depends on length of stream margins and topography of the land. It is assumed that buffers start at the stream or river's border and extend continuously outward away from the banks symmetrically on both sides of river or stream[1]. A riparian buffer of extent A is assumed to have constant width *b* throughout its length *l*. In the absence of consistent experimental findings on the surface/subsurface flow of nitrogen and filtering ability, the following assumptions are made. Surface and subsurface flow component of nitrogen discharge is denoted by φ_i, which is assumed to be 25% of farm nitrogen discharged (Z_i) plus a maximum 25% of Z_i depending on the normalised[2] value of riparian length per ha[3] in each farm (equation 2). The remaining 50% is lost through leaching.

$$\varphi_i = \int_{l}^{MaxL} \int_{0}^{25\%} L_i \, Z_i + 0.25 Z_i \qquad (2)$$

Riparian buffers are assumed to be capable of removing a maximum of 80% of the sub surface/surface flow nitrogen. Thus up to 40% of nitrogen discharged is intercepted. The intercepting ability of a riparian buffer has been modelled by adapting the functional form used by Lankoski & Ollikainen (2003) and Lankoski, Lichtenberg, & Ollikainen (2008a). This functional form is modified by using parameters and specifications from New Zealand based experimental studies (equation 3).

$$q_i = \varphi_i (1 - 0.1 * b_{ij}^{\beta}) \qquad (3)$$

The maximum recommended buffer width for gently rolling landscape is 5 m and 10m for steeper landscapes. Based on this recommendation, maximum buffer width is assumed to be 7m. Topographic differences are not explicitly considered in this analysis of nitrogen removal efficiency within the buffer strip. However topographic differences are taken into account when estimating the nitrogen discharges entering the buffer strip. Further the predominant topography of all dairy farms in the

---

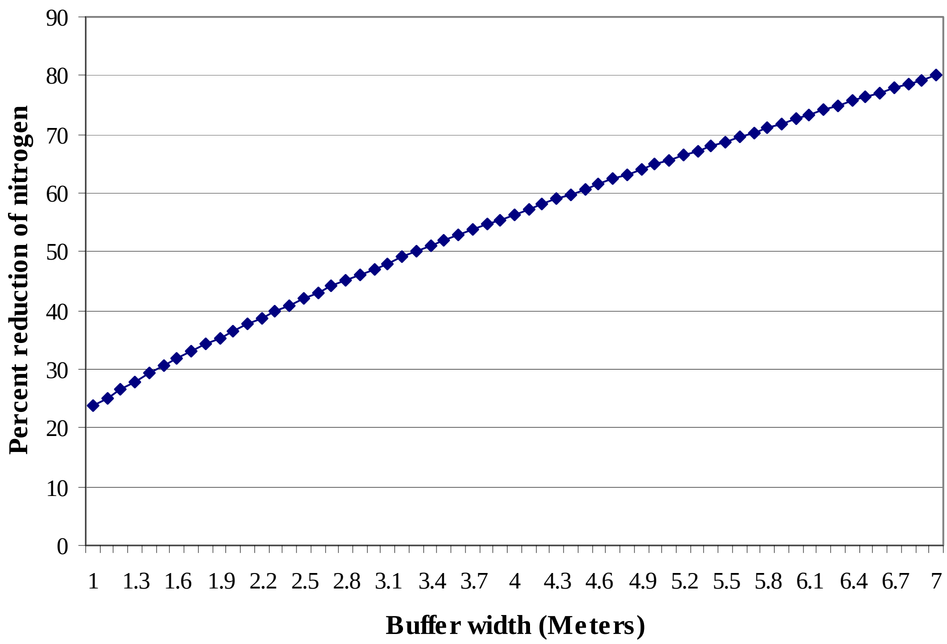
[1] Width (*b*) of the riparian buffer is assumed to be constant for a particular farm type. Therefore the extent of buffer *A* translates into the width b of the buffer at that location.

[2] Riparian length is divided by maximum of riparian length per ha (404.6 km/ha)

[3] It is assumed that runoff and animal contribution are positively related to the riparian length per ha as it exposes more water to surface flow as well as increases probability for stock crossing.

catchment is in the range of easy to rolling, except for 4 dairy farms with steep slopes. The maximum intercepting or filtering capacity is capped at the maximum buffer width. The marginal abatement rate of the buffer is assumed to be declining function of width of the buffer. The β coefficient for equation 4 is derived through non linear optimisation, based on the maximum buffering capacity and the width. Figure 8.3 shows filtering effectiveness as a function of width.

Figure 8.3 Riparian buffer effectiveness



$$q_i = \int_{0m}^{7m} \int_{0\%}^{80\%} \varphi_i (1 - 0.1 * b_{ij}^{0.63}) \tag{4}$$

Quantifying the nitrogen contribution of direct livestock contact is a huge challenge as it is influenced by animal behaviour and the length of streams etc. Davies-Colley, Smith, Young, & Phillips (2004) estimated that herd crossing increased total nitrogen into water by 10% depending on the length of riparian margins. Therefore it is assumed that the farm with the highest length of riparian margin per ha, get its nitrogen contribution increased by 10% of nitrogen discharge per ha (Z_i) as a result of direct livestock contact (Equation 5).

$$S_i = \int_{l}^{Max\,L} \int_{0\%}^{10\%} L_i . Z_i \tag{5}$$

Riparian buffers are assumed to be capable of completely stopping the direct livestock nitrogen component. Therefore total abatement from riparian buffers can be modelled as follows (Equation 6)

$$\varphi_i (0.1 * b_{ij}^{0.63}) + l_i b_{ij} (z_i) + S_i \tag{6}$$

8.3.1 Nitrogen decay function

In order to model the amount of nitrogen received from the farm to the water body with simplicity and no loss of generality, farms along the single tributary, draining into the water body are considered (Figure 8.2). The proportion of nitrogen removed or retained in the flow path to the main stem of the water body is assumed to be a function of distance. The functional form and parameters proposed for this decay function by Skop & Sorensen (1998) were adopted for this study (Equation 7). The rationale for adopting this functional form is that the longer the distance it takes to transport to the water body the more nitrate can be removed by denitrification or retained by accumulation in biomass or sediment. These parameters are constant per unit distance. The decay process (retention /removal) includes processes that occurred from the time when nitrogen is discharged until it appears in the main stem of the water body. It is assumed only leached nitrogen is subject to the decay process. This is consistent with empirical application of Skop and Sorensen (1998).

$$0.5 Z_i (1 - P)^{D_i} \tag{7}$$

In equation 7, $(1 - p_y)^{D_i}$, denotes the fraction of nitrogen not retained or removed from each kg of nitrogen discharged through leaching. *P* indicates the probability of nitrogen detention, which is equal to 0.00085 according to Skop and Sorensen (1998). D_i is the distance from the main stem of the water body to the farm centroid[4] in km.

The total amount of nitrogen potentially delivered to the water body from each farm can be modelled as follows (Equation 8).

$$\Re_i = [0.5(1-P)^{F_i} + \varphi_i + S_i] - (q_i + S_i + v_i) \qquad (8)$$

v_i is nitrogen discharge averted as a result of converting land into riparian buffers. It equates to the total area of the strip multiplied by the discharge rates. For computational convenience the nitrogen discharge from land converted into strip is assumed to be 0. The total amount of nitrogen reaching the main stem of the water body can be defined as follows.

$$T\Re = \sum_{i}^{I} \Re_i \qquad i = 1, 2 \ldots\ldots\ldots I \textit{ farms} \qquad (9)$$

8.3.2 Damage function

The environmental damage cost depends on biological and economic parameters such as habitat degradation and commercial and recreational interests. Water pollution may affect both local residents and the general public living outside the catchment as people derive utility from the amenities and services that the water ecosystem provides. These amenities and services may include good drinking water, scenic lake views, fishing and other recreational opportunities such as water sporting. Water pollution may also causes ecosystem damages that are not fully internalised by local residents. For instance time lags with gradual accumulation of pollutant, impair the ecosystem, but only generates tangible decline in amenities once it crosses a threshold.

---

[4] *Centroid is* a polygon’s mean center which is based on the weighted average of its x and y coordinates.

Most of the bio-economic modeling studies relating to cost aspects of environmental change and account only for on-farm impacts. Typically these studies consider the costs of reduced production and additional expenditure to adopt abatement measures (Bennett, 2005). Modelling the damage function is a complex task. The damage function of nitrogen discharges can be estimated by means of the value of averting expenditure and or non market valuation. The averting expenditure valuation method estimates the costs of corresponding nitrogen reduction such as at municipal water treatment facility. With the non market valuation method the value changes in the quality of water is assessed through estimated willingness to pay. The choice modelling approach is commonly used to elicit monetary values. For instance Mallawaarachchi and Quiggin (2001) used choice-modeling-derived estimates of the values of different types of remnant vegetation with farm profit estimates in bio-economic modeling.

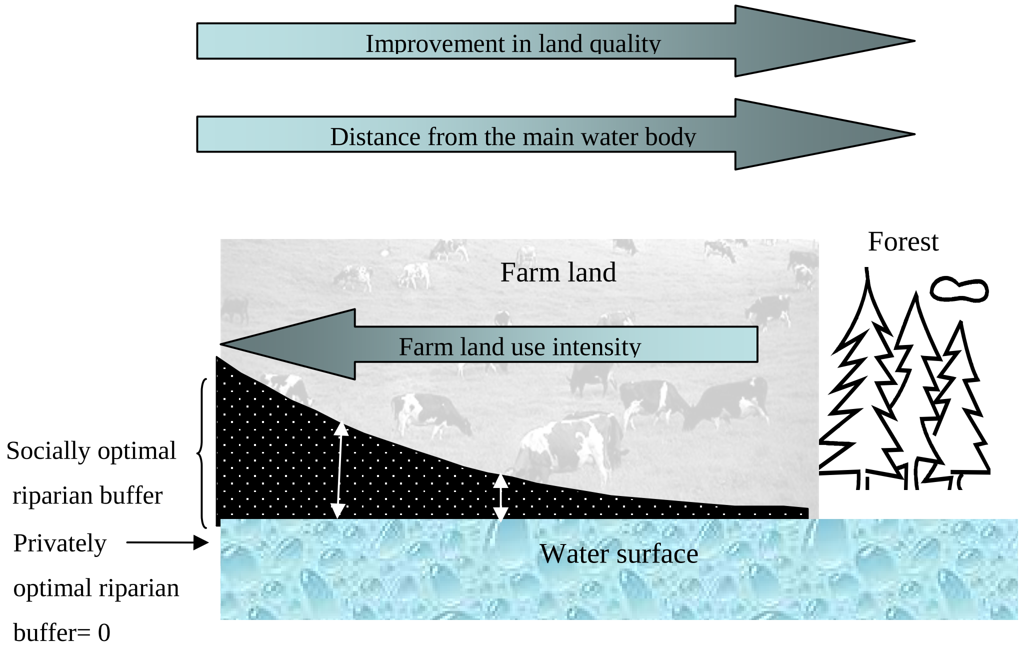
Since there is little information available on damage costs of nitrogen discharges in the New Zealand context, parameter estimates used by Martinez & Albiac (2006) are adapted as a starting value to model damage function. They used a value equivalent to NZ $ 2.50 as a cost to remove kg of nitrogen from water using a discontinuous tertiary biological denitrification treatment. In order to have an increasing function to reflect the cost of environmental damage, the damage function has been approximated by an exponential function of nitrogen delivered to the water body (Equation 10). Total economic damage *E (DF)* is a function of nitrogen delivered to the water body. *D (0)=0, D' (TR)=>0, D'' (TR)=>0.* The parameter lambda is the unit emission cost of nitrogen discharge and is set equal to the cost of removing a kg of nitrogen from water. λ =2.5. However it does not account for economic damage resulted from nitrogen discharges from other dimensions other than the perspective of quality drinking water. *k* is assumed to be 1.2 . A similar functional form has been used by Suter, Vossler, Poe, & Segerson (2008) to model damage cost. They assumed *k* is equal to 1.5.

$$E(DF) = \lambda(\Re_i)^k \qquad (10)$$

8.3.3 Optimum buffer width

Nitrogen received by the main stem of the river is assumed to be affected by land use intensity, land quality and distance, which together determine the effective width of riparian buffer (Figure 8.4).

Figure 8.4 Social and private optimum



The private optimum is based on profit maximising behaviour of producers. The social optimum is derived by incorporating negative externalities associated with environmental damage. The social optimum involves choosing the riparian buffer width (*b_{ij}*) and level of changes at intensive margin. The cost of riparian buffers equals forgone farm income due to land retirement plus the annualised cost of establishment and maintenance of the buffer.

Nitrogen has been treated as an assimilative pollutant (Tietenberg, 2006). Therefore the social optimum of dairy farming is modelled in this chapter in a static context. Nitrogen discharge has previously been treated as a static problem (Lankoski, Lichtenberg, & Ollikainen, 2008a). The static approach is sufficient due to the assimilative nature of nitrogen. If nitrogen is treated as a stock pollutant similar to phosphorus, the optimum conditions could have been derived using the Bellman equation and dynamic programming (Iho, 2007).

The cost of achieving reductions in nitrogen delivered can vary because of differences in production and pollution potential resulting from variations in geophysical factors and other factors affecting productivity such as management. Dairy farm production can be modelled as a function of nitrogen discharge. Modelling in detail is described in Chapter 5. To estimate nitrogen discharges, farm choice of nitrogen fertiliser, stocking rate and feed are considered. Dairy profit function is assumed to be increasing and concave with $f`$>0 and $f``$<0.

$$Max\sum_{i}^{I} A\,[Pf_i(Z_i,\theta)](1-l_i b_{ij})-c_i l_i - E(DF_i) \qquad (11)$$

$$s.t \quad \sum_{i}^{I} \Re_i = T\,\Re$$

8.4 Empirical analysis

8.4.1 Data

A virtual population of farms and other land uses generated using spatial micro-simulation in Chapter 4 is used for empirical analysis. Besides the soil and topographic features and intensity of production, the location of farm and farm exposure to the streams influence the delivery of nitrogen discharged to surface water. Stream length within each farm boundary is estimated using the River Environment Classification (REC) data base (detailed description is in Chapter 4). The minimum distance between each farm and the main stem of the river is calculated by estimating the distance between the Centeroid of each farm polygon to the main stem of the water body. This distance is used to calculate the decay function specified in the equation 7. Figure 8.5 displays the distribution of farm riparian margins and farm Centeroids.

Figure 8.5 Farm riparian margins and centeroids

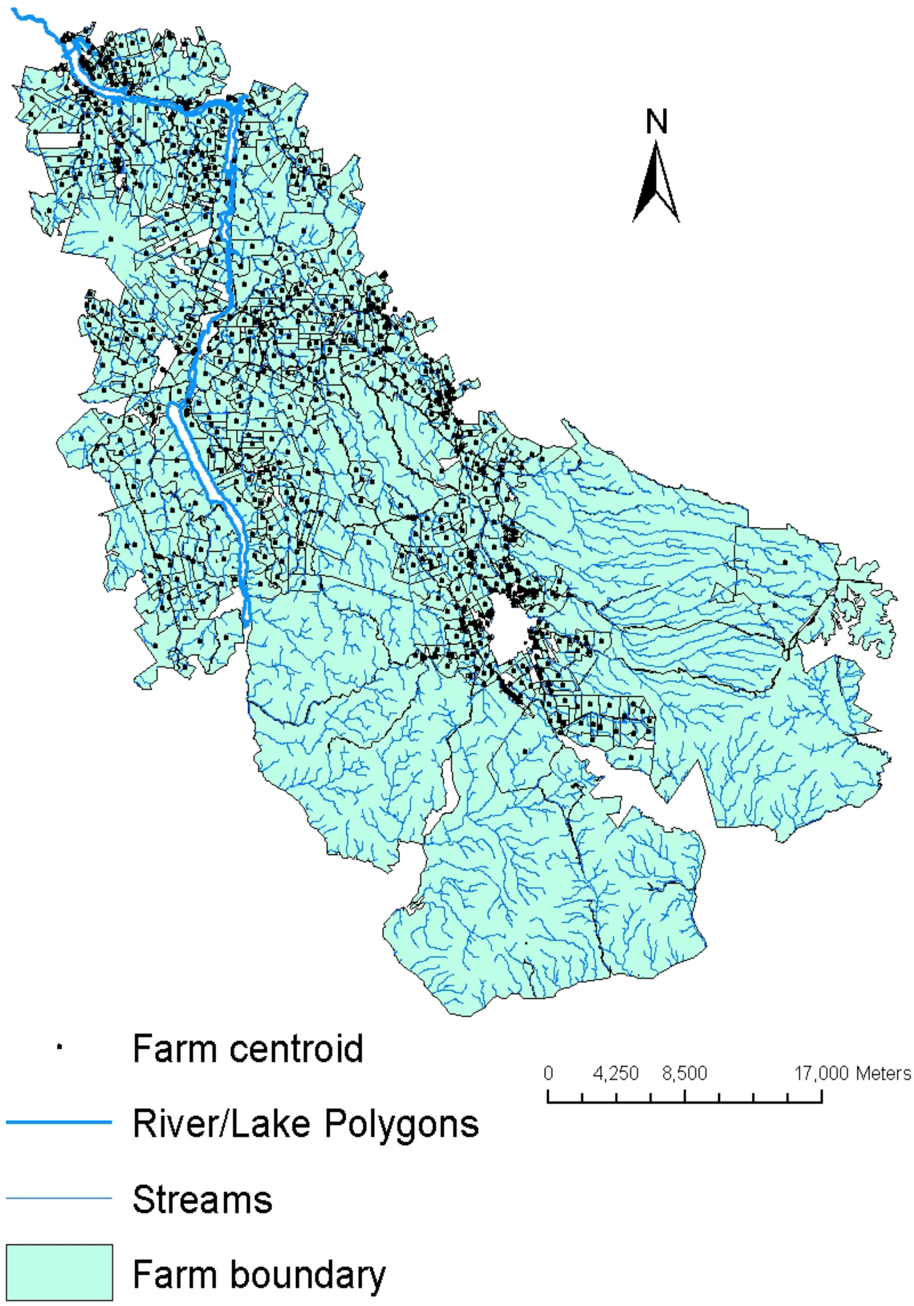
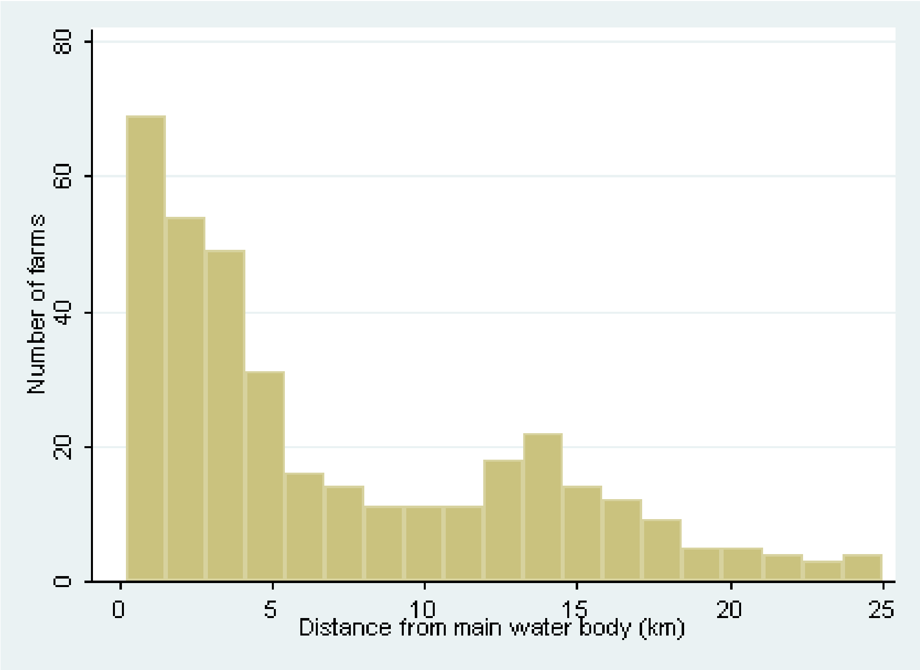


Figure 8.6 Distribution of distance to river



The length of riparian margins is in the range of 0-3 kilometres for more than 50% of farms. The distribution of farm distances from the main stem of the water body is tabulated in the Table 8.1 and displayed in Figure 8.6. More than 50% of dairy farms are located less than 4 kilometres from the main stem of the water body.

Table 8.1 Distribution of riparian margin

Riparian margin (Km)	Frequency	Percent	Cumulative percent
0-<3	197	54.42	54.42
3-<6	123	33.98	88.4
6-<9	30	8.29	96.69
9-<14	10	2.76	99.45

Table 8.2 Distance to river

Distance to river (Km)	Frequency	Percent	Cumulative percent
0-<4	196	54.14	54.14
4-<10	53	14.64	68.78
10-<15	63	17.4	86.19
15-<25	35	9.67	95.86

Since larger farms tends to have greater riparian margins, a riparian length to farm size index is calculated (Figure 8.7). This index ranges from 0 to 0.12 with a mean of 0.03 and standard deviation of 0.02. A higher value indicates a greater proportion of riparian margins within farms. Smaller values indicate a lesser proportion of riparian margins. Since riparian buffers have the opportunity cost of setting aside land, it is estimated by annual returns forgone per ha, annualised establishment cost, annual maintenance cost and cost of accessories. The return forgone is the quasi rent from farming, defined as total revenues minus total variable cost. According to the "Clean streams" guide book of Environment Waikato (2004), a 3 wire electric fence with fixed posts is the best option for riparian fencing on a dairy farm. Based on the per meter costs of fencing obtained from Environment Waikato, the following costs are estimated. Annualised cost of establishing fences is $10 per meter. Cost of establishing stock water supply and herd crossing structures is assumed to be 10% of the per meter cost. Annual maintenance cost is 10% of the per meter establishment cost[5]. The cost of riparian fencing is assumed to be constant through out the catchment. However in undulating and sloppy terrains it may cost more than on the flat terrain. Transaction costs such as negotiating and monitoring, involved with establishing riparian margins are not considered.

Only fenced grass riparian buffers are considered in this chapter. Planting riparian buffers with native plants and production forest species may incur additional cost in terms of planting material and maintenance. However these plantings may bring additional benefits by absorbing percolating nitrogen by deeper root zones as we noted earlier. No benefits are considered, associated with riparian buffers such as harvest of grass for silage making, erosion and flood control, habitat, shade, decreased water temperatures and recreational opportunities.

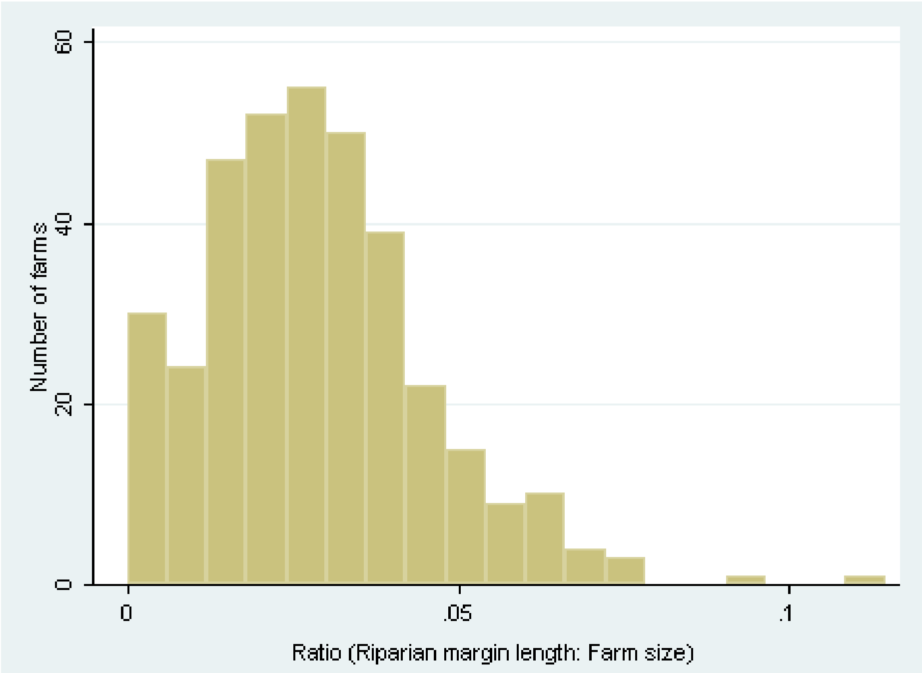
---

[5] Annualised cost is calculated as a sum of annual interest paid +cost of maintenance.

Annual interest ($/year)=((Establishment cost+ Salvage value +Depreciation)/2)*interest rate

Salvage value is assumed to be 0. Depreciation =Establishment cost/ lifespan of buffer

Figure 8.7 Riparian ratios



8.4.2 Functions, parameters and solutions

Numerical solutions for equation 11 require the functions and parameters to be specified. The parameters for the production function of different farming systems, were previously estimated in the Chapter 5 using a hybrid combination of WFM and meta modelling. As an alternative, here the function is estimated using the cross section data from the virtual population of farms in the catchment. Farm production functions can take different functional forms. Polynomial and the Mitscherlich-Baule are two of the specifications used in the literature related to agri-environmental policies (Goetz, Schmidt, & Lehmann, 2006; Hefland & House, 1995; Martinez & Albiac, 2006; Sumelius, Grgic, Mesic, & Franic, 2005). The Mitscherlich- Baule functional form has an attractive biological property of growth plateau beyond the input threshold. The parameters of this function are estimated using the nonlinear regression procedure in Stata version 10 (StataCorp., 2007). However, this specification quite frequently presents convergence problems in optimisation (Martinez & Albiac, 2006). In order to avoid potential difficulties in obtaining numerical solutions, particularly for the policies emphasizing changes at the intensive margin, the quadratic function is also estimated. A quadratic functional form presents a maximum yield level, although it lacks the property of a growth plateau. The empirical functional form derived here differs distinctly from the approaches of Martinez et al and Goetz et al, as it is in terms of nitrogen discharge rather than nitrogen fertiliser use. Table 8.3 presents the functional forms and estimation results for the Mitscherlich- Baule and quadratic response functions in terms of nitrogen discharge per ha *(Z)*. *Y* is milksolids produced per ha

Table 8.3 Functional form and parameters

Functional form	**Mitscherlich- Baule**	**Quadratic**
Parameters	$Y = \alpha(1-\beta e^{-\delta Z_i})$	$Y = \alpha + \beta Z + \delta Z^2$
α	2696 (1.96)	258.39 (2.38)
β	0.923 (39.35)	20.91 (4.02)
δ	0.010 (1.21)	-0.058
Adj R^2	0.95	0.42

*Statistics in parentheses are t statistics.

Average farm profit derived from catchment farms is 1.65 dollars per kg of milksolids[6]. Livestock contribution is a binary choice variable because in the presence of fencing, the animal contribution is 0 regardless of the buffer width. However in order to avoid the complexities in modelling it is treated as a part of runoff.

The maximisation problem stated in the equation 11 is solved by optimizing each farm individually to derive maximum farm returns and optimum riparian buffer width. It resulted in 3,620 serial optimisations for 10 scenarios. Figure 8.8 shows the problem formulation. This serial optimisation process was automated by developing a macro using visual basic applications, which activated Excel's built in solver (8.9). In another scenario the optimisation is implemented for varying nitrogen delivery levels without the riparian buffers to quantify the economic impact of changes at intensive

[6] Farm profit per kg of milksolids is similar to the farm profit defined in the Chapter 7.

margin. Optimisation of each scenario took about 10 minutes computing time on a Pentium IV 2GB RAM PC.

Figure 8.8 Spreadsheet formulation

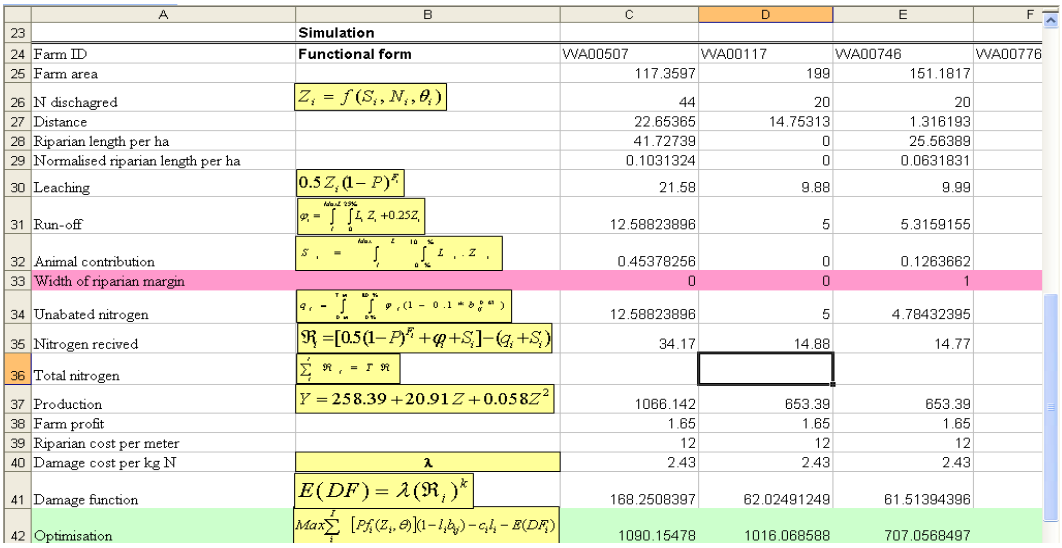
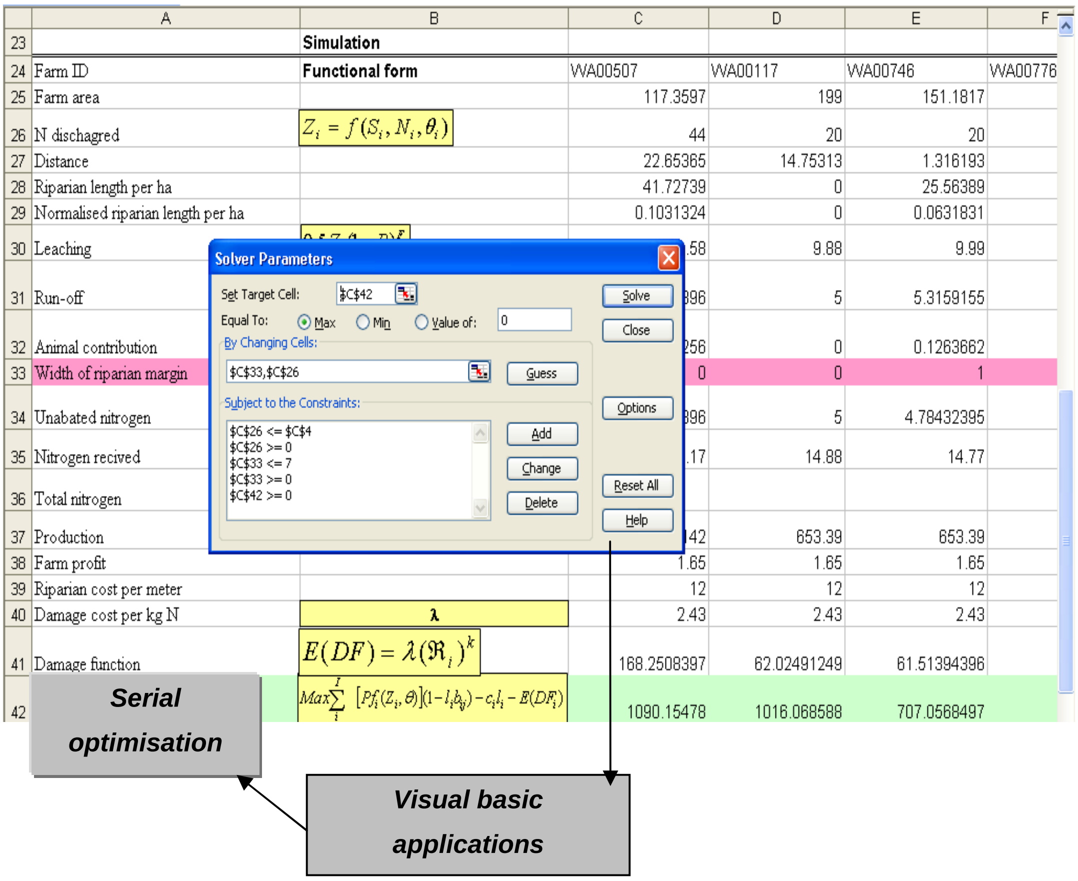


Figure 8.9 Serial optimisation



8.5 Land use change

Land-use change is arguably the most pervasive socioeconomic force driving the change and degradation of watershed ecosystems (Ministry of Agriculture and Forestry, 1997). Recently in the Waikato there has been a significant area of production forest converted to pasture dairying and large scale conversions are anticipated in future. In the Waikato, 67,023 ha forestry land was identified by MAF as at risk of profitable conversion from forestry to pastoral use. This chapter quantifies the extent of land which could potentially be converted and consequent economic and environmental effects using geographic information system and simple analytical techniques.

According to Smith & Horgan (2006) Land Use Capability (LUC) classes 1-5 have the potential for pastoral intensification. LUC is a hierarchical classification identifying: the land's general versatility for productive use; the factor most limiting to production; and a general association of characteristics relevant to productive use e.g., landform, soil, erosion potential, etc (Newsome, Wilde, & Willoughby, 2000). Land use capability classes are intersected with land use boundaries to quantify potential land use conversions.

Smith and Horgan (2006) estimated that expected maximum annualised net income as $800 per ha per year from forestry to dairying, given the cost of conversion, which is reportedly $4000 per ha. This results in net increase in the annual income by $559 per ha[7].

8.6 Results and discussion

Without any agri environmental policy, profit maximising farms will ignore the damages from ambient pollution and maximise profits. Farms undertake no abatement effort, because expected damage is not considered. It results in no riparian buffers. Figure 8.10 illustrates the pattern of changes in farm profit, optimum riparian

---

[7] Annualised net income per ha of forestry is $241. This has been calculated based on 8% discount rate and a $27,300 /ha lump sum in year 30.

width and nitrogen delivered into the water body for every dairy farm in the catchment.

Figure 8.11 shows the pattern of increase in the optimum width of riparian buffer, when the damage cost is varied. Higher environmental damage results in greater width, maximum specified width of buffer become optimum for more farms. Results indicate that the farm returns vary between 7-10% depending on the level of nitrogen reduction required to achieve the social optimum, which is determined by the damage cost. Simulation results for different scenarios are displayed in Table 8.4.

Table 8.4 Simulation results

Scenarios	Average buffer size/farm (m^2)	Average returns /farm ($)	Nitrogen delivered (kg)	Average nitrogen discharge /farm(kg)
Private optimum	0	179,370	1,185,169	4,272
Social optimum at a damage cost of				
$2.50 KgN	14,287	165,244	888,421	4,272
$2.75 KgN	16,022	164,121	871,728	4,272
$3.00 KgN	17,555	163,014	858,283	4,272
Reduction of nitrogen delivery by				
5 %	1,429	178,787	1,125,356	4,272
10 %	4,155	178,360	1,066,487	4,272
15 %	7,713	177,802	1,006,992	4,272
20 %	11,926	177,140	926,529	4,272
25 %	16,692	176,393	888,803	4,272
30 %	21,720	174,803	829,618	4,242
35 %	23,498	168,039	770,360	3,998

Figure 8.10 Farm returns, nitrogen delivered and optimum riparian buffer width for varying nitrogen delivery levels

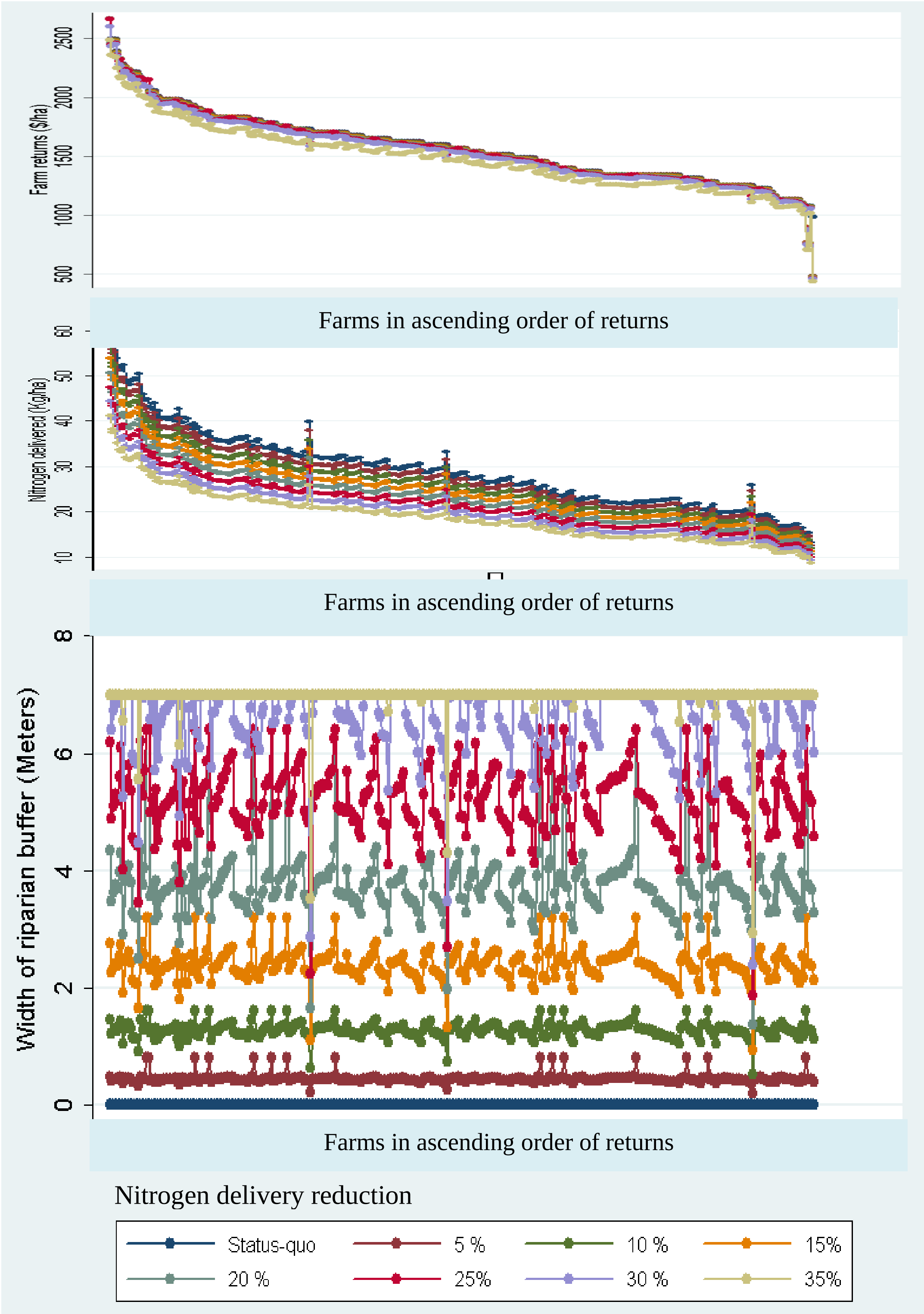


Figure 8.11 Socially optimum buffer widths at different damage costs

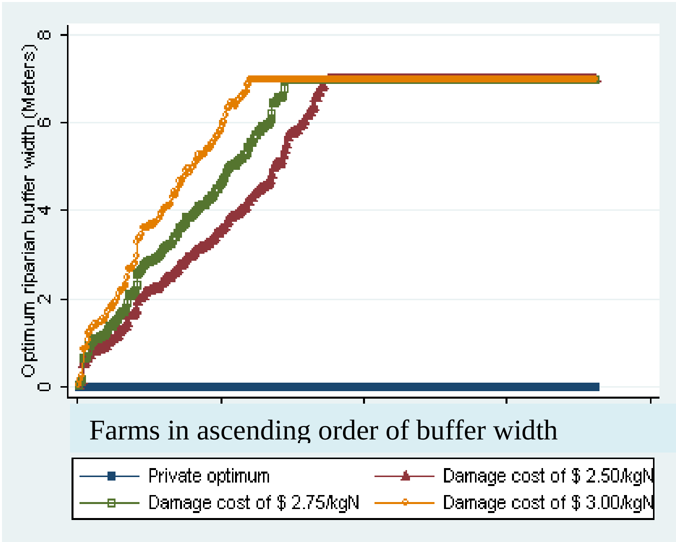
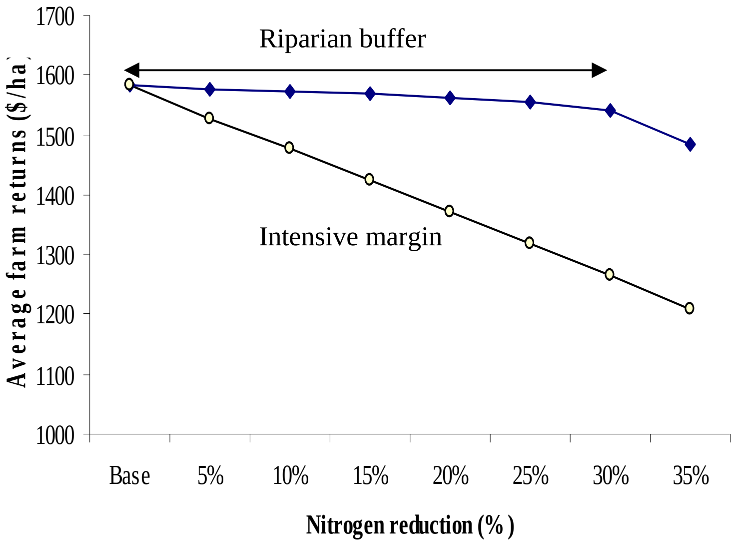


Figure 8.12 Cost effectiveness of riparian buffers[8]



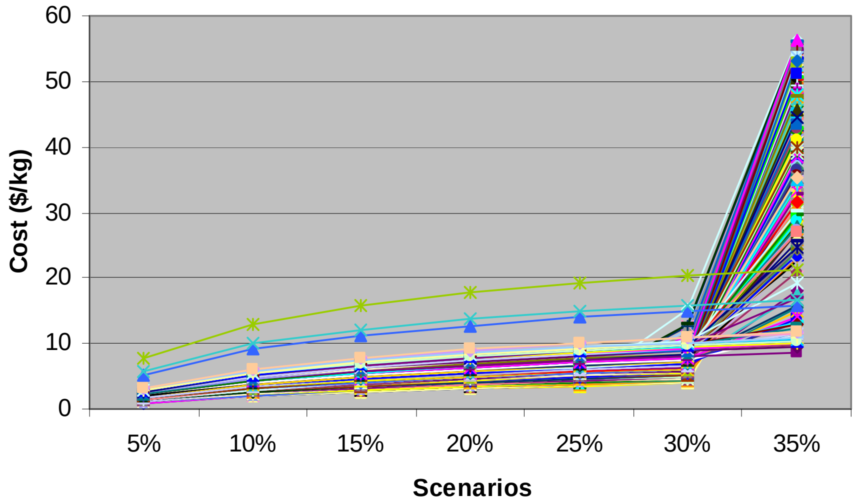
When nitrogen delivered to the water body is parametrically restricted the farming intensity remained unchanged until a 30% reduction (no changes at intensive margin). Given the assumption of nitrogen retention ability it indicates, riparian buffers are capable of reducing nitrogen discharges up to 30%. Beyond 30%, the reduction requires changes at the intensive margin. However riparian buffers are very cost

[8] Average farm returns indicate mean of all farm income under respective nitrogen reduction scenarios

effective when compared to changes at the intensive margin as the impact on average farm returns are less (Figure 8.12). Therefore it is rational to use riparian buffers complementary to changes at the intensive margin, when higher levels of reduction are required. In addition the cost of compliance monitoring associated with the riparian margin is reportedly less because of its visibility and difficulties associated with reversibility (Lankoski, Lichtenberg, & Ollikainen, 2008a).

The marginal cost of nitrogen pollution reduction equals forgone rent due to removing land from production, buffer costs and changes at intensive margin, divided by total reduction in nitrogen discharge. It is interesting to see up to 30% of reduction, the marginal cost of abatement seems to be an increasing and concave function (Figure 8.13). This is as result of dwindling marginal intercepting ability of the buffer and fixed fencing cost. Generally marginal abatement cost tremendously increases when abatement is achieved by changes at the intensive margin. Variations in marginal abatement cost draws attention towards possible implementation of emission trading schemes, which may be politically palatable as it includes potential compensation to farms. However the potential for successful trading partly depends on the initial allocation of permits as well as inherent farm heterogeneity. Socially optimum nitrogen reduction levels can be a basis to determine target levels of nitrogen reduction in potential trading programmes. The total number of discharge permits can be equal to the social optimum measured in terms of nitrogen delivered to the main water body.

Figure 8.13 Marginal cost of abatement



The relationship between riparian ratios shown in Figure 8.7 (higher ratios indicate greater the length of riparian margins per unit area) and the marginal cost of abatement at 5 % reduction in nitrogen delivery is explored (Figure 8.14). It seems that the cost of abatement is lesser in farms with a lower ratio. It is some what consistent with the findings of Bontems, Rotillon, & Turpin (2005), who stated that in France large farms had better ability to adopt land conservation practices. Perhaps farms with a lower ratio can adopt higher buffer widths.

Figure 8.14 Relationship between riparian margin density and abatement cost

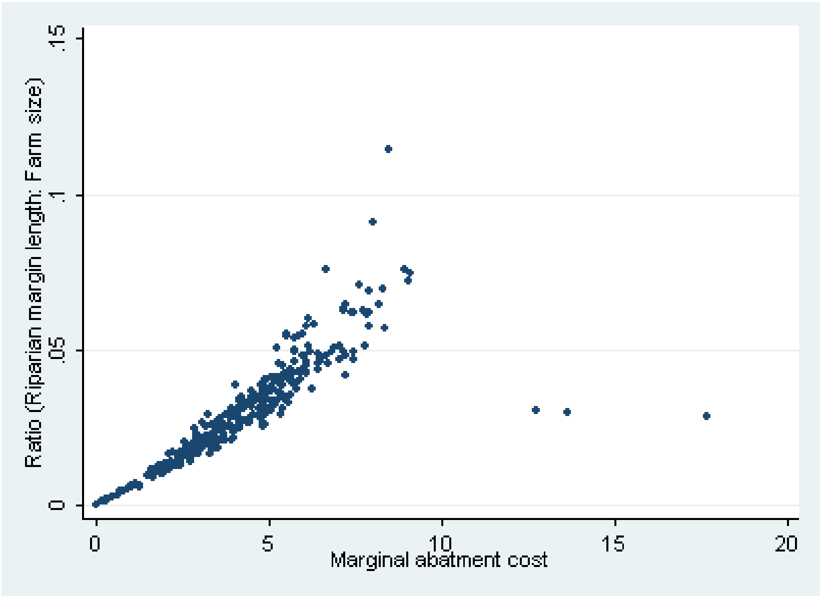
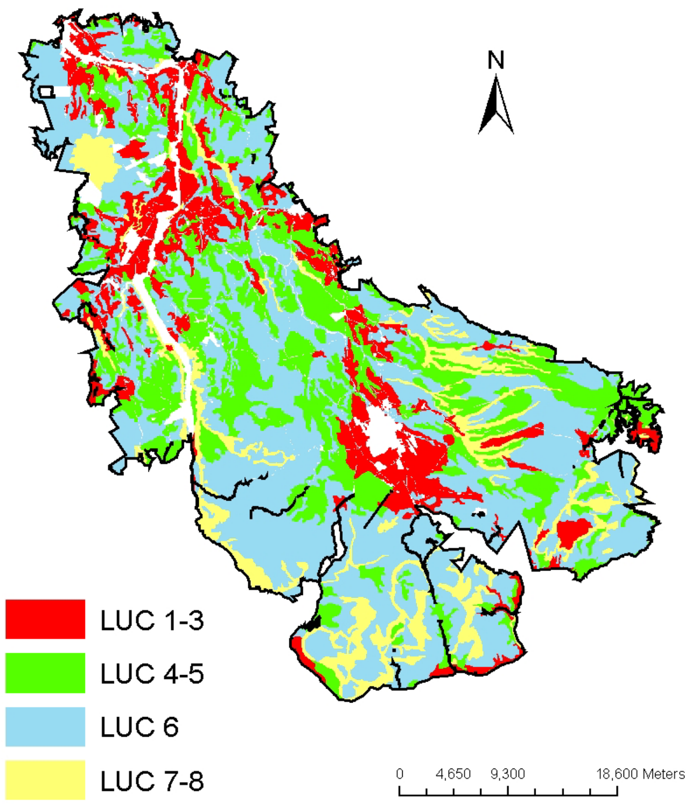


Figure 8.15 displays the distribution of land use classes across the catchment. Table 8.5 shows the extent of land use classes under non dairy land uses. Based on Smith and Horgan's assumption on potential land use capability classes for conversion, 29% of forestry land and 43% of Sheep and Beef land can be categorised as at risk of being converted as they belong to land use classes 1-5. This is equal to 30% of non dairying land use. Unlike existing dairy land uses, in quantifying future land conversion impacts, topographic and soil are differences are not considered. Instead the upper quartile mean nitrogen discharge of 53 KgN/ha of existing dairy farms is assumed to be the potential nitrogen discharge of converted land as pine to pasture conversions are reportedly profitable only for intensive dairying (Smith and Horgan,

2006). Undistributed exotic forestry discharges 3 kgN ha/yr (Menneer, Ledgard, & Gillingham, 2004). Nitrogen export rates from Sheep and Beef farms are in the range of 10 to 16 kg N /ha/Year (Hamilton, 2005). Based on these assumptions potential environmental and economic impact of forestry to dairy conversions are estimated (Table 8.6).

Figure 8.15 Land use classes



Conversion of forest land from 5 to 20% increases the returns by 6 to 27%. But the nitrogen discharge is quadrupled. Programmes, which provide financial incentives to remain in forestry like carbon credits can be a useful tool to avert conversion. For example a $110 million dollar Green-cover Canada programme, provides financial incentives to farmers for converting up to four million acres of economically marginal farm land into vegetative cover for achieving water quality, air quality and wildlife benefits (Yang & Weersink, 2004).

Table 8.5 Land use capability (LUC) classes across major non-dairy land uses

Land use capability	Plantation forestry (ha)	Sheep and Beef* (ha)
LUC 1-3	4316.17	2738
LUC 4-5	17056.40	3253
LUC 6	41060.22	7343
LUC 7-8	12278.21	663

* Sheep and beef farms include sheep farms and beef farms as well, besides sheep and beef farms

Table 8.6 Impact of forestry to pasture conversions

Scenarios	Returns ($)	Nitrogen discharged (KgN)
Status _quo	5,150,789	64,118
Forest to pasture conversion		
5%	5,510,917	117,549
10%	5,849,672	170,981
15%	6,188,428	224,412
20%	6,527,183	277,843

8.7 Conclusion

The main contribution of this chapter is to suggest a stylised model that accounts more precisely for interception technology to abate nitrogen discharge in New Zealand dairy farms. Given the filtering ability of buffers, they are very useful tool for improve environmental outcomes up to a certain level. To achieve higher levels of abatement riparian buffers must be complemented by changes at intensive margin such as stocking rate and fertiliser application. Goetz, Schmidt, & Lehmann (2006) showed regulating the intensive margin has to be complemented by regulating the extensive margin.

Since riparian buffers cause a relatively modest economic impact on farms, they can be more socially acceptable. Buffers are relatively easy to enforce as they are readily observable and costly to remove.

In this study the nitrogen sink function of riparian buffers is considered and the additional benefits they yield such as recreational amenities and biodiversity conservation are ignored.

The absolute results of the analysis presented here should be treated with caution as it is based on simplified assumptions on the process of nitrogen transportation and filtering. Transportation of nitrogen is a complex process, impacted by ongoing in stream process such as deposition and assimilation along the way. There are hydrologic simulation models such as AGNPS designed to capture such complex hydrogeologic and pollutant transport processes. Vegetative Filter Strip Models can also be used to determine the accurate width of buffer strips (Dosskey, Helmers, & Eisenhauer, 2008). However these require use of more detailed spatial information about location and other characteristics of land parcels and higher level of expertise to perform successful model calibration. Further this chapter has not considered the intra farm variability in soil type and topography and not accounted for sub surface drainage[9] on farm land due to lack of information on the distribution of drainage systems.

The stylised model would also be improved by input of better primary data, more detail on hydrology, and reliable region specific estimates of buffering capacity. Nitrogen decay and damage cost parameters were extracted from the literature and better localised estimates might change the results. Therefore research aimed at producing reliable, location specific estimates of parameters need to be carried out. The framework is easily modified to apply for land retirement schemes or conservation reserve programmes or establishing wetlands. The natural extension of this model is to incorporate forest buffers and capture the carbon sequestration capacity, which likely to enhance nutrient abatement efficiency and economic viability. An alternative interception technology known as Denitrification trenches (Schipper, Barkle, & Vojvodic-Vukovic, 2005) can also be explored using the

---

[9] Petrolia & Gowda (2006) firstly recognised the importance of considering tile drained farmland in modelling nitrogen abatement policies.

stylised model developed. This study implemented buffers throughout the stream network in the catchment for each farm. Selectively targeting the areas along the surface runoff pathways for community buffer strip establishment may be a worthwhile extension of this study by engaging an integrated framework that combines economic, hydrologic and geographic information system.

9. Summary and conclusions

9.1 Overview and policy implications

In the Waikato intensification of pastoral farming caused increases in nitrogen discharge into water. It has been estimated that Waikato dairy farms contribute 68 percent of nitrogen entering the water bodies and 35 percent of national dairy production. It is clear that actions to mitigate the problem are necessary if water quality is to be maintained or improved. Therefore it can be expected that dairy farms be targeted as a source of nitrogen discharge reduction to water. This necessitates analyzing the environmental and economic impact of abating the nitrogen discharges.

This study has developed and applied an integrated analytical framework to evaluate the potential economic and environmental effects of various policies. Application of this framework has been demonstrated on a sub-catchment of the Waikato River. This catchment of many dairy farms has experienced increasing nitrogen levels in the surface water as a result of intensification of dairy farming over recent decades. Water quality in the catchment is likely to deteriorate even faster, if large areas of pine continue to be converted to dairy farming.

Variability in the production and geographic characteristics of farms across the catchment affect both economic returns and nitrogen discharges. Consideration of variability at the level of individual decision making units is helpful in designing cost effective environmental policies. This suggests that spatially variable characteristics should be considered when designing policies to manage nitrogen sources of water pollution. However, such spatial information can be expensive to collect and time consuming. Historically, studies used a single representative farm to model a whole catchment. This thesis describes the development of a novel approach using micro-simulation and farm survey data to create a virtual population of farms for catchment scale modeling and policy analysis. The approach provides a solid basis for analysing issues underlying nitrogen discharges involving tradeoffs. The method can be employed with a modest amount of information to analyse environmental policy in a spatial context and so will be extremely valuable to regional councils and other policy making organisations that want to minimise the adverse economic impact of environmental policies. In particular, this study provided policy makers and planners with a decision support tool for designing and implementing policies related to water quality enhancement.

The DairyNZ WFM is used to simulate changes to different farming systems, which are established using the virtual farm population. The utility of the WFM has been extended as a tool for environmental policy analysis by integrating a Metamodel for nitrogen discharges, which are estimated using the Overseer software. The differential evolution optimisation algorithm of WFM is modified by incorporating nitrogen discharge constraints to identify optimum farming activities under different nitrogen discharge restrictions. The result is used to build a profit pollution frontier for each farming system. The profit pollution frontier is versatile. Firstly it is used to derive abatement cost functions for different farming systems. Secondly it is subject to a non linear optimisation in the mathematical programming model to analyse the responses of different farming systems to alternative policies. The metamodel and combinatorial use of differential evolution and nonlinear optimisation techniques saved resources and time.

The differences in abatement costs between farming systems are significant. This can be attributed to differences in production systems, soil type and topography. The average nitrogen abatement cost for the first 10 kilograms of reduction is $10.80 for a moderate farm; $ 2.70 for as an intensive farm and $31.50 for as an extensive farm. Information about abatement cost heterogeneity between catchment farm systems assists good environmental policy making as it helps to identify low cost farms. Targeted policies for taxing or restricting nitrogen discharges on certain production systems may be more cost effective than uniform policies.

Knowledge of abatement costs should enable social planners to design effective policies and reduce the information asymmetry problem. Further, knowledge of differences in abatement costs are likely to encourage farms to engage in trading of nitrogen discharge permits, with low abatement cost farms opting to abate more and sell some of their permits, while high abatement cost farms may prefer to buy more permits and maintain their emission levels.

Challenges to implementing environmental policy are largely related to property rights and information asymmetry. Redefining property rights requires development of scientifically and legally defensible proxies for nitrogen discharges. The profit pollution frontier developed in this thesis reduces information asymmetry by helping to identify a farm's likely emission profile based on easily observable data. Empirical application showed how to deter moral hazard arising from hidden action by targeting farms through differentiated incentives. Targeting farms with lower abatement costs tends to reduce the moral hazard problem as they have smaller payoff for renegade behaviour. This will reduce monitoring efforts and thus cost less. Given this approach may be contentious and face political resistance, it may be appropriate to enforce differentiated compensation plus penalty schemes for the forgone production losses. Compliance can be encouraged through technological subsidies such as extended extension services. Further, the profit pollution frontier facilitates the process of developing a menu of contracts with differential tax, abatement levels and penalties that would make polluters self select an appropriate contract.

Innovative institutions can both improve outcomes and reduce the transaction costs associated with policy implementation. Institutional structures must rely on more than voluntary governance and voluntary compliance and need to evolve to empower social planners to manage nitrogen discharges. There is potential for the Resource Management Act to be refined to enable the development and application of best management practices to improve water quality. The Vittel case in the USA (Depres, Grolleau, & Mzoughi, 2007) shows how rules for achieving desirable water quality can be set by a competent research authority, which can also play a mediating role in ensuring a mutual understanding of the varying interests of the social planner and the farmer.

Various efficiency measures were developed using the flexibility of DEA. These efficiency measures are useful for tracking the potential for environmental and economic improvements. Farms in the catchment have high technical and economic efficiency with mean values of 82 % and 72 % respectively. However environmental efficiency is only 64 % on average. Catchment level consideration of estimated environmental efficiency suggests that 553 tonnes of nitrogen discharge per year could have been avoided if all farms achieved environmental efficiency. However natural geophysical factors such as soil type and topography are likely to limit the extent to which this is feasible. Economic efficiency can be viewed as a private good for farms. Environmental efficiency, on the other hand, has the characteristics of a public good. Therefore it may be necessary to create further incentives through regulatory initiatives (Asmild & Hougaard, 2006). For example DEA environmental efficiency scores can be used to assess farm performance under nitrogen taxes and nitrogen use permits (Picazo-Tadeo & Reig-Martinez, 2007).

Analysis of efficiency variation reveals that lowering animal stocking rates has great potential to improve environmental efficiency. Per cow production potential is positive but has no significant effect. It may be due to the lack of variation in production potential per cow. The market value of cows has been used as a proxy for breed quality and seems to have a slight positive effect on efficiency. Reinhard, Lovell, & Thijssen (2002) also revealed that more productive cow genetics could contribute to environmental efficiency by reducing the stocking rate and increasing feed conversion efficiency. Maize silage has a positive effect on efficiency but it is not significant. This may be due to low levels of usage. The effect on environmental efficiency of having pumice soil is significantly negative, since pumice soils are prone to nitrogen leaching. The negative impact of Volcanic and Podzol soils are found to be less compared to pumice soils. This shows the importance of considering geo physical variations in designing policies for water quality.

A stylised model to facilitate policy making that accounts more precisely the nitrogen abatement by riparian buffers in a New Zealand pastoral farming context has been developed. The model predicts optimum buffer width for every farm under different policy scenarios. Given relatively low adverse economic impact and relatively easy enforcement, buffers can complement policies at the intensive margin to bring cost effective solutions. This model enables a social planner to identify effective ways to allocate their scarce funds among heterogeneous farms. Based on Smith and Horgan's assumption on potential land use capability classes for conversion, 29% of forestry and 43% of Sheep and Beef farm land can be categorised as at risk of being converted as they are on land use classes 1-5. This is equal to 30% of non dairying land use in the catchment.

9.2 Limitations and potentials for future studies

Best management practices are considered in the preliminary analysis based on hypothetical farms, but have not been subject to catchment wide analysis due to data constraints. Some best management practices are not modelled due to lack of local research. For instance Nitrification inhibitor response rates for Waikato farms have not been established yet (Ritchie, 2007). Adoption of best management practices is likely to increase farm environmental efficiency.
It may be of interest to extend the analytical framework by incorporating best management practices.

There is very little information on the extent of nitrogen that is lost in runoff from farms and there has generally been no differentiation between surface and subsurface flows (Thomas, Ledgard, & Francis, 2005). The Overseer model does not differentiate between leached nitrogen and runoff nitrogen. Estimation of the amount of nitrogen delivered into a water body requires estimation of nitrogen transport that depends on the distance, hydrology and terrain features of flow pathways. Complex nitrogen transportation process depends on the hydrogeology of land parcels. This must be considered in future modelling.

This study assumes that each farm has homogeneous soil type and topography for computational convenience and does not consider intra farm variability of them. However considering intra farm variability of soil and topography may be unrealistic due to the difficulties in restricting mobility of animals within certain soil and topography and further it makes policy implementation more complicated.

The present study could be extended in a number of ways, but assessing the impact of price variations would be the highest priority, particularly in view of the recent upsurge in world dairy product prices. Also the diversity of risk preferences should not be ignored in designing environmental policy given the stochastic nature of farming.

The natural extension of the riparian buffer model is to incorporate forest buffers and capture the carbon assimilative capacity, which is likely to enhance nutrient abatement efficiency and economic viability of buffers.

A further useful extension would be the construction of dynamic spatial micro-simulation model using the time tagged population variables. This work could make use of remote sensing and satellite imagery to produce time tagged population variables.

The model omits important influences on decision making such as farmer attitude, intrinsic knowledge base, experience and risk aversion etc. Direct data collection from farms is clearly optimal given no cost, time and effort constraints. However, given such constraints the approach taken substantially overcomes data limitations. The choice of appropriate policy requires estimates of marginal benefits of policy apart from marginal costs of implementation. Hence, the bio-economic modelling approach of this thesis needs to be complemented with farm surveys and non market valuation studies.

The analysis is exploratory rather than a comprehensive assessment of a particular policy. It should be emphasied that given the complexities of the real world and assumptions made in developing the model and asembling the data, the reported results are no more than best estimates However, the results are reasonable, and offer a useful means for comparing alternative scenarios and for reaching general conclusions about alternative policies.

Effective policy implementation requires political acceptability. Gunningham & Sinclair (2005) stated that

"Political acceptability will be crucial to the credibility, legitimacy and success of most policy instruments and unless we take account of it, policy prescriptions are likely to gather dust rather than to be implemented".

The challenge is to devise options that are politically palatable by minimising the tradeoffs between environmental improvement and economic prosperity.

Techniques for the measurement of policies have been successfully developed in order to add significantly to our knowledge of the underlying relationships. The method can be applied elsewhere and for other issues such as greenhouse gases. This thesis provides useful tools for policy makers seeking to develop empirically informed agri environmental policy.

Appendix 1.1 New Zealand Soil Classification subgroups

Source: Hewitt (1998)

NZSC code	Description	NZSC code	Description
Anthropic Soils			
AFA	Artifact Fill Anthropic Soils	AFC	Compacted Fill Anthropic
AFE	Earthy Fill Anthropic Soils	AFST	Stony-tailings Fill Anthropic
AFW	Wet Fill Anthropic Soils	ARB	Buried Refuse Anthropic
ART	Typic Refuse Anthropic Soils	AT	Rocky Truncated Anthropic
ATT	Typic Truncated Anthropic Soils		
Brown Soils			
BAM	Mottled Acid Brown Soils	BAMP	Mottled-placic Acid Brown
BAO	Peaty Acid Brown Soils	BAP	Placic Acid Brown
BAT	Typic Acid Brown Soils	BAX	Pan Acid Brown Soils
BFA	Acidic Firm Brown Soils	BFAL	Acidic-allophanic Firm Brown
BFC	Cemented Firm Brown Soils	BFL	Allophanic Firm Brown Soils
BFM	Mottled Firm Brown Soils	BFMA	Mottled-acidic Firm Brown Soils
BFMC	Mottled-Cemented Firm Brown Soils	BFP	Pallic Firm Brown Soils
BFT	Typic Firm Brown Soils	BLA	Acidic Allophanic Brown Soils
BLAD	Acidic-pedal Allophanic Brown Soils	BLAM	Acidic-mafic Allophanic Brown
BLD	Pedal Allophanic Brown Soils	BLM	Mottled Allophanic Brown Soils
BLT	Typic Allophanic Brown Soils	BLX	Fragic Allophanic Brown Soils
BMA	Acidic Mafic Brown Soils	BMG	Magnesic Mafic Brown Soils
BMM	Mottled Mafic Brown Soils	BMMG	Mottled-magnesic Mafic Brown
BMT	Typic Mafic Brown Soils	BOA	Acidic Orthic Brown Soils
BOC	Calcareous Orthic Brown Soils	BOH	Humose Orthic Brown Soils
BOI	Immature Orthic Brown Soils	BOM	Mottled Orthic Brown Soils
BOMA	Mottled-acidic Orthic Brown Soils	BOP	Pallic Orthic Brown Soils
BOT	Typic Orthic Brown Soils	BSA	Acidic Sandy Brown Soils
BSM	Mottled Sandy Brown Soils	BSP	Pallic Sandy Brown Soils
BST	Typic Sandy Brown Soils	BXT	Typic Oxidic Brown Soils

NZSC code	Description	NZSC code	Description
Gley Soils			
GAG	Granular Acid Gley Soils	GAH	Humose Acid Gley Soils
GAO	Peaty Acid Gley Soils	GAPH	Placic-humose Acid Gley Soils
GAT	Typic Acid Gley Soils	GAY	Ultic Acid Gley Soils
GOA	Acidic Orthic Gley Soils	GOC	Calcareous Orthic Gley Soils
GOE	Melanic Orthic Gley Soils	GOI	Ironstone Orthic Gley Soils
GOJ	Argillic Orthic Gley Soils	GOO	Peaty Orthic Gley Soils
GOQ	Saline Orthic Gley Soils	GOT	Typic Orthic Gley Soils
GRA	Acidic Recent Gley Soils	GRC	Calcareous Recent Gley Soils
GRO	Peaty Recent Gley Soils	GRQ	Saline Recent Gley Soils
GRT	Typic Recent Gley Soils	GSC	Concretionary Sandy Gley Soils
GSO	Peaty Sandy Gley Soils	GST	Typic Sandy Gley Soils
GUF	Fluid Sulphuric Gley Soils	GUFQ	Fluid-saline Sulphuric Gley Soils
GUO	Peaty Sulphuric Gley Soils	GUSQ	Sandy-saline Sulphuric Gley Soil
GXN	Nodular Oxidic Gley Soils	GXT	Typic Oxidic Gley Soils
Allophanic Soils			
LGO	Peaty Gley Allophanic Soils	LGT	Typic Gley Allophanic Soils
LIM	Mottled Impeded Allophanic Soils	LIMI	Mottled-ironstone Impeded Allophanic Soils
LIT	Typic Impeded Allophanic Soils	LOA	Acidic Orthic Allophanic Soils
LOM	Mottled Orthic Allophanic Soils	LOT	Typic Orthic Allophanic Soils
LOV	Vitric Orthic Allophanic Soils	LOVA	Vitric-acidic Orthic Allophanic Soils
LPI	Ironstone Perch-gley Allophanic	LPT	Typic Perch-gley Allophanic Soil
Granular Soils			
NEL	Allophanic Melanic Granular Soils	NEM	Mottled Melanic Granular Soils
NET	Typic Melanic Granular Soils	NOA	Acidic Orthic Granular Soils
NOL	Allophanic Orthic Granular Soils	NOM	Mottled Orthic Granular Soils
NOMA	Mottled-acidic Orthic Granular Soils	NOT	Typic Orthic Granular Soils
NPA	Acidic Perch-gley Granular Soils	NPT	Typic Perch-gley Granular Soils
NPX	Oxidic Perch-gley Granular Soils	NXA	Acidic Oxidic Granular Soils
NXL	Allophanic Oxidic Granular Soils	NXM	Mottled Oxidic Granular Soils
NXMA	Mottled-acidic Oxidic Granular Soils	NXT	Typic Oxidic Granular Soils

NZSC code	Description	NZSC code	Description
Melanic Soils			
EMG	Magnesic Mafic Melanic Soils	EMM	Mottled Mafic Melanic Soils
EMT	Typic Mafic Melanic Soils	EOC	Calcareous Orthic Melanic Soils
EODC	Pedal-calcareous Orthic Melanic	EOJ	Argillic Orthic Melanic Soils
EOJC	Argillic-calcareous Orthic Melanic	EOM	Mottled Orthic Melanic Soils
EOMC	Mottled-calcareous Orthic Melanic Soils	EOT	Typic Orthic Melanic Soils
EPJ	Argillic Perch-gley Melanic Soils	EPT	Typic Perch-gley Melanic Soils
EPV	Vertic Perch-gley Melanic Soils	ERO	Peaty Rendzic Melanic Soils
ERT	Typic Rendzic Melanic Soils	ERW	Weathered Rendzic Melanic Soil
EVC	Calcareous Vertic Melanic Soils	EVM	Mottled Vertic Melanic Soils
EVT	Typic Vertic Melanic Soils		
Organic Soils			
OFA	Acid Fibric Organic Soils	OFM	Mellow Fibric Organic Soils
OFS	Sphagnic Fibric Organic Soils	OHA	Acid Humic Organic Soils
OHM	Mellow Humic Organic Soils	OLBG	Burried-gley Litter Organic Soils
OLBZ	Burried-podzol Litter Organic Soils	OLO	Orthic Litter Organic Soils
OMA	Acid Mesic Organic Soils	OMM	Mellow Mesic Organic Soils
Oxidic Soils			
XNT	Typic Nodular Oxidic Soils	XOB	Brown Orthic Oxidic Soils
XOM	Mottled Orthic Oxidic Soils	XOT	Typic Orthic Oxidic Soils
XPN	Nodular Perch-gley Oxidic Soils	XPT	Typic Perch-gley Oxidic Soils
Pallic Soils			
PIC	Calcareous Immature Pallic Soils	PID	Pedal Immature Pallic Soils
PIM	Mottled Immature Pallic Soils	PIMD	Mottled-pedal Immature Pallic
PIT	Typic Immature Pallic Soils	PJA	Aged Argillic Pallic Soils
PJC	Calcareous Argillic Pallic Soils	PJM	Mottled Argillic Pallic Soils
PJN	Sodic Argillic Pallic Soils	PJT	Typic Argillic Pallic Soils
PLC	Calcareous Laminar Pallic Soils	PLM	Mottled Laminar Pallic Soils
PLT	Typic Laminar Pallic Soils	PPC	Cemented Perch-gley Pallic Soils
PPJ	Argillic Perch-gley Pallic Soils	PPJX	Argillic-fragic Perch-gley Pallic
PPT	Typic Perch-gley Pallic Soils	PPU	Duric Perch-gley Pallic Soils
PPX	Fragic Perch-gley Pallic Soils	PUJ	Argillic Duric Pallic Soils

NZSC code	Description	NZSC code	Description
Pallic Soils			
PUT	Typic Duric Pallic Soils	PXC	Calcareous Fragic Pallic Soils
PXCN	Calcareous-sodic Fragic Pallic Soils	PXJ	Argillic Fragic Pallic Soils
PXJM	Argillic-mottled Fragic Pallic Soils	PXJN	Argillic-sodic Fragic Pallic Soils
PXM	Mottled Fragic Pallic Soils	PXMC	Mottled-calcareous Fragic Pallic
PXT	Typic Fragic Pallic Soils		
Podzols			
ZDH	Humose Densipan Podzols	ZDQ	Ortstein Densipan Podzols
ZDT	Typic Densipan Podzols	ZDU	Humus-pan Densipan Podzols
ZDYH	Ultic-humose Densipan Podzols	ZGH	Humose Groundwater-gley Podzols
ZGT	Typic Groundwater-gley Podzols	ZOH	Humose Orthic Podzols
ZOT	Typic Orthic Podzols	ZPF	Fluid Perch-gley Podzols
ZPH	Humose Perch-gley Podzols	ZPHP	Humose-placic Perch-gley Podzols
ZPHQ	Humose-ortstein Perch-gley Podzols	ZPOZ	Peaty-silt-mantled Perch-gley Podzols
ZPP	Placic Perch-gley Podzols	ZPQ	Ortstein Perch-gley Podzols
ZPT	Typic Perch-gley Podzols	ZPU	Humus-pan Perch-gley Podzols
ZPZ	Silt-mantled Perch-gley Podzols	ZXF	Firm Pan Podzols
ZXH	Humose Pan Podzols	ZXP	Placic Pan Podzols
ZXQ	Ortstein Pan Podzols	ZXU	Humus-pan Pan Podzols
ZXX	Fragic Pan Podzols		
Pumice Soils			
MIM	Mottled Impeded Pumice Soils	MIMW	Mottled-welded Impeded Pumice Soils
MIT	Typic Impeded Pumice Soils	MIW	Welded Impeded Pumice Soils
MOBL	Buried-allophanic Orthic Pumice Soils	MOI	Immature Orthic Pumice Soils
MOL	Allophanic Orthic Pumice Soils	MOM	Mottled Orthic Pumice Soils
MOT	Typic Orthic Pumice Soils	MOZ	Podzolic Orthic Pumice Soils
MPT	Typic Perch-gley Pumice Soils	MPU	Duric Perch-gley Pumice Soils
Raw Soils			
WF	Fluvial Raw Soils	WGF	Fluid Gley Raw Soils
WGFQ	Fluid-saline Gley Raw Soils	WGFU	Fluid-sulphidic Gley Raw Soils
WGQ	Saline Gley Raw Soils	WGS	Sandy Gley Raw Soils

NZSC code	Description	NZSC code	Description
	Raw Soils		
WGT	Typic Gley Raw Soils	WGU	Sulphidic Gley Raw Soils
WHA	Active Hydrothermal Raw Soils	WO	Orthic Raw Soils
WS	Sandy Raw Soils	WT	Tephric Raw Soils
WX	Rocky Raw Soils		
	Recent Soils		
RFA	Acidic Fluvial Recent Soils	RFAW	Acid-weathered Fluvial Recent Soils
RFMA	Mottled-acidic Fluvial Recent Soils	RFMQ	Mottled-saline Fluvial Recent Soils
RFMW	Mottled-weathered Fluvial Recent Soils	RFQ	Mottled Fluvial Recent Soils
RFQ	Saline Fluvial Recent Soils	RFT	Typic Fluvial Recent Soils
RFW	Weathered Fluvial Recent Soils	RHI	Inactive Hydrothermal Recent Soils
ROA	Acidic Orthic Recent Soils	ROAW	Acid-weathered Orthic Recent Soils
ROM	Mottled Orthic Recent Soils	ROMP	Mottled-pallic Orthic Recent Soils
ROT	Typic Orthic Recent Soils	ROW	Weathered Orthic Recent Soils
RSA	Acidic Sandy Recent Soils	RSM	Mottled Sandy Recent Soils
RST	Typic Sandy Recent Soils	RTBL	Buried-allophanic Tephric Recent Soils
RTBP	Buried-pumice Tephric Recent Soils	RTM	Mottled Tephric Recent Soils
RTT	Typic Tephric Recent Soils	RXA	Acidic Rocky Recent Soils
RXOA	Peaty-acidic Rocky Recent Soils	RXT	Typic Rocky Recent Soils
	Semiarid Soils		
SAH	Thick Aged-argillic Semiarid Soils	SAK	Alkaline Aged-argillic Semiarid Soils
SAT	Typic Aged-argillic Semiarid Soils	SAW	Weathered Aged-argillic Semiarid Soils
SIK	Alkaline Immature Semiarid Soils	SIM	Mottled Immature Semiarid Soils
SIQ	Saline Immature Semiarid Soils	SIT	Typic Immature Semiarid Soils
SJK	Alkaline Argillic Semiarid Soils	SJL	Laminar Argillic Semiarid Soils
SJM	Mottled Argillic Semiarid Soils	SJQ	Saline Argillic Semiarid Soils
SJT	Typic Argillic Semiarid Soils	SZQ	Saline Solonetzic Semiarid Soils
SZT	Typic Solonetzic Semiarid Soils		

NZSC code	Description	NZSC code	Description
		Ultic Soils	
UDM	Mottled Densipan Ultic Soils	UDP	Perch-gleyed Densipan Ultic Soils
UEM	Mottled Albic Ultic Soils	UEP	Perch-gleyed Albic Ultic Soils
UEY	Yellow Albic Ultic Soils	UPS	Sandy Perch-gley Ultic Soils
UPT	Typic Perch-gley Ultic Soils	USE	Albic Sandy Ultic Soils
USM	Mottled Sandy Ultic Soils	UST	Typic Sandy Ultic Soils
UYG	Magnesic Yellow Ultic Soils	UYM	Mottled Yellow Ultic Soils
UYMZ	Mottled-podzolic Yellow Ultic Soils	UYT	Typic Yellow Ultic Soils
UYZ	Podzolic Yellow Ultic Soils		

Appendix 1.2 Slope classes

Source: Newsome, Wilde, & Willoughby (2000)

Description: Polygon layer delineating physiographic areas of relatively homogeneous average slope class.

Origin: Derived from stereo aerial photograph interpretation, field verification and measurement as part of the 1:63 360/1:50 000 scale New Zealand Land Resource Inventory survey.

Item code	*Class description*	*Class range*
A	Flat to gently undulating	0–3°
B	Undulating	4–7°
C	Rolling	8–15°
D	Strongly rolling	16–20°
E	Moderately steep	21–25°
F	Steep	26–35°
G	Very steep	>35° (36–42°)[1]
H	Precipitous	(>42°)[1]
estu	estuary	
ice	icefield	
lake	lake	
quar	quarry, mine, other earthworks	
rive	river	
town	urban area, airport, oxidation pond	

The ArcInfo 'world polygon' has a null value, otherwise all records contain values from the list above.

Interpretation: Examples:

C denotes an area of dominantly rolling slopes between 8 and 15°

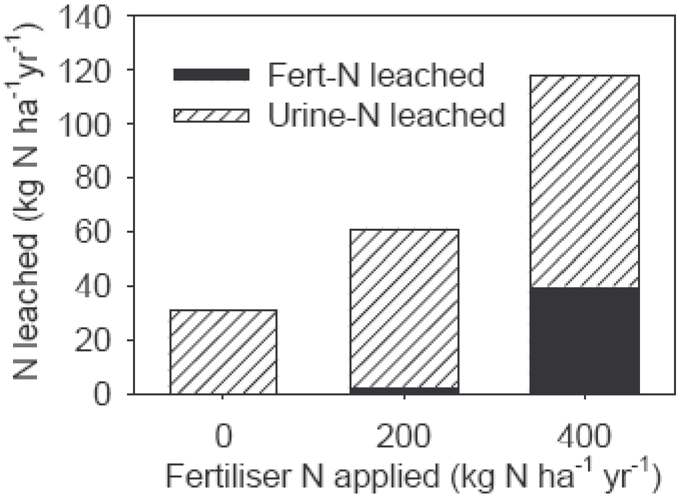
E +F denotes an area of compound slope, dominantly 21–25° but with some significant slopes of 26–35°

D /E denotes an area where average slope is intermediate between strongly rolling and moderately steep

A' denotes virtually flat land dissected by gullies or terrace edges

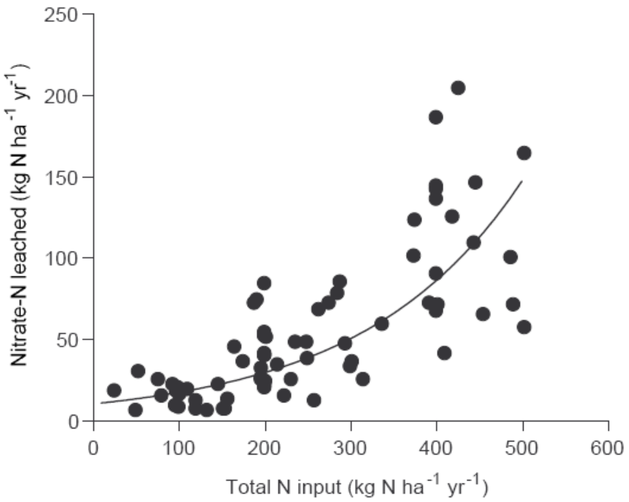
Appendix 2.1 Composition of nitrogen discharge

Following graph illustrates the effect of rate of nitrogen application on nitrate leaching in dairy pasture stocked at 3.3 cows per ha



Source: Ledgard, Penno, & Sporsen (1999)

Appendix 2.2 Effect of nitrogen input on nitrate leaching



Source: Menneer, Ledgard, & Gillingham (2004)

Appendix 3.1 Dairy Operating Profit

Dairy Operating Profit (formerly known as Economic Farm Surplus (EFS)) is a simple benchmarking tool used to indicate dairy farm profitability. In order to compare your level of profitability with other dairy farms it is important to include non-cash items as well as cash. The guidelines described in this FarmFact are based on the industry standard as set by DairyBase.

Operating Profit is the Operating Surplus from cash (Dairy GFR – Operating Expenses) after adjustments have been made for:

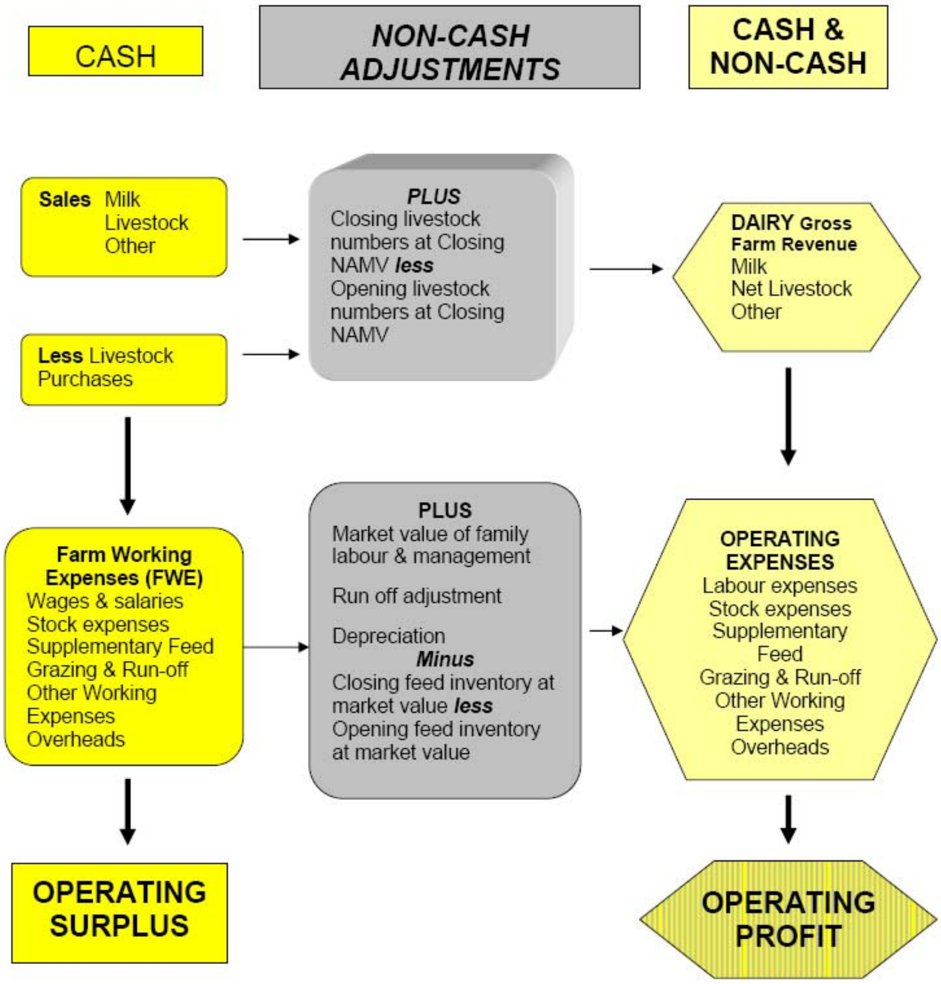
- The value change in livestock numbers
- Unpaid labour and management (Labour adjustment)
- The ownership of run-offs (Run-off adjustment)
- Depreciation
- The value change in supplementary feed inventory

Operating profit

	Milk & Dairy Livestock Stock Sales	
+	**Other Dairy income**	
+/-	**Value of Change in Dairy Livestock Numbers**	
=	**Dairy Gross Farm Revenue**	
-	**Operating Expenses**	
	Farm Working Expenses	(Cash)
-	**Feed Inventory Adjustment**	(Closing feed less opening feed inventory)
+	**Owned Runoff Adjustment**	(if runoff is owned and not leased)
+	**Labour Adjustment**	(for unpaid family management and labour)
+	**Depreciation**	
=	**Dairy Operating Profit (EFS)**	

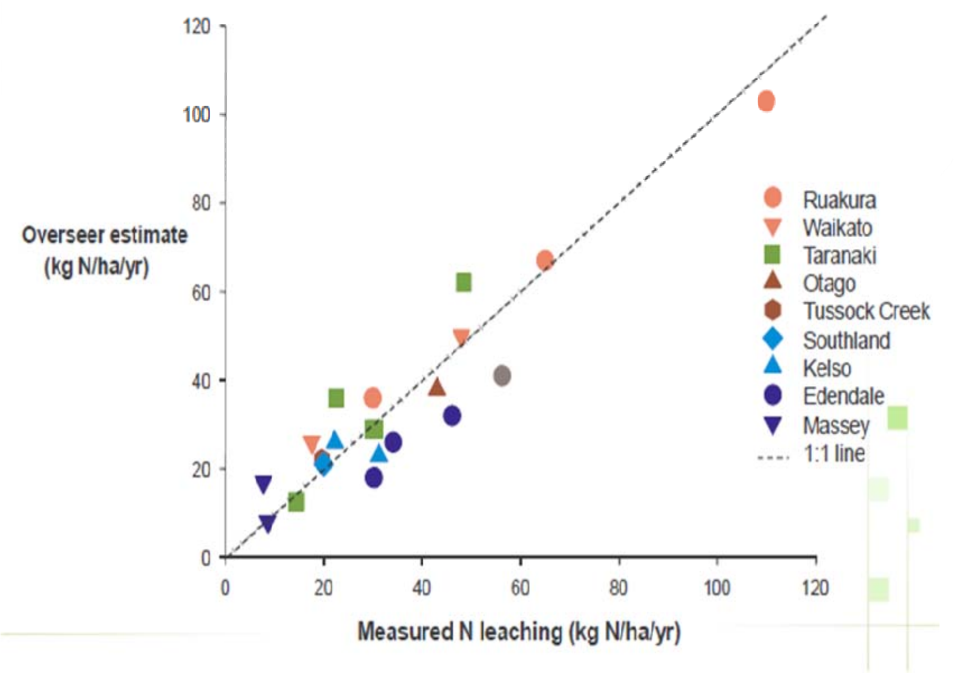
50% Sharemilkers - use your own set of accounts to calculate Operating Profit. This can be compared with other 50% sharemilkers, but not farm owners.
Variable Order Sharemilkers - Combine the accounts of the farm owner and the sharemilker to calculate Operating Profit.

This can be compared with other owner-operator farms. The following diagram shows the adjustments made to the cash income and expense to calculate Operating Profit.



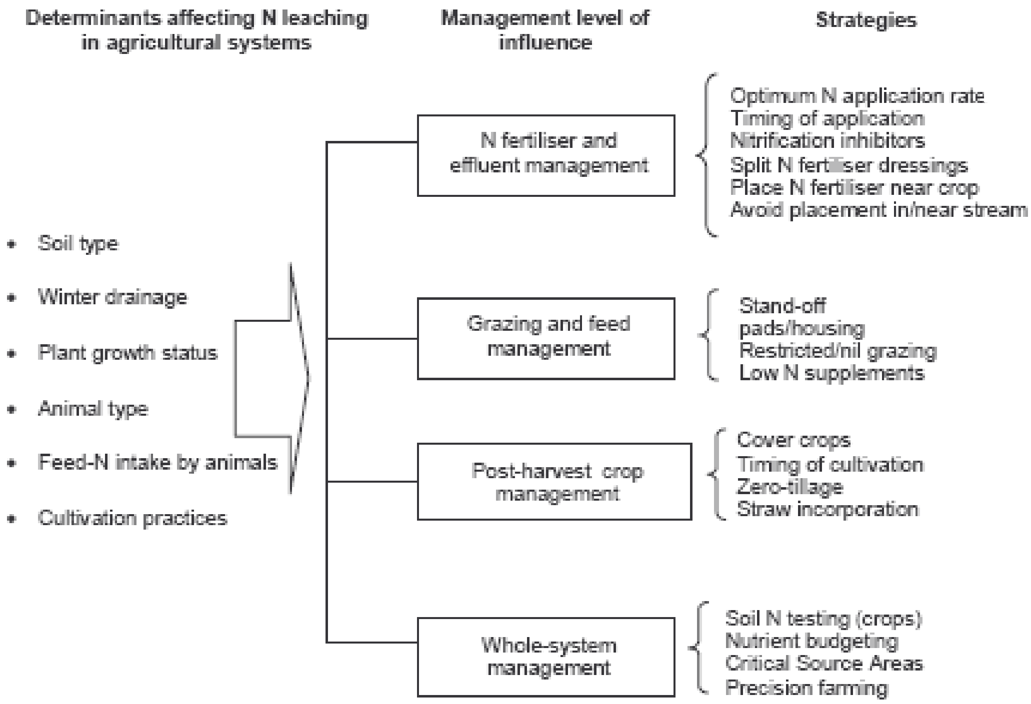
Source: DairyNZ (2008)

Appendix 4.1 *Overseer* validation in Dairy farm systems



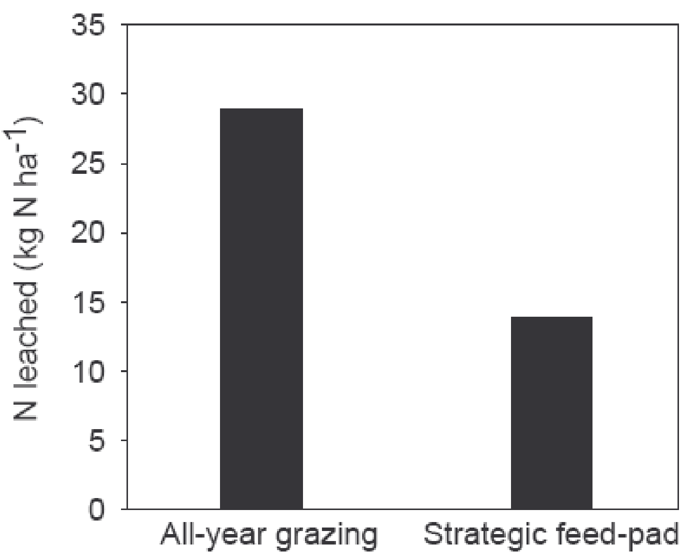
Source: Agresearch (2009)

Appendix 5.1 Overview of best management practices



Source: Menneer, Ledgard, & Gillingham (2004)

Appendix 5.2 Effectiveness of feed pad



Source: Chadwick, Ledgard, & Brown (2002)- a farmlet study in Taranaki

References

Adamowicz, W. (2007). Reflections on environmental policy in Canada. *Canadian Journal of Agricultural Economics, 55*(1), 1-13.
Aftab, A., Hanley, N., & Kampas, A. (2007). Co-ordinated environmental regulation: controlling non-point nitrate pollution while maintaining river flows. *Environmental and Resource Economics, 38*(4), 573-593.
Agresearch. (2009). *An Introduction to the Overseer® Nutrient Budgets Model Version 5.4*. Hamilton.
Agribusiness Group, Agresearch, NIWA, Crop and Food, & Aqualinc. (2007). *Impact of Management Changes on Farm Profitability and Environmental Outcomes*. Christchurch.
Alfrod, A. R., Cacho, O. J., & Griffth, G. R. (2006). *A Bio-Economic Model of a Northern Tablelands Cattle Grazing Enterprise. III. Alternative Optimisation Procedures*. Armidale: University of New England.
Allen, D. W. (1991). What are transaction costs? *Research in Law and Economics, 14*, 1-18.
Allen, D. W., & Lueck, D. (1999a). The role of risk in contract choice. *Journal of Law Economics and Organization., 15*(3), 704-736.
Allen, D. W., & Lueck, D. (1999b). Searching for ratchet effects in agricultural contracts. *Journal of Agricultural and Resource Economics, 24*(2), 536-552.
Allen, D. W., & Lueck, D. (2002). *The Nature of the Farm: Contracts, risk, and organization in agriculture*. Massachusetts: The MIT Press.
Allison, P. (2002). *Missing Data*. Thousand Oaks, CA: Sage.
Anderson, T. L. (2004). Donning Coase -coloured glasses: a property rights view of natural resource economics. *The Australian Journal of Agricultural and Resource Economics, 48*(3), 445-462.
Asmild, M., & Hougaard, J. L. (2006). Economic versus environmental improvement potentials of Danish pig farms. *Agricultural Economics, 35*(2), 171-181.
Atasoy, M., Palmquist, R. B., & Phaneuf, D. J. (2006). Estimating the effects of urban residential development on water quality using microdata. *Journal of Environmental Management, 79*(4), 399-408.
Atkinson, S. E., & Morton, B. J. (2004). Determining the cost-effective size of an emission trading region for achieving an ambient standard. *Resource and Energy Economics, 26*(3), 295-315.
Ball, R. P., & Field, T. R. O. (1982). Responses to nitrogen as affected by pasture characteristics, season and grazing management. In P. B. Lynch (Ed.), *Nitrogen Fertilizers in New Zealand*. Wellington: New Zealand Institute of Agricultural Science.
Ballas, D., Clarke, G. P., Dorling, D., Rigby, J., & Wheeler, B. (2006). Using geographical information systems and spatial microsimulation for the analysis of health inequalities. *Health Informatics Journal, 12*(1), 65-79.
Ballas, D., Clarke, G. P., & Wiemeres, E. (2006). Spatial microsimulation for rural policy analysis in Ireland: The implications of CAP reforms for national spatial strategy. *Journal of Rural Studies, 22*(3), 367-378.

Bartolini, F., Gallerani, V., Raggi, M., & Viaggi, D. (2007). Implementing the water framework directive: contract design and the cost of measures to reduce nitrogen pollution from agriculture. *Environmental Management, 40*(4), 567-577.

Bateman, I. J., Ennew, C., Lovett, A., & Rayner, A. J. (1999). Modelling and mapping agricultural output values using farm specific details and environmental database. *Journal of Agricultural Economics, 50*(3), 488-511.

Baum, C. F. (2006). *An Introduction to Modern Econometrics Using Stata*. Texas: Stata Press.

Baumol, W. J., & Oates, W. E. (1988). *The Theory of Environmental Policy* (Second ed.). New York: Cambridge University Press.

Bayfield, M. A., & Meister, A. D. (2005). *Review of the East Coast Forestry Project, Discussion Paper No. 37*. Wellington: Ministry of Agriculture and Forestry.

Beard, S. (2007). *Regional Rivers Water Quality Monitoring Programme:Data Report 2006*. Hamilton: Environment Waikato Technical Report 2007/12.

Becher, K. D., Kalkhoff, S. J., Schnoebelen, D. J., Barnes, K. K., & Miller, V. E. (2001). *Water-quality assessment of the Eastern Iowa Basins--Nitrogen, phosphorus, suspended sediment, and organic carbon in surface water, 1996-98*. Washington.

Bedard-Haughn, A., Tate, K. W., & Kessel, C. (2004). Using nitrogen-15 to quantify vegetative buffer effectiveness for sequestering nitrogen in runoff. *Journal of Environmental Quality, 33*, 2252-2262.

Bennett, J. (2005). Australasian environmental economics: contributions, conflicts and 'cop-outs'. *The Australian Journal of Agricultural and Resource Economics, 49*(3), 243-261.

Berntsen, J., Petersen, B. M., Jacobsen, B. H., Olesen, J. E., & Hutchings, N. J. (2003). Evaluating nitrogen taxation scenarios using the dynamic whole farm simulation model FASSET. *Agricultural Systems, 76*(3), 817-839.

Beukes, P., Palliser, C., Lancaster, J., Leydon-Davis, J., Levy, G., Folkers, C., et al. (2006). *Predicting cow production based on an estimate of animal genotype within a Whole Farm Model.* Paper presented at the New Zealand Society of Animal Production, Napier.

Beukes, P., Palliser, C., Prewer, W., Serra, V., Lacaster, J., Levy, G., et al. (2005). *Use of a Whole Farm Model for exploring management decisions in dairying.* Paper presented at the International Congress on Modelling and Simulation, Melbourne.

Bontems, P. (2007). On environmental policy and permitting. *Journal of Public Economic Theory, 9*(5), 771-792.

Bontems, P., Rotillon, G., & Turpin, N. (2005). Self-Selecting Agri-environmental policies with an application to the Don watershed. *Environmental and Resource Economics, 31*(3), 275-301.

Borisova, T., Shortle, J., Horan, R. D., & Abler, D. G. (2005). Value of information for water quality management. *Water Resources Research, 41*(W06004), 1-11.

Bousquet, F., Bakam, I., Proton, H., & Le Page, C. (1998). Cormas: Common-pool resources and multi-agent systems. In *Tasks and Methods in Applied Artificial Intelligence* (pp. 826-837).

Boyd, J. (2000). The new face of the clean water act: a critical review of the EPA's new TMDL rules. *Duke Environmental Law and Policy Forum, 11*, 39.

Braden, J. B., & Segerson, K. (1993). Information problems in the design of nonpoint-source pollution policy. In C. S. Russell & J. F. Shogren (Eds.), *Theory, Modeling and Experience in the Management of Nonpoint-Source Pollution* (pp. 1–36.). Boston, MA Kluwer Academic Publishing.
Brady, M. (2003). The relative cost-efficiency of arable nitrogen management in Sweden. *Ecological Economics, 47*(1), 53-70.
Brodnax, R. (2006). The Environment Waikato approach to managing agriculture's impact on the environment. *Primary Industry Management, 9*(2), 7-10.
Bromley, D. W. (1989). *Economic Interests and Institutions; The Conceptual Foundations of Public Policy*. Oxford: Basil Blackwell.
Buchan, D., Meister, A. D., & Giera, N. (2006). *Bridging the Gap Between Environmental Knowledge and Research, and Desired Environmental Outcomes to Achieve Stainable Land Management*. Wellington: Ministry of Agriculture and Forestry.
Burrows, P. (1979). Pigovian taxes, polluter subsidies, regulation, and the size of a polluting industry. *Canadian Journal of Economics, 12*(3), 494-501.
Cabrera, V. E., Breuer, N. E., Hildebrand, P. E., & Letson, D. (2005). The dynamic North Florida dairy farm model: A user-friendly computerized tool for increasing profits while minimizing N leaching under varying climatic conditions. *Computers and Electronics in Agriculture, 49*(2), 286-308.
Carpentier, C. L., Bosch, D. J., & Batie, S. S. (1998). Using the spatial information to reduce costs of controlling agricultural nonpoint source pollution. *Agricultural and Resource Economics Review, 27*(1), 72-84.
Chadwick, D. R., Ledgard, S. F., & Brown, L. (2002). *Nitrogen flows and losses in dairy farms in New Zealand and the UK: Effects of grazing management.* Paper presented at the Proceedings of the workshop on Dairy Farm Soil Management, Palmerston North.
Chalmers, N., & Crabtree, B. (1999). *Modelling policy impacts on nitrate leaching and farm incomes in the Ythan Catchment*. Aberdeen: Macaulay Land Use Research Institute.
Chambers, R. G., Fare, R., Jaenicke, E., & Lichtenberg, E. (1998). Using dominance in forming bounds on DEA models: The case of experimental agricultural data. *Journal of Econometrics, 85*(1), 189-203.
Charnes, A., Cooper, W. W., & Rhodes, E. (1978). Measuring the efficiency of decision making units. *European Journal of Operational Research, 2*, 429-444.
Chatterjee, S., & Hadi, A. S. (2006). *Regression Analysis by Example, 4th Edition*: John Wiley & Sons.
Choe, C., & Fraser, I. (1999). Compliance monitoring and agri-environmental policy. *Journal of Agricultural Economics, 50*(3), 468-487.
Clark, D. A. (2007). Issues and options for future dairy farming in New Zealand. *New Zealand Journal of Agricultural Research, 50*, 203-221.
Coase, R. H. (1937). The nature of firm. *Economica, 4* (16), 386-405.
Coase, R. H. (1960). The problem of social cost. *Journal of Law and Economics, 3*, 1-44.
Coase, R. H. (1974). The lighthouse in economics. *Journal of Law and Economics, 17*, 357-376.
Coelli, T. (1995). Recent developments in frontier modeling and efficiency measurement. *Australian Journal of Agricultural and Resource Economics, 39*(3), 219-245.

Coelli, T., Lauwers, L., & Van Huylenbroeck, G. (2007). Environmental efficiency measurement and the materials balance condition. *Journal of Productivity Analysis, 28*(1-2), 3-12.
Coelli, T. J., Rao, D. S. P., O'Donnell, C. J., & Battese, G. E. (2005). *An Introduction to Efficiency and Productivity Analysis* (2nd ed.). New York: Springer.
Cole, D. H., & Grossman, P. Z. (2002). The meaning of property rights: law versus economics. *Land Economics, 78*(3), 317-330.
Collentine, D. (2006). Composite market design for a Transferable Discharge Permit (TDP) system. *Journal of Environmental Planning and Management, 49*(6), 929-946.
Collier, K. J., Davies-Colley, R. J., Rutherford, J. C., Smith, C. M., & Williamson, R. B. (1995). *Managing Riparian Zones : a contribution to protecting New Zealand's rivers and streams*. Wellington: Department of Conservation.
Cook, H. F., & Norman, C. (1996). Targeting agri-environmental policy: An analysis relating to the use of Geographical Information Systems. *Land Use Policy, 13*(3), 217-228.
Cooper, J. C., & Keim, R. W. (1998). Incentive payments to encourage farmer adoption of water quality protection practices. *American Jouranl of Agricultural Economics, 78*(1), 54-64.
Cooper, W. W., Seiford, L. M., & Tone, K. (2000). *Data Envelopment Analysis : a comprehensive text with models, applications, references, and DEA-Solver* Boston: Kluwer Academic Publishers.
Cullen, R., Hughey, K., & Kerr, G. (2006). New Zealand freshwater management and agricultural impacts. *The Australian Journal of Agricultural and Resource Economics, 50*(3), 327-346.
Dairy Environment Review Group. (2006). *Dairy Industry Strategy for Sustainable Environmental Management*. Wellington: Dairy Insight.
DairyInsight. (2006). The Statement, Newsletter of dairy industry issue 5 April . Retrieved September, 01, 2008, from http://www.dairynz.co.nz/file/fileid/5112
DairyNZ. (2008). Dairy Operating Profit Farm Fact 9.3 Retrieved August, 30, 2008, from http://www.dairynz.co.nz/file/fileid/9288
Dake, C. K. G. (2007). *Modelling nitrogen discharge trading using spatial multi-agent simulation.* Paper presented at the MODSIM 2007 International Congress on Modelling and Simulation Christchurch.
Dake, C. K. G., Mackay, A. D., & Manderson, A. K. (2005). *Optimal trade-offs between financial and environmental risks in pastoral farming.* Paper presented at the MODSIM 2005 International Congress on Modelling and Simulation, Melbourne.
Davies-Colley, R. J., Smith, R. A., Young, R., & Phillips, C. J. (2004). Water quality impact of dairy cow herd crossing a stream. *New Zealand Journal of Marine and Freshwater Research, 38*, 596-576.
Davis, M. (2005). *Nutrient Lossess from Forestry in the Lake Taupo Catchment*. Hamilton: Environment Waikato
De Cara, S., Houze, B. M., & Jayet, P. A. (2005). Methane and nitrous oxide emissions from agriculture in the EU: A spatial assessment of sources and abatement costs. *Environmental and Resource Economics, 32*(4), 551-583.
De Klein, C. A. M., & Monaghan, R. M. (2005). The impact of potential nitrous oxide mitigation strtategies on the environmental and economic performance of dairy systems in four New Zealand catchments. *Environmental Sciences, 2*, 351-360.

De Koeijer, T. J., Wossink, G. A. A., Struik, P. C., & Renkema, J. A. (2002). Measuring agricultural sustainability in terms of efficiency: the case of Dutch sugar beet growers. *Journal of Environmental Management, 66*(1), 9-17.

Decoster, A., Standaert, I., Valenduc, C., & Van Camp, G. (1998.). *Evaluation of simultaneous reforms in personal income taxes and indirect taxes: Belgium 1988-1993', Final Report Project PE/VA/07*. Brussels: DWTC.

Decoster, A., & Van Camp, G. (2000). The unit of analysis in microsimulation models for personal income taxes: fiscal unit or household? In L. Mitton, H. Sutherland & M. Weeks (Eds.), *Microsimulation Modelling for Policy Analysis: Challenges and Innovations* Cambridge: Cambridge University Press.

Demetz, H. (1967). Towards a theory of property rights. *The American Economic Review 57*(2), 347-359.

Demetz, H. (1968). The cost of transacting. *Quarterly Journal of Economics, 82*, 33-53.

Denne, T. (2006). *Economic Instruments for the Environment, Technical Report 2006/23*. Hamilton: Environment Waikato.

Depres, C., Grolleau, G., & Mzoughi, N. (2007). Contracting for environmental property rights: the case of Vittel. *Economica, 75*(299), 412-434.

Dexcel. (2006). *Farm fact 4-1 Understanding Milksolids*. Hamilton.

Dexcel. (2007a). *Dairy Operating Profit 2005/06, Farm Fact 7-3*. Hamilton.

Dexcel. (2007b). *A Guide to Managing Farm Dairy Effluent*. Hamilton.

Dhungana, B. R., Nuthall, P. L., & Nartea, G. V. (2004). Measuring the economic inefficiency of Nepalese rice farms using data envelopment analysis. *The Australian Journal of Agricultural and Resource Economics, 48*(2), 347-369.

Di, H. J., & Cameron, K. C. (2002). Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. *Nutrient Cycling in Agroecosystems, 64*(3), 237-256.

Dosskey, M. G., Helmers, M. J., & Eisenhauer, D. E. (2008). A design aid for determining width of filter strips. *Journal of Soil and Water Conservation, 63*(4), 232-242.

Edmeades, D. C. (2004). *Nitrification and Urease Inhibitors, Environmental Publication 2004/11*. Tauranga: Environment Bay of Plenty

Eigenraam, M., Strappazzon, L., Lansdell, N., Beverly, C., & Stoneham, G. (2007). Designing frameworks to deliver unknown information to support market-based instruments. *Agricultural Economics, 37*(1), 261-269.

Ekman, S. (2005). Cost-effective nitrogen leaching reduction as influenced by linkages between farm-level decisions. *Agricultural Economics, 32*(3), 297-309.

Elliott, A. H., Alexander, R. B., Schwarz, G. E., Shankar, U., Sukias, J. P. S., & McBride, G. B. (2005). Estimation of nutrient sources and transport for New Zealand using the hybrid mechanisticstatistical model SPARROW. *Journal of Hydrology (NZ) 44*(1), 1-27.

Environment Waikato. (2004). *Clean Streams- A Guide to Managing Waterways on Waikato Farms* Hamilton.

Environment Waikato. (2005a). *Land Use Changes in the Waikato Catchment Between Taupo Gates and Karapiro, Report to Policy and Strategy Committee*. Hamilton.

Environment Waikato. (2005b). *Proposed Waikato Regional Plan Variation 5 – Lake Taupo Catchment: Analysis of alternatives, benefits and costs under section 32 of the RMA, Policy Series 2005/03*. Hamilton.
Environment Waikato. (2005c). *Setting Water Quality Targets for Nutrients in Nutrient Management catchments, Report to Policy and Strategy Committee*. Hamilton.
Environment Waikato. (2007a). *Recommendations from Catchment Liaison Subcommittees May 2007 Round, Report to Catchment Services Committee*. Hamilton.
Environment Waikato. (2007b). *Environmental Information for the Upper Waikato Policy Review, Report to Policy and Strategy Committe* Hamilton.
Environment Waikato. (2007c). *Waikato Regional Plan*. Hamilton.
Environment Waikato. (2008). *The Condition of Rural Water and Soil in the Waikato Region- Risks and Opportunities* Hamilton.
Environmental Defence Society. (2007). *Submissions and Evidence on Waikato Regional Plan Variation 5- Lake Taupo Catchment.* Hamilton.
ESRI. (2005). ArcGIS: Release 9.1 [software] Redlands, California Environmental Systems Research Institute.
ESRI. (2007). GIS Online Dictionary. Retrieved August 07, 2008
Fais, A., Nino, P., & Giampalo, A. (2005). *Microeconomic and geo-physical data integration for agri-environmental analysis, georeferencing FADN data: a case study in Italy.* Paper presented at the The Future of Rural Europe in the Global Agri-Food System, Xith seminar of the European Association of Agricultural Economists., Copenhagen, Denmark.
Falconer, K. (2000). Farm-level constraints on agri-environmental scheme participation: a transactional perspective. *Journal of Rural Studies, 16*(3), 379-394.
Falconer, K., & Hodge, I. (2001). Pesticied taxation and multi-objective policy - making : farm modelling to evaluate profit/ environmen trade-offs. *Ecological Economics, 36*, 263-279.
Falconer, K., & Whiteby, M. (1999). *The hidden costs of countryside stewardship policies: investigating policy administration and transaction costs in eight European member states.* Paper presented at the Agricultural Economics Society Annual Conference, Belfast.
Fang, F., Easter, K. W., & Brezonik, P. L. (2005). Point nonpoint source water quality trading: A case study in the Minnesota River Basin. *Journal of American Water Resources Asso, 41*(3), 645-657.
Fang. F, Kieser, M. S., Hall, D. L., Ott, N. C., & Hippensteel, S. C. (2005). *Preliminary analysis of water quality trading opporunities in the Great Miami River Watershed, OHIO.* Paper presented at the Watershed Management to Meet Water Quality Standards and Emerging TMDL (Total Maximum Daily Load), Atlanta, Georgia USA).
Fare, R., Grosskopf , S., & Lovell, C. A. K. (1985). *The Measurement of Efficiency of Production*. Boston: Kluwer.
Fare, R., Grosskopf, S., & Lowell, K. (1994). *Production Frontiers*. Cambridge: Cambridge University Press.
Fare, R., Grosskopf, S., & Pasurka Jr, C. A. (2007). Pollution abatement activities and traditional productivity. *Ecological Economics, 62*(3-4), 673-682.
Fare, R., Grosskopf, S., & Tyteca, D. (1996). An activity analysis model of the environmental performance of firms--application to fossil-fuel-fired electric utilities. *Ecological Economics, 18*(2), 161-175.

Farzin, Y. H., & Kaplan, J. D. (2004). Nonpoint source pollution control under incomplete and costly information. *Environmental and Resource Economics, 28*(4), 489-506.

Fennessy, M. S., & Cronk, J. K. (1997). The effectiveness and restoration potential of riparian ectones for the management of nonpoint source pollution, particularly nitrate. *Critical Reviews in Environmental Science and Technology, 27*, 285-317.

Feyter, C., O'connor, M. B., & Addison, B. (1985). Effects of rates and times of nitrogen application on the production and composition of dairy pastures in Waikato district, New Zealand. *New Zealand Journal of Experimental Agriculture, 13*, 247- 252.

Fischer, C., & Newell, R. G. (2008). Environmental and technology policies for climate mitigation. *Journal of Environmental Economics and Management, 55*(2), 142-162.

Fleming, R. A., & Adams, R. M. (1997). The importance of site-specific information in the design of policies to control pollution. *Journal of Environmental Economics and Management, 33*(3), 347-358.

Fraser, I., & Cordina, D. (1999). An application of data envelopment analysis to irrigated dairy farms in Northern Victoria, Australia. *Agricultural Systems, 59*(3), 267-282.

Fraser, R. (2004). On the use of targeting to reduce moral hazard in agri-environmental schemes. *Journal of Agricultural Economics, 55*(3), 525-540.

Frontline Systems. (2007). Premium Solver Platform for Excel, www.solver.com.

GAMS/MINOS. (2001). *GAMS- The Solver Manuals (MINOS)*. Washignton, DC: GAMS Development Corporation.

Gangadharan, L. (2000). Transaction costs in pollution markets:an empirical study. *Land Economics, 76*(4), 601-604.

Gilbert, R. O. (1987). *Statistical Methods for Environmental Pollution Monitoring*. New York: Van Nostrand Reinhold Company.

Gilliam, J. W. (1994). Riparian wetlands and water quality. *Journal of Environmental Quality, 23*(3), 896-900.

Gilliam, J. W., Parsons, J. E., & Mikkelsen, R. L. (1997). Nitrogen dynamics and buffer zones. In N. E. Haycok (Ed.), *Buffer zones : their process and potential in water protection. Quest Environmental* (pp. 54-61). UK: Harpenden.

Giraldez, C., & Fox, G. (1995). An economic analysis of ground water contamination from agricultural nitrate emissions. *Canadian Journal of Agricultural Economics, 43*, 387-402.

Glassey, C. (2001). *Productivity and growth - does 14% growth equal 4% productivity gain?* Paper presented at the South Island Dairy Event Christchurch, SIDE.

Gleeson, T., & Carruthers, G. (2006). What could EMS offer land management in rural Australia. *Farm Policy Journal, 31*(4), 1-13.

Goetz, R. U., Schmidt, H., & Lehmann, B. (2006). Determining the economic gains from regulation at the extensive and intensive margins. *European Review of Agricultural Economics, 33*(1), 1-30.

Gordon, S. (2003). *Regulatory design for water quality management in Perth, Western Australia: Economic Instruments Series: Paper 2—Innovative economic instruments for nonpoint source water pollution in the Swan-Canning river system*: School of Resources, Environment and Society, Australian National University.

Goulder, L. H., & Parry, I. W. H. (2008). Instrument choice in environmental policy. *Review of Environmental Economic Policy, 2*(2), 152-174.

Greenhalgh, S., & Selman, M. (2006). Nutrient targeting a water quality solution. In K. Parris & T. Poincet (Eds.), *Water and Agriculture: Sustainability, Markets and Policies* OECD.

Griffin, R. C. (1991). The welfare analysis of transaction costs, externalities, and institutional choice. *American Journal of Agricultural Economics, 73*(3), 601-614.

Griffin, R. C., & Bromley, D. W. (1982). Agricultural runoff as a nonpoint externality: A theoritical developement. *American journal of Agricultural Economics, 64*(3), 547-552.

Gunningham, N., Grabosky, P., & Sinclair, D. (2004). Nononpoint pollution, voluntarism and policy failure: lessons for the Swan-canning. *Environmental and Planning Law Journal, 21*(2), 93-104.

Gunningham, N., & Sinclair, D. (2005). Policy instrument choice and diffuse source pollution. *Journal of Environmental Law, 17*(1), 51-81.

Hamilton, D. (2005). Land use impacts on nutrient export in the Central Volcanic Plateau, North Island. *NZ Journal of Forestry, 49*(4), 27-31.

Hanley, N., Shogren, J. F., & White, B. (1997). *Environmental Economics in Theory and Practice*. Basingstoke England: Macmillan.

Hansson, H., & Ohlmer, B. (2008). The effect of operational managerial practices on economic, technical and allocative efficiency at Swedish dairy farms, In Press, Corrected Proof. *Livestock Science, 118*(1-2), 34-43.

Hardaker, J. B., Humie, R. B. M., Anderson, J. R., & Lien, G. (2004). *Coping with Risk in Agriculture 2nd Edition*. UK: CABI.

Hart, R., & Latacz-Lohmann, U. (2005). Combating moral hazard in agri-environmental schemes: a multiple agent approach. *European Review of Agricultural Economics, 32*(1), 75-91.

Hazell, P. B. R., & Norton, R. D. (1986). *Mathematical Programming for Economic Analysis in Agriculture*. New York: Macmillan.

Hefland, G. E., & House, B. W. (1995). Regulating nonpoint source pollution under heterogeneous conditions. *American Journal of Agricultural Economics, 77*(4), 1024-1032.

Helin, J., Laukkanen, M., & Koikkalainen, K. (2006). Abatement costs for agricultural nitrogen and phosphorus loads: a case study of crop farming in South-Western Finland *Agricultural and Food Science, 15*(4), 351-374.

Hewitt, A. E. (1998). *New Zealand Soil Classification*. Lincoln, Canterbury: Landcare Research New Zealand.

Hill, M. J., Donald, G. E., Vickery, P. J., Moore, A. D., & Donnelly, J. R. (1999). Combining satellite data with a simulation model to describe spatial variability in pasture growth at a farm scale. *Australian Journal of Experimental Agriculture 39*(3), 285-300.

Hopkins, J., Schnitkey, G., & Tweeten, L. (1996). Impacts of nitrogen control policies on crop and livestock Farms at two Ohio farms site. *Review of Agricultural Economics, 18*(3), 311-324.

Horan, R. D., Shortle, J. S., Abler, D. G., & Ribaudo, M. (2001). *The Design and Comparative Economic Performance of Alternative Second- Best Point/Nonpoint Trading Markets, Staff Paper No 2001-16* Department of Agricultural, Economics, Michigan State University.

Horizons Regional Council. (2007). One Plan, Discharges to Land and Water. Retrieved Dec 16, 2007, from http://www.horizons.govt.nz/Images/Publications/ResourceManagement/Chapter13_dischargestolandandwater.pdf
Horst, D. V. (2005). *Spatial cost benefit thinking; towards a framework for spatial targeting of policy interventions.* Paper presented at the Multi Functionality of Land Scapes, Giessen, Germany.
Howard-Williams, C. C., & Pickmere, S. (1999). *Nutrient and vegetation changes in a retired pasture stream. Recent changes in the context of a long-term dataset. Science for Conservation 114.* Wellington.: Department of Conservation.
Hughes, W. (2007). *Regional Economic Bulletin*. Hamilton: Department of Economics, The University of Waikato.
Hynes, S., Morrissey, K., & O'Donoghue, C. (2006). *Building a static farm level spatial microsimulation model: statistically matching the Irish national farm survey to the Irish census of agriculture.* Paper presented at the European Regional Science Association. 2006.
Iho, A. (2005). Does scale matter? Cost-effectiveness of agricultural nutrient abatement when target level varies *Agricultural and Food Science, 14* (3), 277-292.
Iho, A. (2007). *Dynamically and spatially efficient phosphorus policies in crop production.* University of Helsinki, Helsinki.
Jaforullah, M., & Devlin, N. J. (1996). Technical efficiency in the New Zealand dairy industry: a frontier production function approach. *New Zealand Economic Papers, 30*, 1-17.
Jaforullah, M., & Premachandra, E. (2003). Sensitivity of technical efficiency estimates to estimation approaches: An investigation using New Zealand dairy industry data. *Economic Discussion Papers, University of Otago,* 22.
Jaforullah, M., & Whiteman, J. (1999). Scale Efficiency in the New Zealand dairy industry: a non-parametric approach. *The Australian Journal of Agricultural and Resource Economics, 43*(4), 523-541.
Johansson, R. C., Gowda, P. H., Mulla, D. J., & Dalzell, B. J. (2004). Metamodelling phosphorus best management practices for policy use: a frontier approach. *Agricultural Economics, 30*(1), 63-74.
Johnes, P. J. (1996). Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. *Journal of Hydrology, 183*(3-4), 323-349.
Johnson, S. L., Adams, R. M., & Perry, G. M. (1991). The on farm costs reducing groundwater pollution. *American Journal of Agricultural Economics, 73*(4), 1063-1072.
Judge, A., & Ledgard, S. (2004). *Nutrient Budgets for Waikato Dairy and Sheep/Beef Farms for 1997/98 and 2002/03, Technical Report, 2004/19*. Hamilton: Environment Waikato
Just, R. E., & Antle, J. M. (1990). Interactions between Agricultural and Environmental Policies: A Conceptual Framework. *American Economic Review, 80*(2), 197-202.
Kampas, A., & White, B. (2003). Selecting permit allocation rules for agricultural pollution control: a bargaining solution. *Ecological Economics, 47*(2-3), 135-147.

Kampas, A., & White, B. (2004). Administrative costs and instrument choice for stochastic nonpoint source pollutants. *Environmental and Resource Economics, 27*(2), 109-133.

Kerr, S., Lauder, G., & Fairman, D. (2007). *Towards a design for a nutrient trading programme to improve water quality in Lake Rotorua, Working Paper 07-03*. Wellington: Motu.

Kim, N., & Smith, J. (2006). *Review of Science Relating to Discharges from the Kinleith Paper Mill, Technical Report 2005/58*. Hamilton: Environment Waikato.

Kompas, T., & Che, T. N. (2006). Technology choice and efficiency on Australian dairy farms. *The Australian Journal of Agricultural and Resource Economics, 50*(1), 65-83.

Lankoski, J., Lichtenberg, E., & Ollikainen, M. (2008a). *Agri-Environmental Program Compliance in a Heterogeneous Landscape, WP 08-05*. College Park: Department of Agricultural and Resource Economics, The University of Maryland.

Lankoski, J., Lichtenberg, E., & Ollikainen, M. (2008b). Point/nonpoint effluent trading with spatial heterogeneity. *American Journal of Agricultural Economics, 90*(4), 1044-1058.

Lankoski, J., & Ollikainen, M. (2003). Agri-environmental externalities: a framework for designing targeted policies. *European Review of Agricultural Economics, 30*(1), 51-75.

Lant, C. L., Kraft, S. E., Beaulieu, J., Bennett, D., Loftus, T., & Nicklow, J. (2005). Using GIS-based ecological-economic modeling to evaluate policies affecting agricultural watersheds. *Ecological Economics, 55*(4), 467-484.

Ledgard, S. F. (2001). Nitrogen cycling in low input legume based agriculture, with emphasis on legume/grass pastures *Plant and Soil, 228*, 43-59.

Ledgard, S. F., De Klein, C. A. M., Crush, J. R., & Thorrold, B. S. (2000). Dairy farming, nitrogen losses and nitrate-sensitive areas. *Proceedings of the New Zealand Society of Animal Production 60 6*, 256-260.

Ledgard, S. F., & Menneer, J. C. (2005). Nitrate leaching in grazying systems and management strategies to reduce losses. In L. D. Currie & J. A. Hantly (Eds.), *Occasional report No. 18,* (pp. 79-92). Palmerston North: Fertilizer and Lime Research Centre, Massey University.

Ledgard, S. F., Penno, J. W., & Sporsen, M. S. (1999). Nitrogen input and losses from grass/clover pastures grazed by dairy cows, as affected by nitrogen fertiliser application. *Journal of Agricultural Science, 132*, 215-225.

Ledgard, S. F., & Power, I. (2006). *Nitrogen and phosphorus losses from 'average' Waikato farms to waterways as affected by best or potential management practices*. Hamilton: Environment Waikato.

Ledgard, S. F., & Thorrold, B. S. (2003). Nitrogen fertiliser use on Waikato Dairy Farms. Retrieved August 30, 2008, 2008, from http://www.dairynz.co.nz/file/fileid/6269

Legg, W. (2003). Presidential address agricultural subsidies: measurement and use in policy evaluation. *Journal of Agricultural Economics, 54*(2), 175-201.

Legg, W. (2006). Policy efforts to achive sustainable agriculture: an OECD perspective. In F. Brouwer & B. A. McCarl (Eds.), *Agriculture and Climate Beyond 2015, A New Perspective on Future Land Use Patterns*. The Netherlands: Springer.

Livestock Improvement Co-operation. (2008). *New Zealand Dairy Statistics 2007-08*. Hamilton.

Lokupitiya, R. S., Lokupitya, E., & Paustian, K. (2006). Comparison of missing value imputation methods for crop yield data. *Envirometrics, 17*, 339-349.
Lowrance, R., Altier, L. S., Newbold, J. D., Schnabel, R. R., Groffman, P. M., Denver, J. M., et al. (1997). Water Quality Functions of Riparian Forest Buffers in Chesapeake Bay Watersheds. *Environmental Management, 21*(5), 687-712.
Lymer, S., Brown, L., Payne, A., & Harding, A. (2006). *Development of 'HealthMod' - A model of the use and costs of medical services in Australia.* Paper presented at the 8th Nordic Seminar on Microsimulation Models, Oslo, Norway.
MacDonald, D. H., Connor, J., & Morrison, M. D. (2004). *Economic Instruments for Managing Water Quality in New Zealand*. Wellington, New Zealand.: Ministry for the Environment.
Macho-Stadler, I., & Perez-Castrillo, D. (2006). Optimal enforcement policy and firms' emissions and compliance with environmental taxes. *Journal of Environmental Economics and Management, 51*(1), 110-131.
Manley, B. (2005). Discount rates used for forest valuation – Results of 2005 Survey. *New Zealand Journal of Forestry, 50*(3), 7-11.
Mapp, H. P., Bernado, D. J., Sabbagh, G. J., Geleta, S., & Watkins, K. B. (1994). Economic and environmental impacts of limiting nitrogen use to protect water quality: A stochastic regional analysis. *American Journal of Agricultural Economics, 76*(4), 889-903.
Martin, T. L., Kaushik, N. K., Trevors, J. T., & Whitely, H. R. (1999). Review : denitrification in temperate climate riparian zones. *Water, Air and Soil Pollution, 111*(1-4), 171-186.
Martinez, Y., & Albiac, J. (2004). Agricultural pollution control under Spanish and European environmental policies. *Water Resources Research, 40*, W10501.
Martinez, Y., & Albiac, J. (2006). Nitrate pollution control under soil heterogeneity. *Land Use Policy, 23*(4), 521-532.
Mayer, D. G., Kinghorn, B. P., & Archer, A. A. (2005). Differential evolution - an easy and efficient evolutionary algorithm for model optimisation. *Agricultural Systems, 83*(3), 315-328.
McCall, D. G., & Bishop-Hurley, G. J. (2003). A pasture growth model for use in a whole-farm dairy production model. *Agricultural Systems, 76*(3), 1183-1205.
McCann, L., Colby, B., Easter, K. W., Kasterine, A., & Kuperan, K. V. (2005). Transaction cost measurement for evaluating environmental policies. *Ecological Economics, 52*(4), 527-542.
McCann, L., & Easter, K. W. (1999). Transaction costs of policies to reduce agricultural phosphorus pollution in the Minnestoa River. *Land Economics, 75*(3), 402-414.
McCann, L., & Easter, K. W. (2000). Estimates of public sector transaction costs in NRCS programmes. *Journal of Agricultural and Applied Economics, 32*(3), 555-563.
McCarl, B. A., Meeraus, A., Van der Eijk, P., Bussieck, M., Dirkse, S., & Steacy, P. (2007). *McCarl GAMS User Guide Version 22.5*. Washington DC: GAMS Development Corporation.
McCarl, B. A., & Spreen, T. H. (2004). *Applied Mathematical Programming using Algebraic Systems*: Department of Agricultural Economics, Texas A & M University.

Menneer, J. C., Ledgard, S. F., & Gillingham, A. G. (2004). *Land Use Impacts on Nitrogen and Phosphorus Loss and Management Options for Intervention* Tauranga: Environment Bay of Plenty.

Ministry for the Environment. (2001). *Valuing New Zealand's Clean Green Image*. Wellington.

Ministry for the Environment. (2003). *Dairying and Clean Streams Accord*. Wellington.

Ministry for the Environment. (2007). *The Dairying and Clean Streams Accord: Snapshot of Progress – 2005/2006*. Wellington.

Ministry of Agriculture and Forestry. (1997). *Impacts of Dairy Conversions in the Taupo District, Technical Document No. 97/9*. Wellington.

Ministry of Agriculture and Forestry. (2005a). *North Monitoring Report*. Wellington.

Ministry of Agriculture and Forestry. (2005b). *Situation and Outlook for New Zealand Agriculture and Forestry*. Wellington.

Ministry of Health. (2000). *Drinking-Water Standards for New Zealand* Wellington.

Monaghan, R. M., Hedley, M. J., Di, H. J., McDowell, R. W., Cameron, K. C., & Ledgard, S. F. (2007). Nutrient management in New Zealand pastures recent developments and future issues. *New Zealand Journal of Agricultural Research, 50*, 181-201.

Monaghan, R. M., Wilcock, R. J., Smith, L. C., Tikkisetty, B., Thorrold, B. S., & Costall, D. (2007). Linkages between land management activities and water quality in an intensively farmed catchment in southern New Zealand *Agriculture, Ecosystems & Environment, 118*(1-4), 211-222.

Monaghan, R. M., & Willcock, R. (2004). Landuse impacts on water quality within four intensively farmed catchments. In *Broadsheet* (pp. 52-57). Palmerston North: New Zealand Association of Resource Management.

Moxey, A., & White, B. (1994). Efficient compliance with agricultural nitrate pollution standards. *Journal of Agricultural Economics, 45*(1), 27-37.

Moxey, A., White, B., & O'Calaghan, J. R. (1995). The economic component of NEULP. *Journal of Environmental Planning and Management, 38*(1), 21-34.

Moxey, A., White, B., & Ozanne, A. (1999). Efficient contract design for agri-environmental policy. *Journal of Agricultural Economics, 50*(2), 187-202.

Munier, B., Birr-Pedersen, K., & Schou, J. S. (2004). Combined ecological and economic modelling in agricultural land use scenarios. *Ecological Modelling, 174*(1-2), 5-18.

Muniz, I. A., & Hurle, J. B. (2006). CAP MTR versus environmentally targeted agricultural policy in marginal arable areas: impact analysis combining simulation and survey data. *Agricultural Economics, 34*(3), 303-313.

Munksgaard, J., Christoffersen, L. B., Keiding, H., Pedersen, O. G., & Jensen, T. S. (2007). An environmental performance index for products reflecting damage costs. *Ecological Economics, 64*(1), 119-130.

Neal, M. (2004). *Re -ranking of individual firms using alternative performance measures: the case of New Zealand dairy farms.* Paper presented at the Asia-Pacific Productivity Conference, Brisbane.

Neal, M., Drynana, R., Fulkerson, W., Levy, G., Wastney, M. E., & Post, E. (2005). *Optimisation of a whole farm model* Paper presented at the AARES Conference, Australia.

Newell, R. G., & Stavins, R. N. (2003). Cost heterogeneity and the potential savings from market based policies. *Journal of Regulatory Economics, 23*(1), 43-59.
Newsome, P. F. J., Wilde, R. H., & Willoughby, R. H. (2000). *Land Resource Information System Spatial Data Layers*. Palmerston North: Landcare Research New Zealand Ltd.
Novonty, V. (1999). Diffuse pollution from agriculture- a worldwide outlook. *Water Science and Technology, 39*(3), 1-13.
Nuthall, P. L. (2001). Managerial ability — a review of its basis and potential improvement using psychological concepts *Agricultural Economics 24*(3), 247-262.
O'Connor, M. B. (1982). Nitrogen fertilizers for the production of out of season grass. In P. B. Lynch (Ed.), *Nitrogen Fertilizers in New Zealand Agriculture* (pp. 65-76). Wellington: New Zealand Institute of Agricultural Science.
OECD. (2007). *Instrument Mixes Addressing Nonpoint Sources of Water Pollution*. Paris: Environment Directorate for Food, Agriculture and Fisheries.
Ondersteijn, C. J. M., Beldman, A. C. G., Daatselaar, C. H. G., Giesen, G. W. J., & Huirne, R. B. M. (2002). The Dutch Mineral Accounting System and the European Nitrate Directive: implications for N and P management and farm performance. *Agriculture, Ecosystems & Environment, 92*(2-3), 283-296.
Ondersteijn, C. J. M., Lansink, O. A., Giesen, G. W. J., & Huirne, R. B. M. (2002). *Nutrient Efficiency and Nutrient Productivity Growth on Dutch Dairy Farms*. Wageningen: Department of Business Economics Wageningen University.
Opaluch, J. J., & Segerson, K. (1991). Aggregate analysis of site-specific pollution problems: the case of groundwater contamination from agriculture. *Northeastern Journal of Agricultural Resource Economics, 20*, 83-97.
Ozanne, A., Hogan, T., & Colman, D. (2001). Moral hazard, risk aversion and compliance monitoring in agri-environmental policy. *European Review of Agricultural Economics, 28*(3), 329-347.
Ozanne, A., & White, B. (2007). Equivalence of input quotas and input charges under asymmetric information in agri-environmental schemes. *Journal of Agricultural Economics, 58*(2), 260-268.
Palliser, C. C., Bright, K. P., Macdonald, K. A., Penno, J. W., & Wastney, M. E. (2001). *Adapting the MOLLY cow model to fit production data from New Zealand animals.* Paper presented at the New Zealand Society of Animal Production.
Palmquist, R. B. (1990). Pollution subsides and multiple local optima. *Land Economics, 66*(4), 394-401.
Pannell, D. J., Malcolm, L. R., & Kingwell, R. S. (2000). Are we risking too much? perspectives on risk in farm modeling. *Agricultural Economics, 23*(1), 69-78.
Parker, D., & Litchenberg, E. (2004). *Impacts of agricultural nutrient regulation in a heterogeneous region.* Paper presented at the American Agricultural Economics Association Annual Meeting, Colorado.
Parkyn, S. (2004). *Review of Riparian Buffer Zone Effectiveness, Technical Paper No: 2004/05*. Wellington: Ministry of Agriculture and Forestry.

Parliamentary Commissioner for the Environment. (2004). *Growing for Good Intensive Farming , Sustainability and New Zealand's Environment*. Wellington.

Parminter, I. (1999, October 1999). Farm Dairy Effluent Treatment in the Waikato Region. *RM update, Ministry of Agriculture and Forestry*.

Payraudeau, S., & Van der Werf, H. M. G. (2005). Environmental impact assessment for a farming region: a review of methods. *Agriculture, Ecosystems & Environment, 107*(1), 1-19.

Peng, W., & Bosch, D. J. (2001). Risk and site factors affecting potential nitrogen delivery in the Virginia coastal plain. *Journal of Agricultural and Applied Economics, 33*(1), 173-188.

Penn, D. A. (2007). Estimating missing values from the general social survey: An application of multiple imputation. *Social Science Quarterly, 88*(2), 573-584.

Peterson, J. M., & Boisvert, R. N. (2004). Incentive-compatible pollution control policies under asymmetric information on both risk preferences and technology. *American Journal of Agricultural Economics, 86*(2), 291-306.

Petrolia, D. R., & Gowda, P. H. (2006). Missing the boat: Midwest farm drainage and Gulf of Mexico hypoxia. *Review of Agricultural Economics, 28*(2), 240-253.

Philippe, G. V., & Hill, A. R. (2006). A landscape based approach to estimate riparian hydrological and nitrate removal functions. *Journal of the American Water Resources Association, 42*(4), 1099.

Picazo-Tadeo, A. J., & Reig-Martinez, E. (2007). Farmers' costs of environmental regulation: Reducing the consumption of nitrogen in citrus farming. *Economic Modelling, 24*(2), 312-328.

Pretty, J. N., Brett, C., Gee, D., Hine, R. E., Mason, C. F., Morison, J. I. L., et al. (2000). An assessment of the total external costs of UK agriculture. *Agricultural Systems, 65*(2), 113-136.

Price, K. V., Storn, R. M., & Lampinen, J. (2005). *Differential Evolution- A Practical Approach to Global Optimisation*. New York: Springer.

Qiu, Z., & Prato, T. (1999). Accounting for spatial characteristics of watersheds in evaluating water pollution abatement policies. *Journal of Agricultural and Applied Economics, 31*(1), 161-175.

R Development Core Team. (2007). R: A language and environment for statistical computing. Vienna, Austria: http://www.R-project.org.

Ramilan, T., & Scrimgeour, F. (2006). *Abatement Cost Heterogeneity and Its Impact on Tradable Nitrogen Discharge Permits.* Paper presented at the New Zealand Agricultural and Resource Economics Society, Nelson.

Ramilan, T., Scrimgeour, F. G., Levy, G., & Romera, A. J. (2007). *Modelling economic impact of agri-environmental policy on dairy farms –A catchment perspective.* Paper presented at the MODSIM 2007 International Congress on Modelling and Simulation, Christchurch.

Randall, A. (1987). *Resource Economics*. New York: John Wiley & Sons.

Reinhard, S., Lovell, C. A. K., & Thijssen, G. (1999). Econometric estimation of technical and environmental efficiency: an application to Dutch dairy farms. *American Journal of Agricultural Economics, 81*(1), 44-60.

Reinhard, S., Lovell, C. A. K., & Thijssen, G. (2000). Environmental efficiency with multiple environmentally detrimental variables; estimated with SFA and DEA. *European Journal of Operational Research, 121*(2), 287-303.

Reinhard, S., Lovell, C. A. K., & Thijssen, G. (2002). Analysis of Environmental Efficiency Variation. *American Journal of Agricultural Economics, 84*(4), 1054-1065.
Ribaudo, M., Gollenhon, N., Aillery, M., Kaplan, J., Agapoff, J., & Christensen, L. (2003). *Manure Management for Water Quality: Costs to Animal Feeding Operations of Applying Manure Nutrients to Land* Washington DC: Economic Research Service, U.S. Department of Agriculture.
Ribaudo, M. O., Richard, D. and Smith, M. E. (1999). *Economics of water quality protection from non point sources: theory and practice: Agricultural Economics Report No (AER782)*: Economic Research Service, United States Department of Agriculture.
Ribaudo, M. O., Heimlich, R., Claassen, R., & Peters, M. (2001). Least- cost management of nonpoint source pollution: source reduction verses interception strategies for controlling nitrogen loss in the Mississippi Basin. *Ecological Economics, 37*(2), 183-197.
Richards, K. R. (2000). Framing environmental policy instrument choice. *Duke Environmental Law and Policy Forum, 10*(2), 222- 285.
Ritchie, R. (2007). *On Farm Nutrient Management Practice - Research and Applicability to Upper Waikato, Technical Report 2007/42* Hamilton: Environment Waikato.
Roberts, A., & Morton, J. (2004). *Fertiliser Use on New Zealand Dairy Farms*. Auckland: New Zealand Fertiliser Manufacturers' Research Association.
Romstad, E. (2002). *Nonpoint source pollution contracts emission based regulations through models.* Paper presented at the World Congress for Environmental and Resource Economist, Monterey.
Rosenthal, R. E. (2006). *GAMS — A User's Guide*: GAMS Development Corporation, Washington, DC, USA.
Rousseau, S. (2001). Effluent trading to improve water quality: what do we know today? *Environment Working Paper Series No. 2001-26.*
Royston, P. (2005). Multiple imputation of missing values: update. *The Stata Journal, 8*(2), 1-14.
Rubin, D. B. (1987). *Multiple Imputation for Nonresponse in Surveys*. New York: Wiley.
Ruhl, J. B., Lant, C., Loftus, T., Kraft, S. E., Adams, J., & Duram, L. (2003). Proposal for a model state watershed management act. *Environmental Law, 33*, 929-947.
Rutherford, K. (2005). *The Effect on Waiakto River Water Quality of Changes to River Flow and Catchment Land Use, Including the December Addendum*. Hamilton: Environment Waikato Technical Report, 2005/03.
Rygnestad, H. (2000). *Intergrating Environmental Economics and Policy Analyses in a Geographical Information System, Working Paper No. 4/2000*: Danish Institute of Agricultural and Fisheries Economics
Sarkar, D. (2008). lattice: Lattice Graphics. R package version 0.17-4.
Šauer, P., Dvořák, A., Lisa, A., & Fiala, P. (2003). A procedure for negotiating pollution reduction under Information asymmetry. surface water quality case. *Environmental and Resource Economics, 24*(2), 103-119.
Schafer, J. L. (1999). Multiple imputation: a primer. *Statistical Methods in Medical Research, 8*, 3-15.
Schipper, L. A., Barkle, G. F., & Vojvodic-Vukovic, M. (2005). Upper limits of nitrate removal in a denitrification wall. *Journal of Environmental Quality, 34*, 1270-1276.

Schmitt, T. J., Dosskey, M. G., & Hoagland, K. D. (1999). Filter strip performance and processes for different vegetation, widths, and contaminants. *Journal of Environmental Quality, 28*(5), 1479.
Schnoebelen, D. J., Becher, K. D., Bobier, M. W., & Wilton, T. (1999). *Selected Nutrients and Pesticides in Streams of the Eastern Iowa Basins, 1970–95 U.S.Geological Survey Water-Resources Investigations.*
Schou, J. S., Skop, E., & Jensen, J. D. (2000). Integrated agri-environmental modelling: A cost-effectiveness analysis of two nitrogen tax instruments in the Vejle Fjord watershed, Denmark. *Journal of Environmental Management, 58*(3), 199-212.
Segerson, K. (1988). Uncertainty and incentives for nonpoint pollution control. *Journal of Environmental Economics and Management, 15*(1), 87-98.
Segerson, K., & Wu, J. (2006). Nonpoint pollution control: inducing first-best outcomes through the use of threats. *Journal of Environmental Economics and Management, 51*(2), 165-184.
Shortle, J. S. (1990). The allocative efficiency implications of water pollution abatement cost comparisons. *Water Resources Research, 26*(5), 793-797.
Shortle, J. S., Abler, D., G., & Horan, R. D. (1998). Research issues in nonpoint pollution control. *Environmental and Resource Economics, 11*(3), 571-585.
Shortle, J. S., & Dunn, J. W. (1986). The relative efficiency of agricultural source water pollution control policies. *American Journal of Agricultural Economics, 68* (3), 668-677.
Shortle, J. S., & Horan, R. D. (2001). The economics of nonpoint pollution control *Journal of Economic Surveys, 15*(3), 255-289.
Simar, L., & Wilson, P., W. (2000). Statistical inference in nonparametric frontier models: the state of the art. *Journal of Productivity Analysis, 13*(1), 49-78.
Simar, L., & Wilson, P. W. (2007). Estimation and inference in two-stage, semi-parametric models of production processes. *Journal of Econometrics, 136*(1), 31-64.
Simkin, S., Verwaart, T., & Vrolijk, H. (2005). Application of a Genetic Algorithm to Nearest Neighbour Classification. In *Innovations in Applied Artificial Intelligence* (pp. 544-546).
Skop, E., & Sorensen, P. B. (1998). GIS-based modelling of solute fluxes at the catchment scale: a case study of the agricultural contribution to the riverine nitrogen loading in the Vejle Fjord catchment, Denmark. *Ecological Modelling, 106*(2-3), 291-310.
Smith, B., & Horgan, G. (2006). *Area of Forest at Risk from Deforestration, Policy Paper*. Wellington: Ministry of Agriculture and Forestry.
Stace, C., & Fulton, V. (2003). Riparian protection in the Rotorua lake catchment. In N. E. Miller (Ed.), *Practical Management for Good Lake Water Quality*. Rotorua: Lake Water Quality Society and the Royal Society of New Zealand.
StataCorp. (2005). Stata Statistical Software: Release 9. College Station, TX: StataCorp LP.
StataCorp. (2007). Stata Statistical Software: Release 10: College Station, TX: StataCorp LP. .
Stephenson, K., Norris, P., & Shabman, L. (1998). Watershed -based effluent trading: the nonpoint source challenge. *Contemporary Economic Policy, 16*(4), 412-421.
Sterner, T. (2003). *Policy Instruments for Environmental and Natural Resource Management*. Washington, DC: Resources for The Future.

Stiglitz, J. E. (2000). *Economics of the public sector* (3rd ed.). New York: W. W. Norton.
Storn, R., & Price, K. (1997). Differential evolution - a simple and efficient heuristic for global optimization over continuous spaces. *Journal of Global Optimization, 11*(4), 341-359.
Stranlund, J. (2007). The regulatory choice of noncompliance in emissions trading programs. *Environmental and Resource Economics, 38*(1), 99-117.
Stringleman, H. (2007, 23 March 2007). Farmers stand against a regulatory wave. *The National Business Review*.
Sudheer, K. P., Chaubey, I., & Garg , V. (2006). Lake water quality assessment From Landsat thematic mapper data using neural network: an approach to optimal band combination selection. *Journal of the American Water Resources Association, 42*(6), 1683-1695.
Sumelius, J., Grgic, Z., Mesic, M., & Franic, R. (2005). Marginal abatement costs for reducing leaching of nitrates in Croatian farming systems. *Agricultural and Food Science, 14*(3), 293-309.
Suter, J. F., Vossler, C. A., Poe, G. L., & Segerson, K. (2008). Experiments on damage-based ambient taxes for nonpoint source polluters. *American Journal of Agricultural Economics, 90*(1), 86-102.
Swinton, S. M., & Clark, D. S. (1994). Farm level evaluation of alternative policy approaches to reduce nitrate leaching from Midwest Agriculture. *Agricultural and resource economic review, 23*, 66-74.
Tagg, F. (2007, September). Dexcel: Helping farmers keep it clean. *New Zealand Dairy Exporter,* 71-71.
Tanner, C. C., Nguyen, M. L., & Sukias, J. P. S. (2005). Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. *Agriculture, Ecosystems & Environment, 105*(1-2), 145-162.
Taylor, M., L., Adams, R, M. and Miller, S , F. (1992). Farm level response to agricultural effluent control strategies: The case of the Willamette Valley. *Journal of Agricultural and Resource Economics, 17*(1), 173-185.
Thomas, S., Ledgard, S., & Francis, G. (2005). Improving estimates of nitrate leaching for quantifying New Zealand's indirect nitrous oxide emissions. *Nutrient Cycling in Agroecosystems, 73*(2), 213-226.
Tietenberg, T. H. (2006a). *Environmental and Natural Resource Economics* (7th ed.). Boston: Addison Wesley.
Tietenberg, T. H. (2006b). *Emissions Trading: Principles and Practice* (2nd ed.). Washington, DC: Resources for the Future.
Tillman, R. (2008). Fertiliser Nitrogen or Clover Nitrogen? Retrieved August 31, 2008, from http://www.dairynz.co.nz/file/fileid/6150
Tyteca, D. (1996). On the measurement of the environmental performance of firms - A literature review and a productive efficiency perspective. *Journal of Envrionmental Management, 46*(3), 282-308.
Tyteca, D. (1997). Linear programming models for the measurement of environmental performance of firms-concepts and empirical results. *Journal of Productivity Analysis, 8*(2), 183-198.
Van Buuren, S., Boshuizen, H. C., & Knook, D. L. (1999). Multiple imputation of missing blood pressure covariates in survival analysis. *Statistics in Medicine, 18*, 681-694.

VanDyke, L. S., Bosch, D. J., & Pease, J. W. (1999). Impacts of within -farm soil variability on nitrogen pollution control costs. *Journal of Agricultural and Applied Economics, 31*(1), 149-159.
Vant, B. (1999). *Sources of Nitrogen and Phosphorus in Several Major Rivers in the Waikato Region, Technical Report 1999/10*. Hamilton: Environment Waikato.
Vant, B., & Smith, P. A. (2004). *Trends in River Water Quality in the Waikato Region, 1987-2002. Technical Report 2004/02*. Hamilton: Environment Waikato.
Vermunt, K., & Magidson, J. (2005). *Latent GOLD 4.0 User's Guide*. Belmont, Massachusetts: Statistical Innovations Inc.
Vrolijk, H. C. J. (2004). *STARS: statistics for regional studies.* Paper presented at the New roads for farm accounting and FADN, The Hague.
Water Programme of Action. (2007). *Impact of Management Changes on Farm Profitability and Environmental Outcomes*: The Agribusiness Group.
Weersink, A., Jeffrey, S., & Pannell, D. (2002). Farm level modeling for bigger issues. *Review of Agricultural Economics, 24*(1), 123-140.
Weersink, A., Livernois, J., Shogren, J. F., & Shortle, J. S. (1998). Economic instruments and environmental policy in agriculture. *Canadian Public Policy, 24*(3), 309-327.
Westra, J. V., Easter, K. W., & Olson, K. D. (2002). Targeting nonpoint source pollution control: phosphorus in the Minnesota river basin. *Journal of the American Water Resources Association, 38*(2), 493-505.
Wheeler, D. M., Ledgard, S. F., DeKlein, C. A. M., Monaghan, R. M., & Carey, P. L. (2003). *Overseer*® nutrient budgets – moving towards on-farm resource accounting, Proceedings of the New Zealand Grassland Association *65*, 191-194.
Wilcock, R. J., Monaghan, R. M., Quinn, J. M., Cambell, A. M., Thorrold, B. S., Duncan, M. J., et al. (2006). Land-use impacts and water quality targets in the intensive dairying catchment of the Toenepi stream, New Zealand. *New Zealand Journal of Marine and Freshwater Research, 40*, 123-140.
Wilcock, R. J., Nagels, J. W., Rodda, H. J. E., O'Connor, M. B., Thorrold, B. S., & Barnet, J. W. (1999). Water quality of a lowland stream in a New Zealand dairy farming catchment. *New Zealand Journal of Marine and Fresh Water Research, 33*, 683-696.
Williamson, O. E. (1985). *The Economic Institutions of Capitalism*. New York, NY: Free Press.
Williamson, R. B., Smith, C. M., & Cooper, A. B. (1996). Watershed riparian management and its benefits to a Eutrophic lake. *Journal of Water Resources Planning and Management, 122*(1), 24-32.
Wilson, P., W. (2008). FEAR: A software package for frontier efficiency analysis with R. *Socio-Economic Planning Sciences, DOI:10.1016/j.seps.2007.02.001*.
Wolf, J., Rotter, R., & Oenema, O. (2005). Nutrient emission models in environmental policy evaluation at different scales--experience from the Netherlands. *Agriculture, Ecosystems & Environment, 105*(1-2), 291-306.
Woodford, K. (2006). The intensification of pastoral agriculture: some trends and implications. *Primary Industry Management, 9*(2), 3-4.
Worrall, F., & Burt, T. P. (1999). A univariate model of river water nitrate time series. *Journal of Hydrology, 214*(1-4), 74-90.

Wossink, A., & Denaux, Z. S. (2006). Environmental and cost efficiency of pesticide use in transgenic and conventional cotton production. *Agricultural Systems, 90*(1-3), 312-328.
Wossink, A., Lansink, A. O., & Struik, P. C. (2001). Non-separability and heterogeneity in integrated agronomic-economic analysis of nonpoint-source pollution. *Ecological Economics, 38*(3), 345-357.
Wu, J. (2000). Input substitution and pollution control under uncertainty and firm heterogeneity. *Journal of Public Economic Theory, 2*(2), 273-288.
Wu, J., & Babcock, B. A. (1999). Metamodeling potential nitrate water pollution in the Cenral United States. *Journal of Environmental Quality, 28*, 1916-1928.
Wu, J., & Babcock, B. A. (2001). Spatial heterogeneity and the choice of instruments to control nonpoint pollution. *Environmental and Resource Economics, 18*(2), 173-192.
Wu, J., Teague, M. L., Mapp, H. P., & Bernado, D. J. (1995). An Empirical analysis of the relative efficiency of policy instruments to reduce nitrate water pollution in the U.S. Southern High Plains. *Canadian Journal of Agricultural Economics, 43*(3), 403-420.
Xabadia, A., Goetz, R. U., & Zilberman, D. (2006). Control of accumulating stock pollution by heterogeneous producers. *Journal of Economic Dynamics and Control, 30*(7), 1105-1130.
Xepapadeas, A. P. (1992). Environmental policy, adjustment costs, and behavior of the firm. *Journal of Environmental Economics and Management, 23*(3), 258-275.
Yang, W., & Weersink, A. (2004). Cost-effective Targeting of Riparian Buffers. *Canadian Journal of Agricultural Economics, 52*(1), 17-34.
Yiridoe, E. K., & Weersink, A. (1998). Marginal abatement costs of reducing groundwater-N pollution with intensive and extensive farm management choices *Agricultural and Resource Economics Review, 27* (2), 169 -185
Yoshizoe, Y., & Araki, M. (1999). *Statistical matching of household survey files, ITME Discussion Paper, No. 10.* Japan: The University of Tokyo.
Zhang, B., & Tillman, R. (2007). A decision tree approach to modelling nitrogen fertiliser use efficiency in New Zealand pastures. *Plant and Soil, 301*(1), 267-278.
Zilberman, D., Khanna, M., & Lipper, L. (1997). Economics of new technologies for sustainable agriculture. *The Australian Journal of Agricultural and Resource Economics, 41*(1), 63-80.