Valuing biodiversity: decision support for biosecurity response

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Abstract

This thesis develops a Decision Support System (DSS) to improve biosecurity response decisions affecting indigenous biodiversity. The key elements of the DSS are a synthesis of three components of non-market valuation: choice modelling, a systematic database of values and benefit transfer, with risk simulation to account for uncertainty. The innovative framework is demonstrated in a manual developed for Biosecurity New Zealand analysts for use during the early days of an incursion when time is severely constrained and uncertainty abounds.

Theoretical approaches to decision making for environmental resource allocation decisions are reviewed with particular reference to decision support tools including non-market valuation, stated preference techniques, database development and benefit transfer. More complex and time consuming tools such as deliberative support and mediated modelling are also discussed. A framework is developed to quantify biodiversity values for use in Cost Benefit Analysis (CBA). The framework incorporates advanced choice modelling techniques demonstrated through four case studies, a systematic database of biodiversity values and transfer of these values using univariate benefit transfer with theoretical adjustment. The uncertainty in the values captured in the panel version of the Random Parameters Logit (RPL) model is integrated into CBA using risk simulation which utilises the means, standard deviations and correlation coefficients of risky variables to quantify the probability of achieving a positive Net Present Value of different response options.

The DSS developed in this thesis has wider application in routine management of pests and diseases, and in other resource allocation decisions by public agencies which impact on indigenous biodiversity.

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List of Acronyms

ABM	Attribute based method
AERU	Agricultural Economics Research Unit
AHP	Analytical hierarchy process
ALARM	Integrated project addressing biodiversity loss risks
B3	Better border Biosecurity
B/C	Benefit Cost Ratio
BNZ	Biosecurity New Zealand
BVD	Biodiversity Valuation Database
BT	Benefit Transfer
CBA	Cost Benefit Analysis
CE	Choice Experiment
CEA	Cost Effectiveness Analysis
CM	Choice Modelling
CPI	Consumer Price Index
CS	Compensating Surplus
CSR	Corporate social responsibility
CUA	Cost Utility Analysis
CV	Contingent Valuation
DFD	Data flow diagram
DoC	Department of Conservation
DSS	Decision Support System
DST	Deliberation support tools
ECMS	Enterprise Content Management System
ECS	Expected consumer surplus
EDP	Electronic Data Processing
EIA	Economic Impact Assessment
EVRI	Environmental Valuation Reference Inventory
FMD	Foot and Mouth Disease
GUI	Graphical user interface
FRST	Foundation for Research Science and Technology
ICT	Information Communications and Technology
IRS	Incursion Response System
LC	Latent Class
LTCCP	Long term community consultation plan
MAF	Ministry of Agriculture and Forestry

MAFBNZ	Ministry of Agriculture and Forestry Biosecurity New Zealand
MCA	Multi criteria analysis
MCE	Multi criteria evaluation
MIS	Management information system
ML	Mixed Logit
MM-DST	Multi-media deliberation support tools
MNL	Multinomial Logit
MoH	Ministry of Health
MRS	Marginal Rate of Substitution
MSE	Management Strategy Evaluation
NOAA	National Oceanic and Atmospheric Administration
NPV	Net Present Value
OLAP	On-line analytical processing
OP	Option Price
OV	Option Value
ppp	Purchasing Power Parity
PV	Present value
QuRA™	Quantified Risk Analysis
RAD	Rapid application development
RPA	Resource Planning Act
RPL	Random Parameters Logit
RUM	Random Utility Model
SD	Standard Deviation
SDC	Socio-Demographic Characteristic
SDLC	Systems development life cycle
TEV	Total Economic Value
TIDDD	Tools for informing discussions, debates and deliberation
VARB	Variance-covariance matrix of attributes
WTA	willingness to accept
WTP	willingness to pay

Chapter 1 : Introduction

1.1 Statement of the problem

Biosecurity New Zealand (BNZ) a division of the Ministry of Agriculture and Forestry (MAF) is charged with protecting New Zealand's indigenous biodiversity from attacks from exotic pests and diseases (MAF, 2003). A key issue in successfully discharging this mandate is how to determine the appropriate response to incursions. Under a funding constraint, this task is made difficult through a lack of money values on indigenous ecosystems. BNZ must advise government on making choices between spending money on protecting export industries, human health and indigenous biodiversity. The overall challenge is to provide a system that allocates scarce funds to maximise the well-being of New Zealanders.

Traditional Cost Benefit Analysis (CBA) can provide reasonable estimates where there are markets for traded goods in order to satisfy The Treasury that the returns from action taken meet the rate of return requirements of the government (The Treasury, 2005). However, valuation is much more difficult for non-traded goods and services, such as indigenous biodiversity, because of the absence of monetary values determined in a market. As a consequence, making a case for indigenous biodiversity protection that will meet The Treasury requirements is a major issue for Biosecurity New Zealand and other agencies such as the Department of Conservation (DoC).

Initial work to develop a model to guide economic impact assessment and CBA was undertaken by Harris, Clough and Shaw (2003). Their aim was to develop and apply, for DoC, a standard model for the assessment of the potential costs and benefits associated with any given biosecurity operation and assist with priority setting between biosecurity operations. They reviewed five potential methods: cost benefit analysis (CBA); cost utility analysis (CUA); cost effectiveness analysis (CEA); multi criteria analysis (MCA), and management strategy evaluation (MSE).

Their recommendation was for DoC to adopt the MAF CBA guidelines and project format (Forbes, 1984), possibly with some refinements. This required the development of monetary valuations of changes in natural heritage and indigenous biodiversity by using the tools of Contingent Valuation, Choice Modelling and implied preference using current expenditure. They also recommended that units of natural heritage and indigenous biodiversity should be derived from Site Value techniques (Stephens, Brown, and Thorney, 2002) in the DoC MCA project. Their work did not proceed beyond the development of an outline framework. Seeing that an outline framework existed, an approach was made to FRST for project funding to further develop and implement a system along the lines envisaged by Harris and Clough, but with BNZ as the primary client. The result is this thesis.

1.2 Research objectives

The research objective is to develop a Decision Support System (DSS) for MAFBNZ that integrates monetary values for indigenous biodiversity into a CBA to assist decision makers allocate scarce resources in response to exotic pest and disease incursions.

The DSS must be robust, yet able to be operated quickly by MAFBNZ personnel. It must be grounded in theory, take weeks not months to generate outputs, and be able to be applied by in-house economists who are severely resource constrained.

1.3 International context

As a small open economy with global connectedness, New Zealand faces increasing risks from incursions of unwanted pests and diseases. While many organisms that arrive in New Zealand do not pose a problem, a few may cause significant harm or even catastrophic harm in certain habitats. Examples include rabbits in the dry high country and possums in native bush. Decision makers need tools that will help them allocate scarce resources to deciding how much to spend on an individual incursion to limit or stop damage. Cost benefit analysis is one such tool that, while not universally accepted, is the best available and has stood the test of time from when it was first introduced in the United States in 1936 (Pearce, 1971). Over the years the tool has been developed and applied to increasingly complex issues.

In New Zealand, cost benefit analysis has been applied since the late 1950s to the analysis of capital allocation problems (Forbes, 1984). Initially these related to rationing scarce capital available to implement major public works programmes aimed at generating agricultural exports. Non-market valuation (NMV) had limited application, for example, estimating aesthetic values associated with national parks and hydro development (Kerr and Sharp, 1985). There was interest in cultural interpretation of NMV studies comparing western values with that of the tangata whenua in contingent valuation (CV) studies (Fahy and Kerr, 1991). Option pricing also received attention with a focus on the uncertainty (Sharp and Cullen, 1991). Economic approaches to environmental management were seen as useful complements to regulation (Meister and Sharp, 1993). Applying NMV to decisions involving biosecurity is a challenge of the new millennium for New Zealand analysts, which requires measuring benefits and costs in the absence of market determined values, usually under considerable time pressure.

Other Pacific rim countries including Australia face similar biosecurity problems to New Zealand and so there is an opportunity to apply the DSS more widely. This general usefulness applies not only to the DSS but also to the primary data inputs that go into the model.

1.4 Multi disciplinary and multi faceted decision making

The basic research question is whether the application of economic tools can improve decision making for protecting New Zealand's indigenous biodiversity. Economic theory gives guidance on the criteria and analysis that should be undertaken, but there are significant challenges with the application of the tools in real world situations where constraints of money and time, force decision makers into processes that fall short of that which is theoretically possible.

There are no signals from markets to help estimate values for indigenous biodiversity. Also, biodiversity itself provides a complex array of goods and services to society that requires a multi-disciplinary approach to quantification. At a minimum, to implement a DSS collaboration is required with ecologists, biosecurity specialists and economists.

Biosecurity decisions are multi-dimensional and require consideration of economic, environmental, human health, social and cultural benefits and costs. While the focus of this research is on environmental benefits and costs, the DSS will also apply to the other benefits and costs.

Biosecurity decision making is characterised by risk and uncertainty. Transferring plants and animals from one environment to another particularly between countries is often subject to unknown or unforeseen consequences that can have major negative implications in the new environment. It is extremely difficult to judge *a priori* how an organism will behave in a new environment. For this reason, biosecurity agencies have to be vigilant at the border and act decisively in surveillance, response and pest management. Traditionally, informing decision makers of these uncertainties and the risks they create has been facilitated by sensitivity analysis, scenario analysis and 'what if?' analysis. However, these techniques say nothing about the probability of outcomes. Risk simulation, which combines the uncertainties associated with the various key drivers of the project and bundles these into a single measure of the risk associated with the response is a significant advance.

1.5 Outline of approach

The research fills the needs of MAFBNZ, DoC and the Treasury by developing a Decision Support System to assess non-market situations in biosecurity utilising a series of economic tools within a cost benefit analysis framework.

While there have been considerable gains made in extending cost benefit analysis to non-market situations internationally, in New Zealand there is still a major gap in knowledge and information to enable its consistent application. This research focuses specifically on incorporating choice modelling, a systematic database of values, benefit transfer and risk simulation into a decision support system that can be implemented by Biosecurity New Zealand staff when time is severely constrained. The techniques developed will have application to other natural resource decision problems.

The analytical tools form the basis of a DSS that provides improved information for incorporation into cost benefit analysis of optimal response options where indigenous biodiversity is at risk.

The contribution this research makes to science is the adaption and integration of three separate tools (choice modelling, benefit transfer and risk simulation) into a robust biosecurity DSS that can be implemented by MAFBNZ analysts in time constrained situations. This is the first application of choice modelling to biosecurity response, the first systematic approach employing benefit transfer to value the impacts of biosecurity response and the first use of risk simulation to incorporate uncertainty into biosecurity decision making. In combination, these three advances provide a significant step forward in the information provided for biosecurity decisions.

1.6 Outline of Thesis

Chapter 2 begins with an outline of MAFBNZ's current decision support system for biosecurity response to exotic pests and diseases. It then reviews allied research that could inform this research and finally identifies gaps and deficiencies that this research aims to fill.

Chapter 3 starts with a review of conceptual and philosophical issues on ecosystem health, biodiversity's place in natural capital and the relevance of social capital to biodiversity valuation. It then explores the tension between ecological and neoclassical economics. This leads to a review of decision support tools including non-market valuation, stated preference techniques, database development and benefit transfer. Decision frameworks are then discussed including decision support tools and the more complex and time consuming approaches of deliberative support and mediated modelling. An economic framework is outlined for quantifying biodiversity values in Cost Benefit Analysis.

Chapter 4 demonstrates the methodology used for the four case studies that form the foundations of a biodiversity valuation database (BVD). The case of Lake Rotoroa, a freshwater lake in the centre of the city of Hamilton, is presented as an example. Advanced Bayesian experimental design, innovative hybrid community surveying and choice modelling techniques that allow for heterogeneity between respondents using the panel version of the Random Parameters Logit (RPL) model are employed to value key biodiversity attributes.

Chapter 5 outlines the approach used to develop a systematic database of biodiversity values (the BVD). First, the major features of the four case studies are outlined and the key insights summarised. Second, the rationale for developing a specific database for biosecurity is outlined with reference made to existing international and New Zealand based non-market valuation databases. Third, the common methodology adopted for the re-evaluated four case studies is outlined and the results presented. Fourth and lastly, the key issues that need to be taken into account when using benefit transfer to utilise the biodiversity values in biosecurity decisions are outlined.

Chapter 6 describes the benefit transfer process whereby biodiversity values are incorporated into a DSS using univariate direct transfer with theoretical adjustment. The uncertainties in the values captured in the RPL model are integrated into CBA using risk simulation which utilises the means, standard deviations and correlation coefficients of risky variables to quantify the probability of achieving a positive Net Present Value of different response options.

Chapter 7, the final chapter, provides an evaluation, critique and conclusions on the contribution of this thesis. The innovative Decision Support System developed for biosecurity is shown to have wider application in routine management of pests and diseases and in other resource allocation decisions by public agencies affecting indigenous biodiversity.

Chapter 2 : MAFBNZ: the policy and management challenge

2.1 The policy and management challenge

Biosecurity New Zealand, a division of the Ministry of Agriculture and Forestry (MAFBNZ), has the responsibility to lead New Zealand's biosecurity system as identified in the Biosecurity Strategy for New Zealand (Biosecurity Council, 2003). A key component of the strategy is the response to incursions of exotic pests and diseases where there are significant public benefits (MAFBNZ, 2009). Such response also includes the long-term mitigation of established pest and disease organisms.

The purpose of this chapter is to outline the procedures currently in force for responding to an incursion. This forms the base on which to build an extra component to the current decision support system that explicitly quantifies the non-market values of changes to indigenous biodiversity.

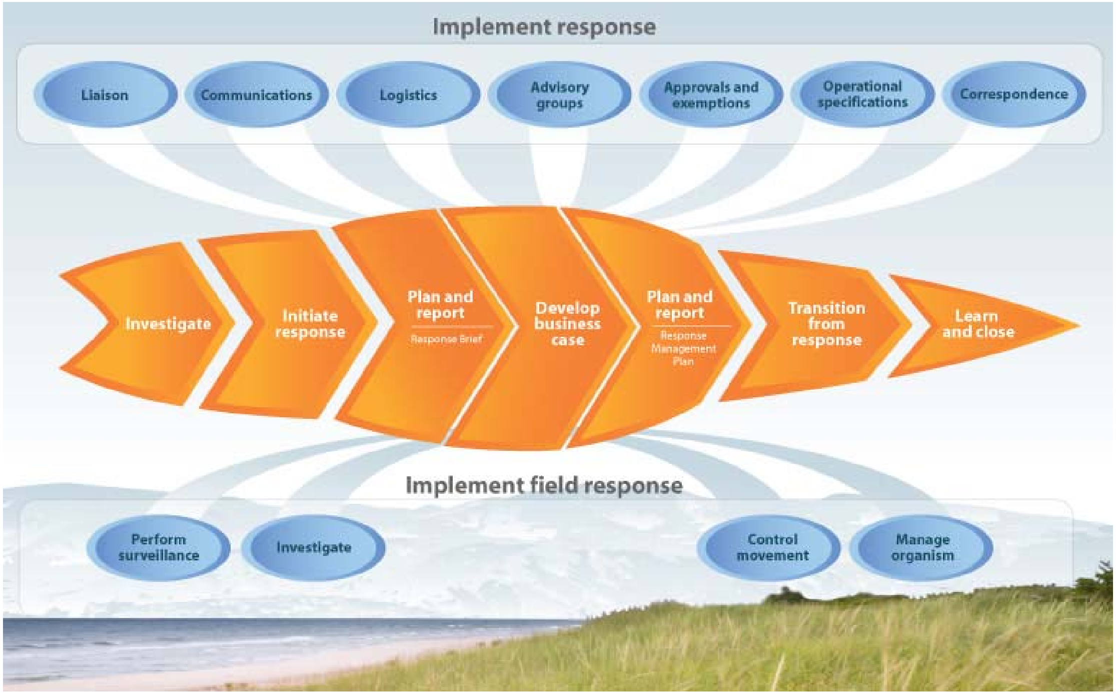
In July 2008, Cabinet approved a new biosecurity response policy that set out the processes and analytical tools to ensure an appropriate response would be undertaken (MAFBNZ, 2008). This replaced the Biosecurity Council policy statement on responding to an exotic organism incursion announced in September 2001 (MAFBNZ, 2001).

The aim is to achieve the best overall outcome for New Zealand by minimising the costs of both the incursion by an organism and the control method (e.g. a spraying programme). The process needs to be completed in a timely way incorporating the best scientific and other information, and ensuring that uncertainty should not inappropriately delay action. Value criteria for assessing benefits and costs cover the full range of effects across all sectors in particular, human health, economic, socio-cultural, environmental and Maori.

Set out below are the essential features of the general response process. The names within brackets (<<*name*>>) refer to documents in the MAFBNZ system (MAFBNZ, 2007). The process maps and procedures that underlie the response policy are contained in a document titled "response model" (<<*response model.doc*>>), which is referred to as the rocket ship (see Figure 2.1). For each task that needs to be completed (e.g. develop response options working paper, assess impacts of options etc), there are processes (process maps), policies, guidelines and templates to assist people in developing these tasks.

There are two critical stages where biodiversity values contribute to improved decision making: *Investigate* and *Develop Business Case*.

Figure 2-1 The rocket ship: A model of the response



Source (MAFBNZ, 2007)

2.1.1 Investigate

When a response is being investigated (first step in Figure 2.2), there is a template that must be filled out called the rapid assessment report (<<*rapid-assessment-report-template.dot*>>). This report helps determine whether or not to initiate a response. As part of the risk assessment section of the rapid assessment report, information is required about the potential for adverse impacts to human health, economic, environmental and socio-cultural values from the pest. It is here that high level values of environmental impacts could be utilized from benefit transfer studies (discussed in Chapter 3.5). Currently only qualitative information is provided.

Figure 2-2 Investigate

Source: (MAFBNZ, 2007)

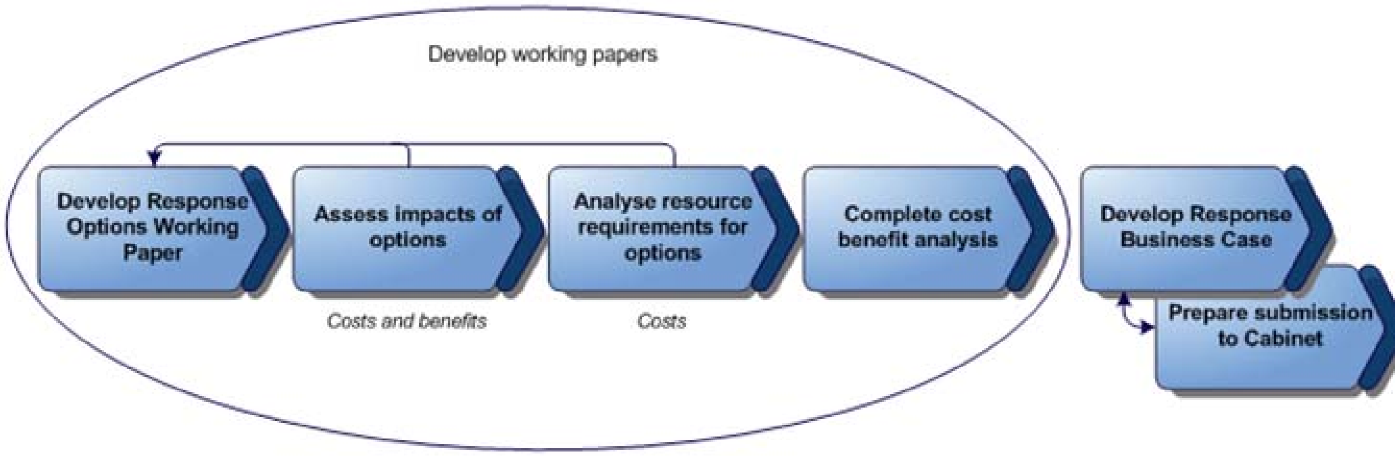
2.1.2 Develop Business Case

This involves the next two steps: initiate response, and plan and report (Figure 2.1). At this stage a response brief (<<*response-brief-template.dot*>>) is completed to clearly state the biosecurity risk, articulate the outcomes the response is trying to achieve and define the approach and resources required to obtain approval for the Business Case. This involves implementing interim measures to ensure the response outcomes are achievable and obtaining Response Strategic Leadership's approval for the resources and funding required to proceed with this work.

Next, the business case is developed (see Figure 2.3) using the following templates: (<<response-business-case-template.dot>> <<response budget working paper - summary.xlt>> <<cost-benefit-analysis-template.dot>> <<response-options-working-paper-

template.dot>>). Guidelines on the use of these templates are provided in the "response model" document above.

Figure 2-3 Development of a Business Case



Source: MAFBNZ 2007

2.1.3 Cost Benefit Analysis

The CBA guidelines in the response model for unwanted organism or pest response options are dated 2002. They comprehensively detail the steps required to undertake a robust analysis.

However, it is often the case that time constraints mean that a primary economic impact assessment (EIA) or CBA is not undertaken and a past study or studies are reviewed and benefits and costs modified or adapted to the new situation. Where quantitative information is not able to be estimated, an indicative breakeven analysis may be employed. This estimates the quantum of benefits required to exceed the costs. A qualitative assessment is then made as to the likelihood of this occurring.

The valuation of effects is a key component of the CBA and the guidelines cover stated preference techniques, which are the standard way environmental benefits can be quantified. The guidelines note that these techniques are expensive and time consuming with estimation of Willingness to Pay (WTP) complex and often contentious. It is recommended that the assistance of researchers with expertise in the area is sought when it is necessary to undertake such studies.

Where benefits and costs can be quantified, a discounted cashflow analysis is undertaken with a standard project life of 20 years (The Treasury, 2005). A standard Treasury discount rate of 10% real was applied for many years and only recently changed to 8% (The Treasury, 2008). The discount rate is applied to estimate the Net Present Value (NPV) and Benefit Cost Ratio (B/C) of the control option. The guidelines note that overseas studies often use a discount rate of six to eight per cent, and it is sometimes suggested that a much lower rate (or even zero) may be applied to non-commercial or social effects. The guidelines state that the preferred approach is to explicitly model increasing values of annual cost or benefit rather than apply a lower discount rate. For example, the annual benefit derived from environmental assets may increase in real terms over time as society becomes wealthier and stocks of environmental assets decline.

In ranking response options the guidelines state the superior criteria are NPV > 0 and or B/C > 1.0. However, because of budget constraints the hurdle rate of return in practice appears to be B/C > 3.0, implying that quantified benefits must be at least three times that of costs in order for the option to be recommended for implementation.

Uncertainty is handled through "what-if", sensitivity and scenario analysis, while the guidelines advise that risk analysis be undertaken by research providers with specialist expertise. Quantitative risk analysis using risk simulation is a logical extension of the analysis to better inform decision makers of the likelihood of a positive NPV or that the B/C ratio hurdle is likely to be exceeded (Bell, 2000).

Timeframes to undertake an EIA or CBA vary depending on the urgency to make a decision and could be as short as one week for an in-house analysis. A minimum of three months is normal if the analysis is contracted out even if the analysis is based on a previously prepared CBA. Such short timeframes rule out techniques that involve primary surveys and detailed analysis.

Once a draft EIA or CBA has been prepared there may be a number of revisions as more information becomes available on the cost of response options or impacts.

While there is often extreme pressure to complete an EIA or CBA the process from Options Analysis to Business Case can be extended in order to undertake a more thorough analysis, where the need is identified early in the response process.

Documents are handled in the Enterprise Content Management System (ECMS) and all staff are provided with training on its use. Stand alone systems that need supporting use the Incursion Response System (IRS) while standard document and spreadsheet files are managed within folders via the ECMS.

2.1.4 Pest Management

Regional Councils have the responsibility to undertake a CBA when a pest or disease is to be accorded status under a Regional Pest Management Strategy. These studies are mostly undertaken by consultants as councils are generally not resourced to do this work in-house. There appears to be little coordination between councils when undertaking CBA. As a result the quality and comprehensiveness of such studies is variable.

Occasionally MAFBNZ is asked to comment on a CBA undertaken for regional pest management purposes. In theory, the MAF CBA guidelines for response should apply to pest management, but there is no requirement for this to occur.

2.2 Theoretical framework

The Government has adopted the "4Rs" approach to risk management in its emergency response system – reduction, readiness, response, and recovery. Each of the 4Rs requires action at individual, business, community and government levels. Reduction relates to reduction of risk at pre-border and border; readiness relates to the preparedness and capacity to manage a response; response relates to the investigation, identification and management of the organism; and recovery relates to efforts to ensure the community recovers from the effects of a biosecurity emergency.

In the BNZ context, risk is defined as "the chance of something happening that will have an impact on objectives" and is measured in terms of consequence - "outcome or impact of an event" and likelihood - "probability or frequency".

Risk management is defined by Australian Standards and Standards New Zealand as the "culture, processes and structures that are directed towards realising the potential opportunities whilst managing adverse effects" (AS/NZS, 2006, pp. 2-4).

Biosecurity is defined as "the exclusion, eradication or effective management of risks posed by pests and diseases to the economy, environment and human health" (Biosecurity Council, 2003, p. 5). It is "a means to achieve outcomes such as the protection of primary production systems, human health, indigenous flora, fauna and biodiversity from harmful organisms and to maintain or improve ecosystem health" (Parliamentary Commissioner for the Environment, 2000, p. 9).

Risk management "involves managing to achieve an appropriate balance between realising opportunities for gains and minimising losses" Australian Standards and Standards New Zealand, 2004 (p.iv). "In general, the costs, including opportunities foregone, of managing risks should be commensurate with the benefits obtained."

The new response policy builds on the Integrated Risk Management Framework which drew in turn on two existing risk management frameworks, Australian Standards and Standards New Zealand, and the New Zealand food safety risk management framework, Ministry of Health and Ministry of Agriculture and Forestry Food Harmonisation Project (MOH and MAF, 2000).

2.3 Allied research

There are a number of studies of the valuation of biodiversity and biosecurity that provide insights relevant for this study.

Kerr, Hughey and Cullen (2001) developed a decision support system to manage the environmental externality problem with fishing under the ITQ management system. Twenty two instruments were identified that could potentially reduce the environmental impact of trawling.

Work by Cullen, Moran and Hughey (2005) on public perceptions of the state of the environment and conservation of threatened and endangered species helps to broaden the understanding of the values the public puts on biodiversity.

Proctor and Qureshi (2005) review Multi Criteria Evaluation (MCE) applications to identify methodological issues with conducting MCE for valuing ecosystems. Bell, Thomas, Koller and Hegarty (2002) utilised a method that incorporated both quantitative and qualitative criteria to assist decision makers in allocating resources to biosecurity surveillance in New Zealand. It utilised cost benefit analysis for monetary values and scores for non-monetary values such as environment, society, public health and culture. The qualitative and quantitative rankings were combined in an overall scoring process using MCE.

A project called Better Border Biosecurity (B3) is a large multi-partner cooperative science programme researching ways to reduce the rate at which new pests cross the border and establish in New Zealand. *Inter alia* it has investigated the probability of eradicating an exotic insect species (Kean and Suckling, 2005).

Rolfe and Bennett (2006) bring together a number of studies focused on the Australian environment, highlighting valuation issues and approaches to quantifying and transferring environmental values to new situations.

The only example found of an attempt to include non-market (existence) values into an EIA in New Zealand was for the impact of Didymo. Branson and Clough (2006) took a benefit transfer approach using an average WTP per household per annum for the existence of a region's freshwater assets beyond its use value and scaled this according to the number of nationally important rivers and lakes for recreation relative to a reference region. They multiply these WTP values by the percentage reduction in existence value under low, medium and high scenarios. They also took one fifth of the local WTP and applied this across all households in New Zealand as a national impact. In addition, they took local existence values of single species then applied one fifth of this as the average for all New Zealand households to account for the loss of a species. They qualified their estimates by highlighting that this approach is very simplistic and one which belies the complexity of valuing biodiversity losses and may not accurately represent the public's WTP to prevent biosecurity loss or cumulative species loss due to Didymo. The EIA study was not subsequently incorporated into a CBA of response options.

A project called Marine Value Mapping aimed to close a major gap in the information available on marine values for biosecurity purposes (NIWA, 2008a, 2008b). Four core values were identified for mapping: environmental, economic, social and cultural values. The environmental report documented the findings of the environmental values incorporating the following subcomponents of environmental value: specific diversity, overall biodiversity, non-indigenous species, at risk or threatened species, habitat area, primary productivity, marine mammal distribution, area of marine protected areas (MPAs), sanctuaries and restrictions. The value assigned to each dataset/subcomponent was not a monetary value, but rather a quantitative value to enable comparisons between environmental measures between different areas of New Zealand. In the author's view, the depth of information required to assign monetary values to biodiversity is not presently available on a New Zealand-wide scale.

2.3.1 Gaps and deficiencies

The recently revamped biosecurity response policy has resulted in a world class DSS of policies, processes and guidelines for decision making on allocating resources to a response. However, there are two facets where development could further improve the information available to decision makers. Firstly, quantifying the benefits of response programmes where environmental impacts are significant. This can be broken down into high level values for use at the Investigate stage and use of primary studies at the full CBA stage. Secondly, to more clearly demonstrate the likelihood of different outcomes, and in particular the probability that a response will have a positive welfare effect for society. The policy and management challenge is to incorporate these two facets into the existing DSS. The focus of this thesis is on adapting and developing tools and processes for such a purpose. This is done by utilising stated preference techniques, specifically choice modelling to estimate biodiversity values, benefit transfer to apply the values to new incursions and risk analysis to quantify the uncertainty inherent in the values.

2.4 The way ahead

The next chapter reviews the literature on biodiversity valuation including its place in natural and social capital, the alternative views between neo-classical and ecological economists, and valuation tools. Following this, alternative frameworks for making decisions that are based on community preferences are reviewed. The DSS approach utilises the expertise of specialists across technical fields as the primary input to decisions. An alternative approach, Deliberative Modelling, places the community at the core of the process and uses models that try and mimic the outcomes that different paths take. A third approach, mediated modelling, seeks environmental consensus building through system dynamics. The final section of this chapter develops an economic framework for incorporating biodiversity values into CBA and puts forward an optimal response strategy.

Chapter 3 : Theory and practice of decision support and non-market valuation

3.1 Introduction

Having introduced MAFBNZ's problem of including biodiversity values in biosecurity decisions, there are conceptual and philosophical issues that need to be addressed before there can be any discussion on the appropriate tools for valuing. Questions include: What state of an ecosystem should be protected? How important is social organisation to protecting biodiversity? What is the appropriate overarching framework? How should community values be incorporate into this framework? How should the gap between ecology and economics be bridged?

This chapter initially explores the concept of ecosystem health, then considers biodiversity's place in natural capital. This is followed by an exploration of social capital and its relevance to biodiversity valuation.

A major part of the chapter is concerned with non-market valuation. Firstly, the conceptual economic framework under pinning non-market valuation is outlined. This is a prerequisite to correctly extending CBA to include non-market values. Then the concept of Total Economic Value (TEV) is introduced, which is the way economists categorise the various components of market and non-market values. This is followed by a discussion on how the public's values should be taken into account and how uncertainties can be incorporated into the analysis.

Interactions between ecological and economic concepts have created a tension between some advocates from the two disciplines and attempts to bridge this has led to the development of ecological economics. The ecological economists' world view is considered before returning to mainstream economics where the range of tools for valuing biodiversity is discussed. This leads to an approach that is theoretically robust and practically sensible for implementation by MAFBNZ.

The penultimate section of this chapter discusses three possible frameworks for decision making; Decision Support Systems, Deliberative Modelling, and Mediated Modelling. Implications are then drawn for generation of a database of values using case study analysis. Finally, the chapter is summarised before moving on to a case study in Chapter 4 to demonstrate the methodology for deriving biodiversity values for input into a CBA of biosecurity response.

3.2 Ecosystem Health

The question of what are we trying to value is a serious one as it is not altogether clear what this should be. Biodiversity is not just a number of species existing in isolation to one another. Indigenous biodiversity consists of complex associations of plants and animals that are in a dynamic state, constantly adapting to long term change. Questions concerning the state and interpretation of these associations are discussed below.

Hearnshaw, Cullen and Hughey (2005) review the various views concerning nature and conclude there is no one phase or specific assemblage of species within a system state that is ecologically more important or better than another.

Other writers (Kapustka and Landis, 1998; Lackey, 2001), contend that demarcating such features as naturalness, diversity, stability and resilience as "good", while extinction of species and change is "bad", injects subjectivity under the supposed objective guise of scientific research and should be avoided.

Costanza (1992) and Sagoff (1995) contend that the most appropriate means of characterising ecosystem health should be through a set of criteria which reflects the subjective values of society. Furthermore, the health of an ecosystem should be treated as a "normative" concept because ultimately society has to decide what state is considered "good". In addition, certain axioms of economic theory require that all societal values and preferences concerning resource use are morally equivalent and thus, decisions made concerning resource use should be determined solely in a market environment (Randall, 1988) i.e. it follows the laws of supply and demand.

Hearnshaw *et al.* refer to Leopold (1939) who as "an unfaltering conservationist all his life …stated; when land does well for its owner, and the owner does well by his land; then both end up better by reason of the partnership, we have conservation. When one or the other grows poorer, we do not" (p. 294). They argue that the economic concepts of supply and demand (and costs and benefits) are what truly matters. In this context, sustained supply and demand over the long term is conservation at equilibrium.

Hearnshaw *et al.* put forward a simple theoretical framework which is a high level of abstraction of a social utility model. Their framework assumes that society wants to maximise its "utility" (and therefore ecosystem health) from ecosystem management subject to certain budgetary constraints. However, this model will only determine the most desirable and efficient state of a portfolio of ecosystems at a point in time. What is needed is to model efficiency at future points in time. This is best done by modelling the ability of the system to adapt to changes in the environment (Potts, 2000), and adopting management systems that maintain the capacity to adapt to ever changing ecological conditions (Reid, 1994).

Such adaptive management is based on the ideologies of co-evolutionary development (Gunderson, Holling, and Light, 1995; Holling, 1978) and differences between how the future actually unfolds and how it was envisaged, are seen as opportunities for learning (Lempert, Schlesinger, and Bankes, 1996). This is the way nature achieves "balance"; that is, through maintaining and self organising its complexity. Hearnshaw *et al.* conclude that the concept of ecosystem health is fundamentally normative and best modelled by an economic paradigm through a social utility function, in a way that captures the ever changing dynamics of nature.

More generally, issues arise trying to apply values on behalf of future generations. What would enlightened, forward thinking leaders have done during the Industrial Revolution when Britain's forests were cut down for fuel? Would they have stopped development to protect the environment? What sort of society would we have now if that had happened? No one foresaw the technological developments that have enabled our current lifestyles and supported today's global population at a far higher standard of living than common people enjoyed then. How can we know what the future will bring? Should it look after itself? Or should we try and impose our values on future generations who will live in completely different environments?

These are some of the dilemmas faced when considering contemporary environmental challenges.

If society is to decide what state of ecosystem health including indigenous biodiversity is relevant for valuation, then it must specify exactly what should be counted. The next section, considers both natural and social capital.

3.3 Valuing natural and social capital

3.3.1 Money as the metric

Economists typically use money as the metric to place relative values on goods and services. However, there are challenges when attempting to estimate "values" for assets for which there are no explicit markets.

Hatfield-Dodds (2005) argues existing decision support tools to facilitate the optimal allocation of conservation resources are not adequate. His focus is on the conventional economic wisdom that non-market values should be valued in dollar terms if they are to be considered in government policy-making. He claims this is based on the assertion that it is marketed goods and services that matter and non-market aspects are less important. These considerations undoubtedly influence the use or non-use of various tools, but possibly more influential is the difficulty and additional expense of applying non-market valuation tools.

He asserts that the appropriate application of non-market valuation can only be assessed on a case-by-case basis and it is the quality of analysis rather than quantity that is important. Decision makers should ensure that there is appropriate decision support and that such input is balanced and open to both expert and lay scrutiny. Decision makers must be held accountable and responsive to the community.

The move to broaden the scope of valuation is in part based on questioning the assertion that increased monetary wealth is equivalent to increased social utility. Happiness research indicates that everything else being equal, those above the norm feel happier than those below. But, the norm changes over time and this creates a hedonistic treadmill that prevents sustained increase in perceived well-being (Easterlin, 1973). It throws into question the use of market values as the prime measure of welfare, and it is particularly relevant to choices involving non-market considerations.

Hatfield-Dodds lists the attributes of ideal tools for analysing complex decisions in highly contested situations where data is incomplete or poorly understood. Tools should be based on robust empirical data; enhance transparency; avoid any loss of confidence in underlying data; recognise trade-offs; give insight into the distribution of social values; and be compatible with a range of methods. Existing decision support tools typically do not meet all these requirements.

He quotes Arrow's contention (1963) that it is impossible to construct a coherent social welfare function from individual preferences because of incommensurate preference ordering across individuals. This implies that social values are expressed through both the rules of society and prices in the marketplace.

Walzer (1983) espoused the view that there are limits on the use of money in allocating resources in a western democracy, such as the unacceptability of bribes. This idea is relevant to the concept of a license to pollute. If money cannot capture total "value" then other values such as social sanctions, participation, esteem and culture need to be included in the objective function, although there are limits to their usefulness.

Traditional cost benefit analysis ignores distributional impacts. Yet, in high income societies, such as New Zealand, distributional and other non-monetary factors are often given weight in decision making. There is thus a need to broaden the base of analytical tools to account for this.

According to Hatfield-Dodds (2005) a key research challenge is the testing of alternatives to monetised metrics for estimating relative values. Also to be addressed is the issue of acceptable upper and lower boundaries of social choices. In his view, if such alternatives can be developed they would provide an important input into decision making, enhancing transparency and improving public confidence in attaining robust resolution to decisions in highly contested areas. These issues are seen relevant to the management of exotic pests and diseases which could adversely affect New Zealand's indigenous biodiversity.

3.3.2 The value of natural capital (indigenous biodiversity)

Hatfield-Dodds and Pearson (2005) observe that it is critical to be able to recognise a decline in natural capital as it is now widely agreed that natural capital plays an important role in underpinning present and future well being. Furthermore, identifying whether development (increased income) is sustainable requires the measurement of changes to natural capital.

A definition of sustainable development is provided from the World Commission on Environment and Development, (WCED, 1987) "that which meets the needs and aspirations of the current generation without impairing the ability of future generations to meet theirs" (p. 40).

A difficulty with this definition is, however, the need for the current generation to make value judgements about the needs and aspirations of future generations. In contrast, the definition of Pearce, Barbier and Markandya (1990) which states that "sustainable development is a process of change in the economy that ensures that welfare, aggregated from economic, social and environmental dimensions is non-declining over the long-term" does not require value judgements about future generations' needs and aspirations.

Hatfield-Dodds' view of sustainable development is framed in terms of maintaining total capital stocks. This follows the Hartwick-Solow proposition that the value of the total stock of capital assets available to society may be maintained by investing the scarcity rents associated with any net depletion of natural resources into human or built capital. This is as long as these forms of capital are substitutable and so enable an equivalent stream of human well-being over time (Becker, 1982; Hartwick, 1977; Solow, 1986). Solow also argues that technological change, interpreted as improvements in the relationship between the total capital stock and human well-being, must at least keep pace with population growth if per capita well-being is to be maintained. So far the world has been more than able to achieve this goal.

The best-known formal definition of capital dates back to Hicks' (1946) distinction between capital and income, which defined income as the maximum benefit stream that can be enjoyed in a period without reducing the value of the underlying capital assets or stock available for future enjoyment. Income is thus analogous to the interest paid on a bank account. With this as a basis, Schumacher (1973) defined natural capital as the irreplaceable capital which man has not made, but simply found.

Critics of development often draw attention to the importance of maintaining the integrity and productivity of ecosystems and natural resources that underpin current and future human well-being. In New Zealand, the report of the Parliamentary Commissioner for the Environment entitled *'Growing for Good'* (Parliamentary Commissioner for the Environment, 2004) made this very point. The Commissioner's primary concern was that in striving for increased profits farmers were intensifying land-use to the point of mining their natural capital in an unsustainable way. Criticisms that could be levelled at the Commissioner's report are the lack of recognition of the role of technological change in substituting for natural capital, and a lack of analysis of the counterfactual. A private sector initiative is looking at this issue and identifying best management practice that can result in increasing profitability with improved environmental outcomes. The publishing of the dairy industry's strategy (Dexcel, 2006) on sustainable development is a significant step towards environmentally sustainable dairying. Preservation of indigenous biodiversity located on farms is part of the strategy.

3.3.3 Social capital and indigenous biodiversity

Social capital is the value that social relations provide by underpinning cooperation and collaborative action and assisting individuals and groups achieve their goals. It includes networks, norms and more formalised rules for institutional arrangements (Winter, 2000). The benefits generated by these networks are characterised as local public goods or club goods (Ostrom, 2005). In addition, there are also other more formal aspects of social relations, such as legal and regulatory frameworks, institutional arrangements and wider civil and political regimes (Coleman, 1990; North, 1990; Olson, 1982). Social capital includes the institutional memory of organisations, the level of corruption or lack of it, organisational culture and level of transparency and accountability.

Social capital and its measurement are potentially critical components in assessing the costs of protecting indigenous biodiversity from incursions of exotic pests and diseases. This is because of the importance firstly, of individual and special interest groups in biosecurity issues such as identification of new pests; and second, of institutional arrangements to the efficient operation of biosecurity and protection of indigenous biodiversity through the operations of key departments such as MAFBNZ, the Department of Conservation and the Ministry for Environment.

At the national level a number of different approaches have been developed to extend traditional national accounts measures of well-being such as Gross Domestic Product and Gross National Product. These include the genuine progress indicator (Daly and Cobb, 1990) and the green GDP (Economy, 2007); the genuine savings approach (Pearce and Atkinson, 1993); and the inclusive wealth framework

(Arrow, Dasgupta, and Maler, 2003). But as yet, according to Hatfield-Dodds and Pearson (2005) there is no existing measure of sustainable development that includes a comprehensive measure of social capital. The authors contend that there is substantial evidence that social capital makes a material contribution to living standards and human well-being in two key areas: through good governance and appropriate regulatory frameworks; and in providing incentives for innovation. The impact of different governance regimes on outcomes in, East and West Germany (communism and western democracy), Zimbabwe and the Asian tigers (dictatorship and Asian democracy) illustrate this point.

Partitioning natural capital from other forms of capital has been argued on the basis that loss of certain types of natural capital may be irreversible, and functional substitutes are not available (Pearce, Markandya, and Barbier, 1989). Other arguments include statements that natural capital is multifunctional, that the impacts of resource depletion and ecosystem modification are often poorly understood (counselling a precautionary approach) and that access to natural capital is often important on equity grounds. Latterly emphasis has been on assessing 'critical capital' some of which may not be natural capital (Hatfield-Dodds, 1998; Pearce and Warford, 1993). Other writers have emphasised that the major threats to natural functions derive from inappropriate institutional arrangements (Bromley, 1991; Soderbaum, 1992). They have called for the development of policy tools and institutional arrangements to meet such challenges. Parminter (2009) has focused on explaining social behaviour using the theory of reasoned action. This is considered to show promise in predicting human behaviour change in response to policy interventions on indigenous biodiversity.

The above discussion shows that changes to social capital may have a significant impact on natural capital. Yet to be tested is the proposition that protecting important aspects of social capital is likely to be dramatically more cost effective than seeking to replace it once lost as the case with natural capital. The benefit of recognising the role of social capital in sustaining indigenous biodiversity may be high given its significant potential contribution to effective biodiversity protection and enhancement.

3.3.4 The impact of including social capital as a well-being criteria

According to Hatfield-Dodds and Pearson taking social capital into account is likely to have benefits in at least three areas. First, it can help to identify opportunities where enhancing social capital would promote community well-being and sustainable development. Second, it can help prevent the implementation of simplistic policy prescriptions that give inadequate attention to social and institutional factors. Third, it can assist the development of sound development strategies by helping to identify the spectrum of advantages and disadvantages associated with alternative policy approaches.

They conclude that at least some aspects of social relations satisfy a capital test. Social capital complements but is distinct from built, human and natural capital. Other types of capital are able to provide some but not all of the functions provided by social capital. Important aspects of social capital appear at risk of being eroded due to weak incentives and the lack of explicit management structures for investment in conservation. Some social capital warrants protection as critical capital as it is threatened, non substitutable, and would be difficult or impossible to replace if lost.

3.3.5 Summary

Well-being goes beyond income and includes social attributes; hence policies for protecting indigenous biodiversity should consider social as well as monetary values.

Indigenous biodiversity is a form of natural capital. The sustainability of natural capital is a key concern of many people and in the limit cannot be replaced by human or manufactured capital. In valuing biodiversity, a key question is whether, at the margin, that capital is at risk for it is then that a change in value is likely to be significant to society.

Also at issue is whether the form of natural capital at risk has any particular attributes that make it more or less valuable today than compared with its value to a previous or future generation.

Superficially, decisions about biodiversity may seem to be only about natural capital, but the discussion above highlights the importance of social capital, which not only includes the community, but the bureaucracy as well. These are difficult considerations to model although explicit consideration of them is likely to improve the quality of the conversation on resource allocation and decision making for the benefit of society.

In conclusion, including social capital in decision making about the threat of pests and diseases to indigenous biodiversity is likely to provide additional insights. What is needed is a tool which has the ability to incorporate a range of values, including social values, in the overall valuation of indigenous biodiversity. Furthermore the tool should involve community participation in estimating biodiversity values, which would help maintain and enhance social and natural capital. The next section considers the economic theory that underpins non-market valuation.

3.4 Non-Market Valuation

3.4.1 Conceptual framework

Correctly applying non-market valuation (NMV) techniques to CBA requires an understanding of the under pinning economic theory because it forms the basis of the goals of NMV. The theory is based on a model of individual choice that explicitly recognises the public good nature of non-market goods (Flores, 2003).

From the individual's point of view it is a basic premise of neoclassical economics that people have preferences for goods (both market and non-market) and that these can be ordered to maximise the individual's utility. This can be specified by the individual's utility function (Flores, 2003). The optimal amount of biosecurity services, which are non-market goods (q_i), is determined in conjunction with market goods (x_i) based on the individual's utility function.

Utility function

$$U = U(x_1, x_2, x_3 ... x_n; q_1, q_2, q_3, ... q_k) \qquad (3.1)$$

The utility function assigns a single number U(X,Q) for each bundle of (X,Q).

For any two bundles (X^A, Q^A) and (X^B, Q^B) the respective numbers derived from the utility function are such that $U(X^A, Q^A) > U(X^B, Q^B)$ if and only if (X^A, Q^A) is preferred over (X^B, Q^B).

For market goods, individuals allocate their budget based on their preferences, relative prices of market goods $P = [p_1, p_2, p_3, \ldots p_n]$ and available income y. Non-market goods are rationed as individuals cannot unilaterally choose the level of these goods. People make choices to maximise utility from their income by purchasing market goods subject to a rationed level of non-market goods.

$$\underset{X}{Max}\, U(X,Q) \text{ subject to } P'X \leq y,\ Q = Q^0 \qquad (3.2)$$

For each market good there is an optimal demand function defined as

$$x_i^* = x_i(P,Q,y) \qquad (3.3)$$

and the vector of demands is

$$X^* = X(P,Q,y) \qquad (3.4)$$

Inserting the optimal level of demand into the utility function yields

$$U(X^*,Q) = v(P,Q,y) \qquad (3.5)$$

Demand functions provide the quantity of goods demanded for a given price vector and income level. They can also be interpreted as marginal value curves, since consumption occurs up to the point where marginal benefit equals marginal cost (Flores, 2003, p. 29).

Flores' basic model can be adapted to the problem of biosecurity where non-market goods such as indigenous biodiversity are involved. The first measure is compensating surplus (CS) which is the amount of income an individual would be willing to give up after the project has been implemented that would exactly return him or her to the previous level of utility. This can be shown algebraically:

$$v(P^0, Q^0, y^0) = v(P^1, Q^1, y^1 - CS) \qquad (3.6)$$

The superscript (^{o}) identifies the initial conditions and (^{1}) the conditions after implementation. CS is the amount of income willing to be given up to return the individual to the position before the project was implemented. CS could be positive or negative depending on how relative prices change and/or the size of any additional taxes paid. If costs are less than CS then the project should be implemented as the individual is better off. If costs are more than CS then the individual is worse off.

The second measure is the equivalent surplus (ES) measure and is the amount of additional income an individual would need before the project was implemented to have the same utility as after it was implemented i.e.

$$v(P^0, Q^0, y^0 + ES) = v(P^1, Q^1, y^1) \qquad (3.7)$$

"The two measures differ by the implied assignment of property rights" (Flores, p. 30). For the CS measure the initial utility level (status quo) is the basis of comparison and for ES the subsequent conditions (the change) are relevant. Which measure is appropriate depends on the situation. When considering a new policy to improve a situation CS is used, for example, restoring indigenous biodiversity after a long period of possum damage or response to a new pest invasion. Alternatively, when the property right already exists and the project would return the situation to that level then ES is appropriate, for example, the income required to accept the possum damage or the damage from the new pest.

There are two other terms that are often used to substitute for compensating and equivalent measures, namely willingness to pay (WTP) and willingness to accept (WTA) compensation. WTP is usually associated with a positive change and WTA with a negative change. It should be noted that considerable controversy has surrounded the construct validity of WTP and WTA as the latter construct may give values “several times larger” than the former (Freeman, 2003b, p. 178). WTP and WTA are used to describe impacts on individuals. Benefits in CBA are the sum of WTP for changes that are perceived as gains and WTA for changes that are perceived as restored losses. On the other hand, costs are the sum of the WTA changes that are perceived as losses and WTP changes that are perceived as foregone gains.

In order to ensure as many of the values of a natural resource that are of concern to people are included in economic analysis economists have categorised then according to use, which is captured in the term Total Economic Value (TEV).

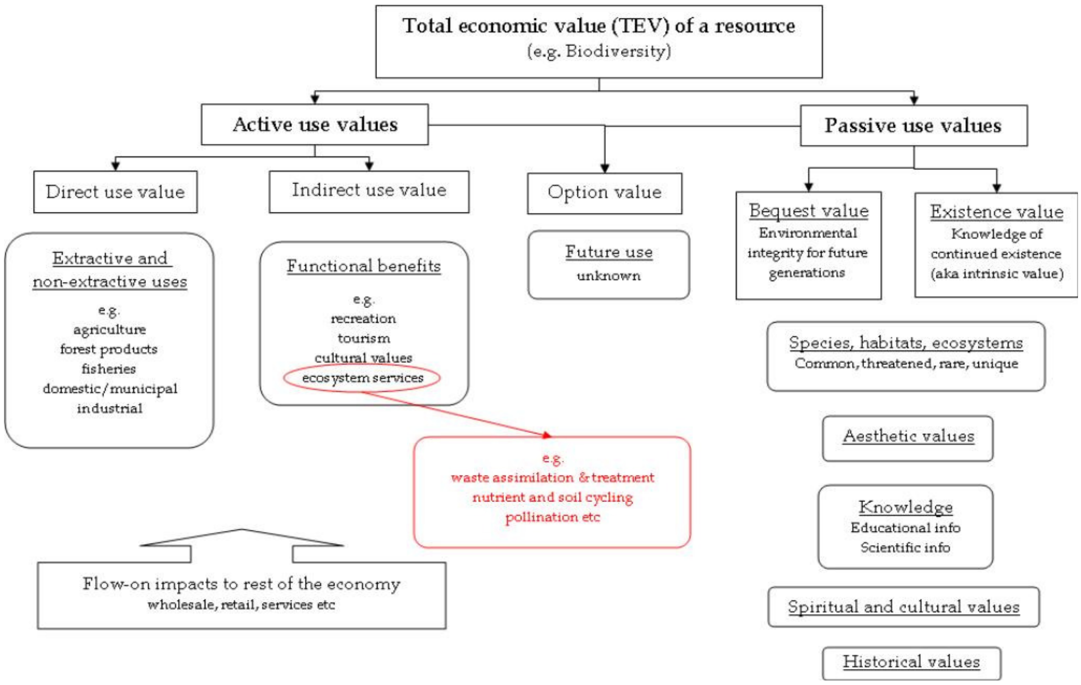
3.4.2 Total Economic Value

Typically economists have focused on values that have prices in markets when evaluating changes to the use of natural resources. These values are termed active use and include direct uses such as agriculture, forestry and fisheries, and indirect uses including recreation, tourism and eco-system services.

Passive use values, which do not have market prices and are thus by definition non-market values, typically are not quantified in these analyses. Passive uses include aesthetics and cultural and spiritual values. They are divided into bequest values (integrity for future generations) and existence values (knowledge of continued existence). Option value is the term used to describe the value of some future use that is unknown, which may be active or passive. The full range of such values is encompassed in the concept of Total Economic Value (TEV). Figure 3-1 set out the different values that make up the TEV of biodiversity.

Often passive use non-market values are those held by the public or society as a whole compared active use values that are captured by individuals or firms. Not quantifying non-market values thus has the potential to under account for the values of large segments of society when making decisions about natural resource use. Taking account of the public's values is discussed next.

Figure 3-1 Total Economic Value



Source: adapted from EVRI (2009)

3.4.3 Taking into account the public's values

In the first line of the preface to their book *A primer on nonmarket valuation* Champ, Boyle and Brown (2003, p. ix) state: "Public policy should reflect an understanding of the public's values". Until recently, bureaucrats and politicians have largely determined this indirectly. However, the trend is for the community (civil society) to have an increasing role in policy formulation through participatory processes and use of tools such as mediated modelling (explored below in section 3.6.3).

In a world of scarcity, choices must be made about how to manage the human impact on natural systems. Increased protection of natural systems results in less of something else with resources being diverted from other ends, implying tradeoffs (Champ e*t al*.). To claim that natural ecosystems are invaluable, leads to the unrealistic proposition that an infinite amount of resources could be dedicated to their protection. More realistically, to determine how much can be justifiably spent on protecting an ecosystem, we need an estimate of its value to society. This proposition is at the heart of this research.

Champ *et al*. highlight key issues to be addressed when valuing natural systems: 1) "if individuals do not understand the contribution that an ecosystem makes to their well-being, then their observed behaviours or responses to questions will reflect that ignorance rather than the true value of the ecosystem to them; 2) individual choice and response to questions of valuation are constrained by income. This leads to proposals to include equity and fairness criteria into policy evaluation; and 3) preference orderings may be different from the perspective of an individual's civic or consumer viewpoints, but unless there is evidence to the contrary the standard economic assumption (that of the individual) should be the basis for economic evaluation" (p. 13-15).

Conventional cost benefit analysis is an application of the Kaldor-Hicks potential compensation test which states: "a policy is accepted if in principle, those who gain from the intervention could transfer income to those who would lose" (Champ e*t al.,* p. 16). The winners and the losers do not have to be identified or the compensation paid. The Hicks (1939) test measures changes in welfare associated with the project and the Kaldor (1939) test measures changes in welfare without the project. Consider the situation where a regional council is contemplating imposing regulations to improve lake water quality by reducing nutrient pollution. The starting point is that society has a "right" to clean water. The amount of compensation they would need to forego for improved water quality is the monetary equivalent of the increase in welfare associated with clean water. In this case equivalent measures of welfare change are appropriate and the proper test is the Hicks version of the potential compensation test.

Because a measure of the aggregate net benefit does not incorporate distributional consequences, many economists argue that it should not be the sole basis for a decision rule. In New Zealand, the result of a cost benefit analysis is only one of a number of criteria used to evaluate policies for natural resource management.

Also, aggregate net benefit usually only counts active use values derived from market prices. This misses out non-market passive use values. Flores (2003) notes that stated preference methods including contingent valuation and attributed-based methods such as choice modelling, which draw inferences from hypothetical tradeoffs "are the only viable alternatives for measuring non-use or passive use values" (Flores, 2003, p. 48). Otherwise analysts must rely on relative (non-monetary) values which can be derived from multi-criteria analysis (MCA).

"The important social issue is the need to incorporate the values of all those who value the non-market good." (Champ e*t al.,* p. 49). This creates a logistical and financial problem for stated preference analysts who have to decide on the boundary of their surveys. Should it be in the local vicinity, the region, the country or indeed the world? Judgement must be applied as to what is reasonable and financial constraints usually determine the boundary at the limit, resulting in a conservative estimate of value.

Willingness to pay for environmental improvements or to protect existing ecosystems is often presented in the abstract form, for example as hypothetical special taxes. But there are also examples of willingness to pay working in tangible ways. For example, bond issues have been used for the provision of open space, usually with a single specified payment over time. In this case the amount of open space that can be purchased is uncertain. Uncertainty is a feature of stated preference surveys, which generally estimate option price the analogue for compensating surplus (Flores, 2003, p. 52).

Willingness to pay and willingness to accept compensation are now standard approaches to the non-market valuation of ecosystem goods and services. These are generated through stated preference surveys. However, a number of contentious issues remain and analysts need to ensure that they are addressed (see section 3.4.8 on stated preference techniques).

3.4.4 Taking account of uncertainty in value estimates

Typically, biosecurity decisions are based on best estimates (mean values) with little quantitative consideration of the variation in these estimates. Providing estimates of the variance of the net benefit when comparing response options offers decision makers considerably greater insights than a focus on the mean alone. For example, decision makers would be expected to prefer project A if the net benefit exceeds that of project B, but if the variance of project A is considerably greater than project B then this may not be the case. The uncertainty around whether the net benefit of project A is negative could be greater than project B and this may not be acceptable.

Such uncertainties can be taken into account by assigning probability distributions to key variables and using the Excel add-in @RISK to simulate the overall probability distribution for the project. Nimmo-Bell has developed a standardised framework to undertake this analysis in a consistent way that has been tested rigorously in the evaluation of research and development projects at the industry level (Bell, 2003). Application of the method is demonstrated in Chapters 4, 5 and 6.

Non-economists have approached the problem of valuation from a bio-physical angle, while the ecological economists, have tried to bridge the gap between economics and bio-physical disciplines. The issues they have been grappling with are outlined in the next section.

3.4.5 Neoclassical Techniques versus Ecological Economics

At the core of the problem of valuing ecosystems is the divide between the two disciplines of ecology and economics. While the ecologist's perspective lacks consideration of social processes and human preferences that guide resource use, economists tend to ignore the bio-physical and ecological processes that sustain ecosystem services. A special issue of Ecological Economics has been devoted to issues around valuation and the integration of economics and ecological perspectives (Costanza and Farber, 2002).

The ecological economists have tried to incorporate the qualitative attributes of ecosystem services that they perceive are not embodied in monetary values. They endeavoured to broaden the perspective of the neoclassical models, and to incorporate the bio-physical. The following discussion highlights the key issues.

Ecological economics addresses the relationships between ecosystems and economic systems (Costanza, 1991). Its core issues include the sustainability of interactions between economic and ecological systems, the social fairness of resource distribution, and the allocative efficiency of the economy when natural capital, fairness, and sustainability are taken into account. Ecological economics thus involves issues that are fundamentally cross-scale, trans-cultural, and trans-disciplinary, all of which call for innovative approaches to research, policy and social institutions (Costanza, Cumberland, Daly, Goodland, and Norgaard, 1997; Costanza and Daly, 1987). Resource allocation for protecting biodiversity from biosecurity risks involves many of these concepts.

There is a question around whether biosecurity policy should attempt to embrace the concept of a shared vision of a sustainable future (Senge, 1990). Arguably the current vision in New Zealand that receives most attention is that of a "100% pure or clean green New Zealand". Such a vision has obvious implications for biodiversity valuation, because people generally are likely to assign significant weight to these values.

Hannon (2001) attempts to integrate ecological and economic theory for the estimation of "technical system efficiency" through an input -- output framework. His technical system efficiency ratio is defined as the ratio of the monetised value of the combined net input to the monetised value of the net outputs. But Jollands' (2006) view is that Hannon's approach is so methodologically complex and data hungry, even if the approach was ideal, its complexity would likely limit wider adoption.

Patterson (2005a) reviews a range of valuation methods which include: neoclassical (contingent valuation, hedonic pricing, avoided cost, replacement cost, travel cost and factor income); group based methods (citizen based juries, consensus conferences, focus groups and deliberative contingent valuation); other anthropocentric methods, which place humans at the centre of valuation (conjoint analysis, choice modelling and multi-criteria analysis); and biophysical methods (ecological pricing and energy analysis). In his view CBA can give misplaced confidence, as it is typically limited in coverage. This gives rise to a case for the use of a range of tools. As no one method is sufficiently comprehensive the appropriate method depends on the context. That said, there is a lack of methods for directly measuring ecological and intrinsic values.

Straton (2005) points out the limitations of existing valuation methodologies for ecosystems. She outlines two approaches to estimating the economic value of non-market ecosystem services. First, demand-side valuation methodologies ascribe value on the basis of subjective preferences of individuals through their willingness to pay for an additional unit of the service. Second, supply-side valuation methodologies base estimates of value on the actual cost of production of the services provided.

In Straton's view the demand-side approach is limited because it does not consider objective bio-physical properties of ecological resources and supply-side methods fail to provide welfare measures. Usually only one approach is used while both are needed.

Straton proposes a complex systems approach to valuation that incorporates recent developments in economic models of choice.

These models are increasingly influenced by findings from neuroscience, biology, psychology, sociology, anthropology, economics, and game theory. The conceptual framework clarifies the source of value as being "a system" that is made up of components (functionality) and their connective structure (value).

Straton's aim is to capture the two essential elements of quality and capacity, neither of which is adequately captured through monetary valuation. She concludes that the complex systems approach to the allocation of ecological resources is a difficult and ambitious task. On her own admission this places the approach in the too hard basket as far as valuing ecosystems for biosecurity purposes.

Ruth (2006) puts forward six challenges to traditional efficiency based economics in order to find greater acceptance in the wider community: integration of resource and environmental economics; consistency with physical and biological principles; development of a system perspective; acknowledgement of legacy effects; recognition of inter-dependencies of allocation, distribution and scale; and demonstration of policy relevance.

However, Ruth quotes Sagoff (2004) who states that extending research into ever more ecosystem goods and services is not likely to help resolve current conflicts surrounding resource use and allocation. Ruth argues that a better understanding of consumption requires a larger systems context; one in which socio-economic (behavioural), bio-physical and engineering insights are combined. Combining these aspects into one concept of "natural economics" entails four themes: building on concepts from nature; including the roles of efficiency and effectiveness in decision-making; the need for adaptive and anticipatory management; and the need for holistic impact assessments.

In Ruth's view, to develop and select system designs that are sustainable will require a natural economics that establishes the economic, legal, institutional and ethical basis for humans to interact with the environment.

Ruth uses the example of the Centre for Ecological Economics at Palmerston North as adopting this approach. Here the limitation of narrowly defined optimal resource use has been recognised. The Centre attempts to use stakeholder guided explorations of the dynamics of complex human-environmental systems taking into account knowledge processes that occur at different temporal scales and system hierarchies.

Ruth argues that the costs of such system design-orientated activities are small compared with the benefits associated with the consensus they generate across infrastructures and institutions, and the ground they lay for effective implementation of technologies and policies. Expert based, efficiency-driven advice on cause and effect relationships is enriched, and increasingly supplanted by adaptive and anticipatory management systems, oriented and informed by stakeholders. Put like this, Ruth's approach has many of the elements of mediated modelling.

Patterson, Wake, McKibbin and Cole (2005) are motivated to provide a new system of ecological pricing (eigenvalue-eigenvector) because of the limitations of the neoclassical approach. They argue that the neoclassical approach systematically undervalues (or ignores) some species and ecological processes, as the approach is dependent on human valuers who have imperfect knowledge. Their method builds on a Sraffa-type approach (Sraffa, 1960) which objectively measures the flow of mass and energy between species as an indicator of the interdependencies between species. This then becomes the basis for objectively measuring the contributory value of species (ecological price), in terms of how one species contributes to the value (livelihood) of another species.

Ecological prices derived by the model are determined by the biophysical interdependencies or linkages inherent in the system. They ascribe three main advantages in using the eigenvalue-eigenvector method: 1) numerical relativities of ecological prices don't vary according to the choice of dependent variable; 2) the method does not require the arbitrary aggregation of equations to form a square matrix and hence define a unique set of ecological prices; and 3) the method does not assume equilibrium prices. While appearing intuitively appealing and mathematically tractable, application to the valuation of indigenous biodiversity would require estimates of values of inputs and outputs for individual species and whole ecosystems.

Ecological economists such as Costanza, Jollands, Daly, Hannon, Peet, Patterson, Straton, and Ruth struggle with the complex and theoretically difficult issues of extending neoclassical economic models to include environmental and social values. They are as much concerned with issues such as sustainability and fairness as dollar values. Their models set the stage for alternative approaches of valuing indigenous biodiversity, but as yet there is little evidence of use in actual decision making. Much of what they propose would involve time and resources beyond that which is feasible in the context of MAFBNZ. However, their work does remind us of the complexities of ecological systems and the range and potential size of values at risk. For these reasons their contribution should not be overlooked.

3.4.6 Multi-criteria Analysis

A tool that has been used widely to value quantitative benefits and costs in a structured way is multi-criteria analysis (MCA). It has been used in a number of applications including transportation, the environment and biosecurity. In the analysis below its strengths and weaknesses for economic analysis of incursions of exotic pests and diseases are discussed.

Proctor and Qureshi (2005) provide a taxonomy for multi criteria decision-making models. The distinction is made between discrete models where alternatives are predefined and continuous models where alternatives are generated by mathematical models. Under the discrete line, the distinction is made between quantitative (or cardinal methods) and qualitative (or ordinal methods). However there is no reference to methods that incorporate both quantitative and qualitative criteria.

Bell, Thomas, Koller and Hegarty (2002) utilised a method that incorporated both quantitative and qualitative criteria to assist decision makers in allocating resources to biosecurity surveillance in New Zealand. This approach utilised CBA for monetary values and scores for non-monetary values for the environment, society, public health and culture. The quantitative and qualitative rankings were combined using an overall scoring process. As a further aid to decision makers, the maximum value of the portfolio of projects based on the results of cost benefit analysis alone was estimated by taking the net present value of the sum of projects within the funding envelope. This was compared with the value of the projects chosen utilising multi criteria analysis and the difference between the overall sum of net present values indicated the dollar trade-off between using monetary values alone in decisions and including non monetary values as well.

The use of MCA as a tool for measuring environmental values gained traction in the debate over sustainability. According to Hearnshaw, Saunders and Dalziel (2004) this represents a dynamic co-evolutionary process (where development is considered as adaption to a changing environment while itself being a source of change) towards a more ecologically sound and equitable way of being (Norgaard, 1994; Ring, Klauer, and Watzold, 1999). The definition also allows for the trade-offs required to make decisions under a funding constraint.

MCA has been proposed by some analysts who are not convinced that economic values can be inferred for non-market goods such as the environment, human health, culture etc. MCA places relative values on different components making up a multi-criteria function. These are expressed as scores in a range, for example, 1 to 100. The scores for each component are added up to form a composite score. The scores may be weighted with equal or different weights. The Analytical Hierarchy Process (AHP) is a technique for determining weights based on the pair wise preferences of individuals.

A major criticism of MCA is that as it lacks absolute objectivity the problem cannot being fully defined mathematically (Arrow and Raynaud, 1986; Dyer, 1990).

Geoff Kerr's (*pers. com.*, 2006) main concern about MCA is that it is often applied in a way that conceals values, rather than openly revealing the values of stakeholders or decision makers. In his view it can be a method for subverting either the democratic process and/or achievement of economic efficiency. Furthermore he states that it can also cause double counting. In particular, "if CBA results enter the MCA it is important that anything that entered the CBA does not turn up in any of the other criteria, which is pretty much impossible to achieve".

Kerr notes that a valuable feature of a choice modelling experiment is that it can be used to derive the weights that enter an MCA. A study by Bennett and Blamey (2001a) illustrates trade-off rates in non monetary terms between different attributes that are likely to enter an MCA objective function (jobs, regional income, hectares of wet land, etc.).

One of the key issues arising with implementing multi-criteria analysis is deciding on the weights assigned to economic, environment, social and cultural elements. AHP is an approach that offers theoretical rigour and practical application to this issue.

Hearnshaw, Saunders and Dalziel (2004) set out six attributes necessary for a method to be suitable as a theoretical framework for composite indicator construction. It must allow for the weighted aggregation of quantitative individual indicators, which requires that the method is utility or value based, quantitative in format and provides a cardinal measurement of the weighted differences among indicators and not merely ordinal differences; it must facilitate a participatory process; it must be transparent so that the method of construction can be disseminated for robustness; be internally consistent; be flexible in methodology; and be easy to use. They proposed the AHP as a method that encompasses these attributes. It was derived from operations research (Saaty, 1977, 1980) and is a multiple step analytical process of judgement, which synthesises a complex arrangement into a systematic hierarchical structure . Since then there have been many thousands of diverse applications in which the AHP method results were unreservedly accepted (Saaty, 1994).

Hearnshaw *et al.* state that "the most suitable approach for the generation of evaluation elements is where open discussions or BOGSAT (bunch of guys sitting around a table) (Peterson, Schmoldt, and Silsbee, 1994) are used. However, it is ultimately an individual evaluation that is employed to determine appropriate weights to be attached to evaluation elements (Hwang and Lin, 1987).

SWOT analysis can be used to help complete the set of individual indicators (OECD, 2003). Each indicator's analytical soundness, measurability, cost effectiveness and time-series completeness should determine its ultimate usefulness for evaluation (Maclaren, 1996).

Software such as Expert Choice (http://www.expertchoice.com/) is available for extracting the relative weights and is specifically designed for AHP.

However, aggregating indicators into a single index results in only a "weak" measure of sustainable development, the alternative, which is to have separate economic, social and environmental indicators, in contrast to one composite indicator, is more transparent and therefore may be more useful for policymaking (Den Butter and Verbruggen, 1994).

There is a large body of research critical of AHP. The most critical objection is that it lacks a complete axiomatic foundation. A particular shortcoming is that it lacks transitivity (Dyer, 1990; Dyer and Wendell, 1985). This arises when an indicator j can be preferred to an indicator k and k to indicator l while l can still be preferred to j. There is also the problem of rank reversal where the ranking of a set of indicators changes upon the introduction of an" irrelevant" indicator. Software such as 1000Minds (http://www.1000minds.com/) may overcome much of this criticism. Another notable criticism stems from the fact that aggregation models of preference can impose a very restrictive structure on the respondents. It is implicitly assumed that each preference is assessed independently of all other stakeholders. This independence assumption ignores implicitly the dynamics present with group settings. Nevertheless, this approach to group decision making remains the most widely accepted.

In summary, while having appeal as an easily understood and easily applied technique MCA suffers from a number of deficiencies compared with economic tools such as choice modelling. First, it tends to result in black box type decisions where the underlying basis for scores is hidden. Second, the technique is open to double counting particularly where a rigorous economic evaluation has been conducted using a tool such as choice modelling. This is because the economic values expressed by individuals embody more than just market values as expressed through WTP and WTA concepts. Third, there is no satisfactory way of determining weights that is not open to criticism regarding their stability. Lastly, in practice, the scores for different projects particularly when they are close offer little insights for decision makers to differentiate between projects in practical decision making (Wansbrough *pers. com.*, 2006).

In conclusion, AHP appears to be a step forward for determining weights in MCA. Most of the concerns regarding AHP can be accommodated, and awareness of them by the practitioner will allow respondents to rationalise their views in most cases. Perhaps the key concern is that of double counting with monetary values and the likelihood that if monetary values have been specified then it is highly likely non-monetary values will already have been included at least in part in the monetary values. For this reason alone MCA is not recommended for estimating values of biodiversity.

The following two sections discuss revealed preference and stated preference as potential valuation tools for biodiversity. The principle distinction between these two methods is the source of the data (Mitchell and Carson, 1989). Revealed preference relies on observing individual's actual behaviour while stated preference relies on their responses to hypothetical questions (Freeman, 2003, p.23).

3.4.7 Revealed preference

"Revealed preference methods draw statistical inferences on values from actual choices people make within markets" (Boyle, 2003b, p. 259). There are four commonly used methods: travel cost, hedonics, defensive behaviour and damage cost. Unfortunately, these are of little use to valuing indigenous biodiversity in natural ecosystems as no one has developed a means of using them to estimate non-use or passive values. But indigenous biodiversity has values other than non-use: the travel cost method can be used to value specific iconic ecosystems in national parks; and hedonics can be used to value unique landscapes where people build holiday homes. "The key to using the various revealed preference methods is to identify first the change to be valued then the affected groups" (Boyle, 2003b, p. 263).

3.4.8 Stated preference

“The basic idea behind any stated preference technique for estimating non-market environmental values is to quantify a person’s willingness to bear a financial impost in order to achieve some potential (non-financial) environmental improvement or to avoid some potential environmental harm” (Bennett and Adamowicz, 2001, p. 38).

In its first use, the technique applied as contingent valuation (CV) relied on asking people directly what they would be prepared to pay for a change to an environmental service. This was in the form of a yes or no question to a certain amount. Would you be willing to pay $x for…? (Freeman, 2003 p. 25).

Boyle (2003a) provides a succinct review of the history of CV starting with Davis (1963) who used the tool to value big game hunting in Maine. In the author's view, Mitchell and Carson (1989) provided the most substantive contribution to the design of a CV study.

Use of CV came under intense scrutiny by Hausman (1993) supported by Exxon, who critiqued the fundamental premises of CV over the Natural Resource Damage Claim for the Exxon Valdez oil spill. The National Oceanic and Atmospheric Administration (NOAA) responded with a high level panel which evaluated the use of CV to estimate non-use values (NOAA, 1993). The panel made specific recommendations on how a CV study should be designed and conducted to develop "reliable" estimates. The Exxon case was settled out of court and the CV study was not tested in court; however the NOAA recommendations set off another round of research.

Boyle in his analysis focuses on post-1990 research using the valuation of ground water as an example. He states that "reliability of CV estimates is not an issue of concern" (p.155), and concludes that "the influence that the researcher has over the design and ultimately the outcome of any CV study is not any different from any other line of research or empirical analysis. Simply put, CV like any other empirical method requires a high degree of skill and considerable art that must be combined with careful pretesting and validity checks" (p. 158). Much of the criticism of CV arises from the focus on a single attribute and the consequent framing issues that arise. CV is highly dependent on the way the valuation question is framed and there is a concern that it is too easy for analysts to engineer a desired answer even if inadvertently. Attribute-based methods (ABMs) such as choice modelling (CM) overcome much of this potential criticism.

ABMs are a relatively new class of stated preference tools that have emerged from creative links across marketing, psychology, transportation and economics disciplines (Holmes and Adamowicz, 2003, pp. 171-219). They are "used to estimate economic values for a technically divisible set of attributes on an environmental good ... that can provide resource managers and policy makers with detailed information about public preferences for multiple states of the environment. The inclusion of price as an attribute permits a multi-dimensional valuation surface to be estimated for use in benefit-cost analysis" (p. 171). ABM techniques combines elements of consumer demand theory (Lancaster, 1966) and conjoint analysis (Luce and Tukey, 1964).

A further development by McFadden (1974) took advantage of earlier insights and rapidly increasing computing power. He used discrete choice theory to simplify the choice process and gave it a stronger economic foundation. In a major step forward he combined hedonic analysis of alternatives with random utility maximization into an econometric model known as the multinomial logit (conditional logit) model (MNL). A further advance allowed the assessment of welfare measures.

Environmental economists have used ABMs in three major response formats: rating, ranking and choice. Attribute based experiments tend to be based on random utility maximization theory (RUM) where the choice behaviour is deterministic from the individual's view, but stochastic from the researcher's perspective. This provides the theoretical basis for empirical models based on consumer choice between competing alternatives. Analysis of ranking alternatives is based on random utility theory. Rating data are most often assumed to contain information on ordinal rather than cardinal preferences and although subject to simple econometric analysis are not recommended for environmental valuation as using choice or ranking is more direct.

ABMs provide estimates of the indirect utility function and are thus useful to calculate welfare measures for gains, losses, or any combination of changes in attributes for policy analysis using utility maximization theory (Holmes and Adamowicz, 2003).

One form of ABM, conjoint analysis, was developed from a "measurement technique in mathematical psychology for decomposing overall judgments regarding a set of complex alternatives into the sum of weights on attributes for the alternatives" (Holmes and Adamowicz, 2003, p. 173). Conjoint analysis has found many commercial applications and in particular for predicting market share for new products.

Validity assessment of nonmarket valuation addresses that which a researcher who designs and executes valuation studies needs to do to enhance the validity of the results (Bishop, 2003). The three Cs of validation - content, construct and criterion - are applied at three levels: at the Methods level, which are specific procedures used in valuation, such as mail surveys; at the Approach level, which is a set of methods characterised by a common theme, such as CV or CM; and at the Application level, which is a set of methods to obtain one or more value estimates. The three Cs apply to all three levels, but in different ways.

While CM has methodological advantages over CV it is by no means without criticisms and deficiencies. Properly constructed choice experiments can minimise these, but it is important to recognise they exist and to scrutinise such experiments to ascertain how the researcher has constructed the experiment. Summarised below are the main areas of concern where deficiencies and limitations can reduce the validity of inferences from CM studies, taken from Bennett and Blamey (2001b).

CM has advantages over CV in regards to establishing an appropriate frame of questioning because the cost question is only one of a number of considerations requiring trade-offs among the other attributes. But, there is still the opportunity for the analyst to bias the results, even inadvertently, through inappropriate framing. A choice set will contain only a few of the potential options. Use of focus groups and pre-testing should ensure that the attributes and levels chosen are appropriate for most people. Informing respondents of the range and relative importance of substitutes and complements at the start of the questionnaire will help embed the appropriate frame. This will help ensure that different qualities and quantities of an environmental service result in different economic values. Also, reminding respondents of their other financial commitments and constraints on their budget will help make ensure answers are embedded in reality. Failure to deal with these issues will lead to values that are too high.

Because CM incorporates a number of repetitive questions with variants of the same theme there is the possibility respondents will become fatigued and either opt out or adopt simplified decision strategies, or heuristics. Analysts face the dilemma of having too many attributes and/or repetitions of choice sets or too few. Restricting the number of environmental attributes to three or four as well as the cost attribute seems to be within the range of most people to comprehend. Also eight to 16 repetitions of choice sets does not seem to cause fatigue for most people. If the experimental design requires more of either then respondents can be offered a randomly selected sub-set of the choice sets.

More than two alternatives in a choice set offer respondents additional freedom in strategic behaviour. For example, this may result in some respondents choosing an alternative that they think will be a "winner" even though this is not their preferred choice. This could result in the choice of an alternative even though the cost is more than they would be willing to pay from their own budget. Such strategic bias could wrongly influence public policy.

There is a possibility that the many contingencies on which the questionnaires are based mean the hypothetical scenarios underlying the choice sets are over simplified. This may cause respondent confusion and hence misrepresentation of economic value.

The underlying econometrics of choice modelling is becoming increasingly complex with the development of models that are better able to model the real world. For example, the random parameters logit model allows for heterogeneity of responses between individuals, which was not a feature of earlier models. This complexity requires significantly higher skill levels than for CV and earlier CM models. There are risks therefore that analysts are not sufficiently aware of the complexities and thus apply the models inappropriately.

Finally, there is the issue of cost. To get the most out of a choice experiment requires rigor in setting up the experiment involving the use of focus groups, pre-testing and iterative sampling. Policy budgets are often constrained to the point that the cost of a primary choice modelling survey is prohibitive.

In weighing up the strengths and weaknesses Bennett and Blamey (2001) conclude that CM is no "magic bullet" (p. 241), but that it has characteristics that make it appealing compared with CV. Further research is required to reduce the technical hurdles and make it more accessible.

3.4.9 Summary of non-market valuation tools

There is a wide range of valuation tools that have potential for valuing indigenous biodiversity. Ecological economists strive for a more holistic approach to valuation, emphasising social, environmental and cultural values together with the neoclassical economic values. While there are some examples of this approach in practice there is still a large gap between the theory and practical application that could be integrated into a DSS used routinely by MAFBNZ staff to optimise biosecurity decisions.

Natural resource economists, often with agricultural backgrounds, working from the neoclassical cost benefit analysis base applied to productive sectors have broadened their world view and attempted to incorporate public good benefits and costs in the analysis of public policy issues. Perspectives from psychology and welfare economics have been combined with market good valuation to reveal preferences as a basis for indirect valuation of environmental values. This has been extended to stated preferences where passive use values have been derived for existence and bequest values. A number of tools have been developed that, while being controversial, have been widely used and accepted internationally by government and the courts. Included in these are multi criteria analysis (and its refinement the analytical hierarchy approach), contingent valuation, choice modelling and benefit transfer.

A challenge of the current research is to identify a set of valuation tools that meet the needs of MAFBNZ and demonstrate their application for routine use. This will inevitably entail some simplification of more sophisticated tools, while still ensuring that the approach adopted is soundly grounded in theory. There may well be a trade-off between rigor and transparency. The latter will take precedence when greater public involvement and acceptance is a prime consideration. Also there are the practical considerations imposed by limitations on time and money.

This discussion leads to benefit transfer, which in its simplest form is merely taking the results of previous studies and applying them to new issues. Use of benefit transfer has grown as a rapid, low-budget approach to providing non-market values, features often required in policy making situations.

3.5 Benefit transfer

Wilson and Hoehn (2006) review the state of the art for benefit transfer. A more recent analysis focuses on CM studies as the source of transfer values (Rolfe and Bennett, 2006). It provides an excellent coverage of the issues involved in using non-market valuation techniques for the transfer of environmental values.

3.5.1 Development

Rosenberger and Loomis (2003) describe the history and development of benefit transfer to inform policy and the decision making process at various stages, including whether original research is warranted.

The first recorded use of benefit transfer was in 1973 when the U.S. Water Resources Council published unit day estimates for recreation activities for use in evaluating water-related projects. These were derived from a combination of past empirical evidence, expert judgement and political screening. Subsequently other U.S. agencies developed similar estimates of values for recreation, timber, forage, minerals and water. In the early 1980s Freeman (1984) began the formal process of reviewing benefit transfer studies and a special section of Water Research in 1992 brought together many of leading resource economists to critique the studies done to date.

To that time, most studies of benefit transfer used point estimates, measures of central tendency or administratively approved estimates (value transfer). However, Loomis (1992) proposed incorporating more information through transferring entire (demand, benefit or WTP) functions and Smith and Kaoru (1990) and Walsh, Johnson and McKean (1992) used meta-regression (collectively these are referred to as function transfers). Since the early 1990s there have been many more studies and the current trend is "to build models that are more sensitive to underlying nuances of data collected, either from multiple sites or a single study" (Rosenberger and Loomis, 2003, p. 447).

Demand or benefit function transfer effectively uses regression coefficients from a research site to plug into a transfer function utilising summary statistics from the policy site. Meta-regression analysis summarises and synthesises outcomes from several studies either through pooling the actual data from multiple studies or using summary statistics, such as value estimates, from multiple studies.

There have been a number of studies across different geographical and political domains. Of particular interest to this project is a study by Woodward and Wui (2001) who used the technique for valuing wetland. The authors used meta-regression analysis to evaluate the relative value of different wetland services using the results from 39 separate studies with 65 observations of value. They concluded that while some general trends were beginning to emerge, the prediction of a wetland's value based on previous studies remained highly uncertain and the need for site-specific valuation efforts remained large.

Empirical evidence shows that function transfers are more reliable than value transfers, although the errors on the former can still be very large. Rosenberger and Phipps (2001) reduced the error in a meta-regression transfer function from 140% to 20% by modelling site characteristics, such as physical and sample population attributes.

3.5.2 Good practice

In categorising ideal transfer conditions, Boyle and Bergstrom (1992) state that source and target sites should be identical. However, a more realistic criteria is that source and transfer sites should be similar across key aspects (Rosenberger and Stanley, 2006). Similarly Morrison and Bergland (2006, p. 426) also found that "when sites and populations are similar, value estimates at policy and study sites have shown to be statistically equivalent. However, as the policy and study sites become more different, or the populations sampled become more different, the value estimates generally diverge...they conclude that these results are positive and offer an indication of construct validity."

As most studies have been conducted overseas an important issue is whether benefit transfer is valid between countries. Shrestha and Loomis (2001) provide evidence that it can be done, at least for recreation activities.

Brouwer (2000) proposed a protocol of good practice for benefit transfer and while it was compiled for CV it applies equally for CM. First, define the environmental goods and services. Next, identify the stakeholders and the values held by different stakeholder groups. Involve stakeholders in determining the validity of monetary valuation. Carefully select study sites. Account for methodological value elicitation effects (most important design effects are: payment mode, elicitation format, the level of information, and sensitivity to scope and/or embedding effects). Finally, involve stakeholders in value aggregation.

For direct transfer the following steps have been proposed by Rosenberger and Loomis (2003). First, define the policy context, including the characteristics of the policy site, the information needed and the units required. Then gather original research outcomes. Screen the original research for relevance, fit to the policy context, type of units, quality of the original research. Select the best estimates, transfer the estimates. Finally, aggregate to provide the total value estimate at the policy site.

3.5.3 Pitfalls and adjustments

Rolfe and Bennett (2006) bring together a number of CM studies to demonstrate the process, pitfalls and adjustments required for accurate benefit transfer. Their book starts with a short history of how CM became to be the focus of BT studies. In their view deficiencies identified in CV studies as a result of the litigation process for the Exxon Valdez ecological disaster in 1989 reduced the level of confidence of policy makers in this form of stated preference technique. As a result analysts then turned to CM as an environmental valuation tool (Bennett, 2006). And while CM has also been challenged on its capacity to produce unbiased estimates of value, choice questionnaires have been shown to be more incentive compatible than CV when designed to provide realistic frames of reference for respondents (Bennett and Blamey, 2001b).

But bias was not the only issue with state preference techniques. Primary surveys, which are required for both CV and CM are costly and hence the much cheaper desk-top process of benefit transfer became attractive. However, the validity of BT depends on five conditions being met (Bennett, 2006): 1) the biophysical condition must be similar between the source and target areas; 2) the scale of environmental change must be similar; 3) the socio-economic characteristics of the source population need to be similar to the target population; 4) the frame or setting of the valuation must be similar; and 5) the source study has to be technically sound.

Meeting all five conditions is not easy and as a result guidelines have been developed for the use of specific adjustment factors to reflect differences between the source and target (van Bueren and Bennett, 2006). For example, when transferring values from the national context to the regional context scaling factors need to be applied to reflect the higher values held regionally. Another factor, pointed out by Loomis (2006) and Whitten and Bennett (2006) is that diminishing marginal benefits need to be taken into account over different overall levels of supply.

In an application specifically designed as a source study Morrison and Bennett (2006) demonstrated that a strong and detailed database of source studies was needed to deliver accurate assessments for benefit transfer. This was an insight reinforced by Hanley, Wright and Alvarez-Farizo (2006) who found that source studies of riverine ecology values in the United Kingdom were inadequate as a database for benefit transfer.

In a sequence of experiments to detect differences between environmental and social attributes for different Queensland river sub-catchments and populations of different distances from the source sites Rolfe, Loch and Bennett (2006) found that the BT process was difficult and sometimes contradictory, but CM held promise in some circumstances. Kerr and Sharp (2006) also conceded that potentially large errors could occur from the simple transfer of value estimates for Auckland urban streams. A similar conclusion was reached by Whitten and Bennett (2006) when attempting to transfer environmental values of wetlands in South Australia and New South Wales. Systematic differences due, for example, to self selection by respondents with particular strong views resulting in potential bias need to be specifically adjusted.

Including social and cultural values as attributes helps to reduce bias in the environmental values by highlighting the trade-offs often implied when valuing the impacts of environmental policy change. Recent examples include: the number of farmers leaving a region as a result of wetlands protection measures (Whitten and Bennett, 2006); the number of people leaving a region as a result of water resource allocations (Rolfe *et al.*, 2006); differences between the cultural heritage values held by different sub-groups in a population (Rolfe and Windle, 2006); restrictions on children's ability to enjoy the sea shore (Bell, Menzies, Yap, and Kerr, 2008); wasp stings (Kerr and Sharp, 2008); and job losses in the dairy industry (Marsh, 2010). The ability of CM to incorporate multiple attributes and highlight both the benefits and costs of environmental policy change increases the richness of data available for benefit transfer.

Benefit transfer is still an evolving discipline, and when based on choice modelling, is still relatively new; hence there is a need to recognise its limitations and the uncertainty inherent in the information. The burgeoning literature includes examples of very high transfer errors (Brouwer, 2000). Bateman, Jones, Nishikawa, & Brouwer (2000, p. 4) state that "when considering the options of rejecting benefit transfer, transferring values and accepting some bias or systematically adjusting the value the latter is clearly preferable from a policy perspective". Bateman (2009) showed that theory driven transfers (i.e. including variables such as 'income') outperform both univariate and best-fit function transfers.

Once values have been estimated, consideration should be made as to whether they have answered the policy question with the desired level of accuracy. If the answer is yes then no further work is necessary, but if not a specific study may be needed.

3.5.4 Conclusions on benefit transfer

Conclusions drawn from a review of the literature are that there exists a key role for CM and benefit transfer to contribute to more informed decisions in a policy environment characterised by uncertainty. Primary studies will always be better than benefit transfer studies, but there is a role for the latter when time and budget are constrained. While uncertainties abound in non-market valuation, the information generated is only one component of the decision-making process, sitting alongside a range of other uncertain information (scientific, market, etc). In addition to providing quantitative values, non-market valuation can help by clarifying trade-offs, identifying magnitudes of directions and effects, and providing new insights (e.g. identifying new stakeholders). So long as state-of-the art methods are used and any limitations are clearly identified and communicated, the decision-making process will be improved.

Having reviewed the economic tools for biodiversity valuation the following section examines frameworks for decision making.

3.6 Decision Making

3.6.1 Decision Support Systems

House (1983) argues that the capabilities of Decision Support Systems (DSS) are difficult to define and generalise. In 1983 there was no accepted definition of what constituted a DSS; however he characterised most systems as being flexible, dealing with structural problems and at least partially interactive (p.3). Decision support systems (including a data base, a model base, and the decision maker) focus on supporting decision making rather than systems of information flows and reports. "Observed characteristics of typical systems include orientation towards less structured problems, combined with data access, retrieval functions, model use, ease of use by non-technical users, and adaptability to environmental changes and varying decision styles" (p.9). They tend to be oriented to top management rather than at the lower levels of management.

Sprague (1983) outlines a sensible approach to develop a DSS. It needs to be built with short, rapid feedback from users to ensure that development is proceeding correctly. It must be developed to permit change quickly and easily. The typical steps are analysis, design, construction and implementation. These are often combined into a single step which is iteratively repeated until a relatively stable system evolves. An example of a DSS to manage fisheries externalities in New Zealand's Exclusive Economic Zone is described by Hughey, Cullen, Memon, Kerr and Wyatt (2000).

Using this approach a broad definition of a DSS that is relevant to biosecurity decision making is: an interactive computer-based system that helps people use computers, communications, data, documents, knowledge, and models to solve problems and make decisions (Power, 2002). In order to add value to decision making the system needs to be used and become a significant strength or capability of the organisation with the advantage sustainable over a realistic period. Power argues that most managers only require summaries of transactions, and typically prefer charts and graphs to tables. They want the right information, at the right time, in the right format, and at the right cost. As these requirements are likely to be sought by BNZ they pose significant challenges to developing the system.

MAFBNZ needs support for strategic decision-making covering the allocation of resources. This involves solving semi-structured problems that have routine elements in terms of the process of analysis but also are uncertain as to their occurrence and outcome. Given these characteristics it is expected that a specialist analyst will interface between the managers making the decisions and the information going into the system. The information coming out of the DSS may be used at the operational, managerial or policy level depending on the scale and potential impact of the incursion.

Any DSS development project requires a mix of complementary skills and it is unusual to find all these in one person, therefore a team approach is needed. The key roles include the project manager, an executive sponsor or project champion, potential DSS user(s), a DSS analyst, technical support staff, and a DSS toolsmith (Power, p. 68).

Given widespread computer literacy, decision support systems have evolved from primarily personal support tools to becoming a shared commodity across an organisation through intranets and the internet (Turban, Aronson, and Liang, 2005). Today's DSS tools utilise the web to view and process data and models. Advances in software and hardware allow easy access to important information and tools. Decision support for groups continues to improve and artificial intelligence methods are improving the quality of decision support and becoming embedded in many applications freeing up time.

Any DSS for BNZ will have to address the issues of communication with managers and other stakeholders. Indeed, if the core elements of the deliberative approach (see below) are to be utilized, then at least parts of the DSS will need to be available via the Internet and/or intranet.

More recently, researchers have attempted to move beyond DSS to include interaction between the researcher and the target community. The next two sections explore two such approaches, some aspects of which may have application to MAFBNZ's problem.

3.6.2 Deliberation Support Tools

O'Connor (2005) explores prospects for the exploitation of new information and communication technology (ICT) for representing and aiding the resolution of collective problems of governance of common environmental and natural resources. The emphasis is on the process of discovery, learning and multi stakeholder deliberation that can contribute, directly and indirectly, to good sustainability decisions. The tools developed include multi-media Deliberation Support Tools (MM-DST or DST) and tools for informing discussions, debates and deliberation (TIDDD).

O'Connor seeks to use DST, in the context of policy and programme evaluation, to replace the traditional DSS concept. In his view it is a richer concept employing argument and dialoguing to inform decisions and make visible contradictions rather than burying them or setting up conflicts among protagonists. O'Connor is interested in addressing questions about society's future and problems in situations involving scientific knowledge, value systems and consequences of environmental change.

These situations characterise what Rittel and Webber (1973) termed "wicked problems" where it is difficult to formulate and justify simple rules of action. Simple criteria such as the maximisation of net benefits (with monetary cost benefit analysis) or avoidance of risks (such as the precautionary principle) are said to fall down because either they do not adequately address the decision issues, or they are not plausible or acceptable to key stakeholders, i.e. there is no clear bridge between knowledge and right action. O'Connor sees this as a choice between dictatorship or inconsistency. This means that reasoning must be employed in a complex deliberative way and new tools developed to assist decision making. Starting from a situation of conflict, dissent, misunderstanding or antagonism, some reconciliation might be possible through processes of dialogue and deliberation.

Conflicts will arise, particularly when there is a budget limitation. This is pertinent to the allocation of resources to protect or sustain indigenous biodiversity as opposed to revenue generating agricultural, forest or marine assets. All the elements of a wicked problem exist.

The concept of sustainability raises challenges in scientific, economic, moral and political areas that require reconciliation between various interests that are in conflict with each other (O'Connor, 2005). These conflicts are between local and national interest (not in my backyard - NIMBY); present and future generations; self-interest and interest in the lives of others; the human and the non-human world; our culture and other cultures; what is 'internalised' in the marketplace and what remains an 'externality'; and any given region or territory and the rest of the world.

One of the deliberation support tools developed by O'Connor and his team was VIRTUALIS an acronym for a multi partner project called Social Learning on Environmental Issues with Interactive Information and Communication Technologies (http://www.virtualis-eu.com). It brought together a consortium of specialists in information technology, sustainable development, environmental modelling, public policy and governance, learning psychology and open learning, to develop computer-based learning tools on ecosystems and natural resources. The DST combined spatial representation, scenario simulation, multiple criteria analysis and interactive user-friendly computer interfaces.

According to O'Connor, use of the tool creates a platform for the exploration of governance issues, decision options and policy choices. This led to the idea that comparative evaluation of scenarios could be undertaken simultaneously with respect to several different criteria and from several different points of view. In turn this led to the concept of a three dimensional 'deliberation matrix' which is an intuitive framework incorporating a number of scenarios reflecting different technological, economic and governance features; a diversity of stakeholders; and allows for multiple evaluation criteria.

DST makes explicit the structure of the political process that is multi stakeholder, multi actor, multi-criteria deliberation. The framework is supported by technical information including spatial representation of key scenario indicators and an online presentation and classification of indicators.

Further work on DST in Europe, the ALARM Integrated Project addressing biodiversity loss risks, (ALARM, 2010) has developed the concept of a 'virtual nature walk' embodying eight main types of ecosystem: inland waters, wetlands, forests, grasslands and dry scrub, agro-ecosystems, mountains, polar habitats and urban ecosystems. For each ecosystem type there are five types of services: natural resource, waste assimilation, scenery, site of production and consumption, and life-support. With four change vectors for damage to each of these services: chemicals, invasive species, pollinators, climate, and land-use change.

These models offer the opportunity for different stakeholders (representing such groups as business, public administration and civil society) to explore alternative scenarios of policy action. While this sounds intuitively reasonable and possibly helpful in resolving conflict and establishing consensus, the outcomes are still susceptible to being based on the views of the individuals who have built the models. Because these are complex systems unforeseen outcomes are likely to occur. These models are resource intensive to develop and implement and as such are likely to be beyond the scope of MAFBNZ to utilise to aid response decision making.

The next section reviews a modified approach which aims to involve groups of stakeholders in building models to aid decision processes.

3.6.3 Mediated Modelling

Van den Belt (2004) describes mediated modelling as a system dynamics approach to environmental consensus building and provides examples of its applications. In the foreword to his book, Thomas Deitz describes mediated modelling as a flexible and innovative tool for linking science and democratic process.

Mediated modelling has evolved from system dynamics, which is concerned with understanding of how systems change over time, which has led to the development of simulation models. Here, the behaviour of systems is studied through identifying a minimum number of building blocks that can explain the bulk of the behaviour. Feedback loops and time lags characterise the relationships among the building blocks which helps to understand time delays, non linearity and feedbacks. Mediated modelling builds on system dynamics thinking and emphasises the interactive involvement of affected stakeholders in the learning process about the complex system they are in. It also allows policymakers and other stakeholders to see the consequences of their actions over longer timescales. (Van den Belt, 2004, p. 3).

The term "bounded rationality" is used to describe a feature of human decision making that is limited, or bounded, and this limitation can create persistent judgemental biases and systematic errors (Simon, 1948). The human mind works in a rather short term manner and favours linear relationships over dynamic systems perspective (Ehrlich, 2000; Weiner, 1985). Personal positions are often static and defended on the basis of convictions and perceptions, and people select information that reinforces their initial position (Bakken, Gould, and Kim, 1994, p. 4). This thinking and the lack of understanding of system dynamics and the participation of affected stakeholders can cause policy decisions to have unintended, potentially disastrous consequences. By integrating the ecology and economics from the start of the decision making process, mediated modelling results are more multi-dimensional, dynamic and interactive with the goals and trade-offs clearer from the outset.

Supporters state that mediated modelling by broadening participation, while being more costly at the front-end, should make the overall decision making process more effective and less expensive over the whole process. Early involvement of broad stakeholder groups (government, industry, environmental non governmental organisations, etc) seeking common goals and consensus on an issue may increase the shared level of understanding in a community and reduce the conflicts and costs at the implementation phase.

Richardson and Anderson (1995) provide five distinct roles for a support team of a group modelling process; facilitator, modeller/reflector, process coach, recorder, and gatekeeper. The gatekeeper is the champion and without this initiative and promotion for both the human and the technical aspects these projects don't materialise. A recorder is a project team member who is not participating in the modelling directly but who is observing (as objectively as possible) what is happening in the room. The facilitator is an impartial party who manages meetings as part of the collaborative process while a mediator is an impartial party who intervenes in a negotiation. In general a modeller tries to create an abstract, simpler representation of reality. This can vary from drawing pictures or a map of activities with a stake in the sand (rapid rural appraisal), to a flip charts, to a computer equipped with system dynamic software (as is the case was mediated modelling). A group involved in mediated modelling needs to understand that the facilitator's role as one of a process coach rather than a fixer of all problems.

Forgie and Richardson (2005) explore the scope for mediated modelling to be used as a tool by local authorities in helping them to determine and implement community outcomes. They use the Palmerston North City Council as a case study. The Local Government Act 2002 requires local authorities to produce a Long Term Council Community Plan (LTCCP) in order to promote the social, economic, environmental and cultural well-being of communities. This plan covers a 10 year period and focuses on the preferred future and vision of the community (in this context the desired future is referred to in the Act as "community outcomes").

As part of the process of developing the LTCCP local authorities are required to engage the public, different stakeholder groups, central government, and non-governmental agencies in the development of the community outcomes. Given the range of people, interest groups, and government agencies involved, local authorities are being challenged to think of effective and innovative ways to bring diverse and sometimes conflicting goals together when deciding community outcomes. Mediated modelling may be an appropriate method in such circumstances.

In this section mediated modelling has been outlined in order to consider it as a potential approach to improving decision making on pest response. Combining disciplines under the ecological economics banner and adopting a shared vision were also touched on in the search for such a process. While a formal adoption of mediated modelling does not seem to fit the response process it is possible that preparing the community for future pest responses programmes that could involve conflicts, wicked problems and unintended consequences could utilise the approach. This moves into the realm of public relations and is not part of this research.

While intuitively appealing, there are limitations to the implementation of a mediated modelling approach. Most obvious is the intellectual resources and time required.

3.6.4 Summary of decision making

The current thrust of research seems to be aimed at going beyond DSS, where the participants are primarily researchers and bureaucrats, to having significant involvement of the "community" in the decision making process. One avenue is through using sophisticated multi-media deliberation support tools to aid learning and information transfer on complex problems where conflict is a feature. The other major avenue is through mediated modelling which focuses on involving wide groups of stakeholders in developing simulation models to aid decision making right from the beginning of the process. There is also a move towards integrating different disciplines into thinking about resource allocation issues and decision making processes. The integration of ecology and economics into ecological economics is a case in point.

The DSS developed in this study integrates certain elements of all three approaches, recognising the constraints of time and money faced by MAFBNZ and the need for a relatively simple process.

In the next section the economic theory within which the non-market valuation tools and DSS will be used for decisions on biosecurity response involving impacts on biodiversity is outlined.

3.7 Summary of the literature

In this chapter a survey of the literature has concluded that the state of the ecosystem that is relevant for protection should be determined by reference to society's utility function in a way that captures the ever changing dynamics of nature. Broadening the scope of analysis to include non-market benefits and costs will result in more informed decisions. Such estimates should include not only changes to natural capital (indigenous biodiversity), but also social capital (the community and government) in order to promote community well-being.

Within the response component of biosecurity the change in individual utility resulting from a change in biosecurity status can be aggregated by estimating compensating surplus. This is where income is to be given up, also known as willingness to pay (WTP), or equivalent surplus where additional income is required to restore utility - willingness to accept (WTA). The conceptual framework underlying this, which is outlined in this chapter is utilised in Chapter 4 where the tool of choice modelling is used in a case study. The estimates of value are then incorporated into a cost benefit framework based on the Kaldor-Hicks compensation test which only requires that winners should be able to compensate losers and still be ahead for a project to be judged as being of a benefit to society.

As economists the aim is to quantify in money terms as much of Total Economic Value as possible. Valuing changes to indigenous biodiversity, which include non-market passive use values, extends the range of benefits and costs that can be quantified. Stated preference techniques are the only methods available and one of them , choice modelling, is demonstrated in Chapter 4..

MCA cannot be dismissed as a first cut methodology for optimal allocation of biosecurity resources, however, to be useful it implies expending a considerable amount of time setting up and maintaining focus groups of experts and/or representatives of interested stakeholder groups. It is not the preferred choice as a tool that can be integrated into biosecurity decision making regarding response because it provides a relative measure only and one that cannot add monetary values for non-market benefits and costs to the standard market ones.

Revealed preference techniques, such as hedonic pricing, are not useful as there are no market prices for valuing the passive values of biodiversity.

Of the stated preference techniques choice modelling seems to offer the best tool for valuing ecosystems. It has the ability to tease out the important multiple attributes of value and quantify them in monetary terms. Each case needs to be assessed on its merits and there may be situations where contingent valuation will be appropriate. In most cases the additional power of choice modelling will prevail. CM requires a high level of technical skill to achieve good results. Also it is expensive and time consuming. There is the need to elicit the key attributes and the relevant range of each, which requires the use of focus groups and pre-testing a questionnaire. Designing the choice experiment involves attention to many issues any one of which has the potential to bias results if not correctly handled. Also, obtaining representative and statistically significant community estimates of willingness to pay requires the undertaking of expensive and time consuming primary surveys.

Given the expense and time required for a well constructed and implemented choice modelling experiment the benefit transfer technique is likely to be called on to provide non-market value estimates. This approach has an even greater opportunity for inappropriate estimates compared with CM because similarities between the source and target sites and populations are often weak, the frame and scale are likely to be dissimilar and the quality of the source study or studies may be dubious.

The survey of decision support processes identified a trend towards increasingly sophisticated methods involving community participation such as deliberative process and mediated modelling. A decision support tool for MAFBNZ response is likely to obtain elements of these techniques, particularly as to community involvement, but fall short of the use of multi-media discovery and learning tools and systems simulation to aid the decision process.

The next chapter turns to practical application and generation of biodiversity values for MAFBNZ.

Chapter 4 : Freshwater case study

4.1 Introduction

This chapter sets out the research methodology. A case study approach is used to generate robust biodiversity values for the database in a form that can be used in the DSS.

The research undertaken in this thesis forms part of a wider research programme, which involved estimating biodiversity values in four case studies. Two of the case studies were undertaken as part of the thesis (freshwater and marine) and two were undertaken by other researchers (South Island high country and beech forest). In this chapter the freshwater case study methodology and results are reported in detail. Summaries of all the case studies are provided in Chapter 5.

The aim of the freshwater case study was to elicit dollar values of impacts on indigenous biodiversity due to a hypothetical incursion of the exotic weed hydrilla in Lake Rotoroa (also known as Hamilton Lake). It applies a choice experiment to estimate dollar values for five environmental attributes, including three biodiversity values, plus a cost attribute. Values were estimated for four population samples located at varying distances from the lake.

4.2 Freshwater system: Hypothetical weed incursion

Hydrilla was chosen for the case study as it was BNZ's top priority weed when the research started in 2006. Although restricted to three lakes in the Hawkes Bay region, it had the greatest potential for negative impacts on New Zealand's freshwater systems. In conjunction with BNZ, Lake Rotoroa was chosen as the freshwater system under threat as it has a high risk of hydrilla invasion, has a long history of management, has a high profile due to shoreline housing and recreational use and has some indigenous biodiversity similar to other New Zealand lakes (Harrison pers. comm., 2008).

Hydrilla is a submerged freshwater perennial plant that is characterised by prolific growth and tolerance of a wide range of freshwater habitats including clear or murky, still or flowing water, temperature between 0 and 35^{o}C, water depths from a few centimetres to 9 meters, low light intensity to full sun, and a wide range of acidity and nutrient levels.

The threat of hydrilla to the lake ecosystem is far greater than that of the current exotic incursions of oxygen weeds. Hydrilla would likely develop into extensive weed beds at all depths and smother the native charophytes in particular. While eels are likely to be unaffected, the remaining species of native fish and mussels would be severely impacted through a reduction in available space and change to the habitat. It is also likely that the shags would stop frequenting the lake as the areas of clear water reduced. Swans would be attracted and this would help clear water to a depth of around 1m, but their aggressive behaviour particularly towards children has a down side. Boating would be severely hindered.

If hydrilla was ever to become well established, there would be no realistic prospect of elimination without the long term use of grass carp. A small incursion detected early could be controlled with the herbicide endothall, or other methods, such as weed matting, but use of these techniques would depend very much on where the specific incursion was, and how established it had become. As hydrilla would eventually eliminate all native vegetation, especially charophytes and the underlying seed beds, the use of grass carp would be justified to prevent irreversible damage to the lake ecosystem. Arguably, the best management strategy is either to target effort towards investing in preventing the introduction of hydrilla, or eradicating hydrilla before it became established (de Winton, 2005), Clayton 2008a pers comm.; Hofstra 2008 pers. comm.).

4.3 Economic problem

The introduction of hydrilla into Lake Rotoroa would result in very serious impacts on indigenous biodiversity as well as on how people would interact with the lake; thus, the benefits of eradication or control of hydrilla are the negative impacts avoided. These include loss of native species particularly charophytes, fish, mussels and birds. As the clarity and quality of the water progressively became reduced, there would be increasing negative impacts on humans through a reduction in the quality of the experience of visiting the lake for boating, a gross deterioration in the view presented and eventually odour issues.

The ability to eradicate or control an infestation is dependent on prevention and early detection. Depending on the management strategy adopted, different states of the ecosystem are possible. The attributes associated with the different states of the ecosystem become the basis for framing the choices put to survey participants. Through carefully constructed questionnaires which present participants with alternative choices of the attributes of the ecosystem along with a money cost to each household, it is possible to elicit their willingness to pay (WTP) for a particular state of the environment. This forms a proxy for the value of a change to the ecosystem allowing environmental values to be included in the CBA.

4.4 Choice modelling

Choice modelling (CM) is the stated preference tool used to elicit marginal dollar values for the key attributes of the lake. CM has gained most credence in performing non-market valuation of environmental goods and services (Rolfe and Bennett, 2006). CM has emerged from utility theory and belongs to the suite of tools referred to as stated preference techniques as they rely on people stating their preference when faced with a number of choices about changes to key attributes given some cost to them. Different levels of the key attributes (e.g. levels of the lake's native species, particularly charophytes, fish, mussels and birds) along with a money attribute (e.g. cost to the household) describe options on future states of the lake. Respondents are presented with a limited number of options (a choice set typically comprised of a status quo alternative plus two other alternatives) and are asked to indicate their most preferred state from the choice set. This process is repeated a number of times (i.e. answering a number of choice sets) to go through a relevant subset of the range of options. Statistical experimental design allows the selection of a relevant subset of options that provides the best information to infer values from the choices of respondents.

The hypothetical question is the willingness to pay for maintaining or limiting deterioration of key environmental aspects of Lake Rotoroa due to the weed hydrilla (*Hydrilla verticillata)* with the focus on impacts on indigenous biodiversity.

"The random utility approach underlying the CM technique provides the theoretical basis for integrating choice behaviour with economic valuation. In a Random Utility Model (RUM) the probability of an individual choosing a good is assumed to be dependent on the utility of the that good relative to the utility of alternative goods" (Rolfe, 2006, p. 38). That is, an individual will choose an alternative only if the utility is greater than the utility of the alternative.

An analyst can only observe the systematic (explainable, *V*) component of utility plus an error (unexplainable, ε) component. This is shown in equation 4.1 as the utility of alternative *j* for respondent *n* in choice task *t* as:

$$U_{jnt} = V(\beta_{kn}X) + \varepsilon_{jnt} \qquad (4.1)$$

where β_{kn} denotes the vector of random taste intensities (utility coefficients for attribute *k* across respondents *n*), associated with a vector of attributes *X,* which is the explainable component of utility and ε_{jnt} is the the Gumbel distributed (unexplainable) error component.

Equation 4.2 takes this generic function and applies it to the hydrilla case study. The *k* attributes are described in Table 4.1.

$$\begin{aligned} U_{jnt} &= \beta_{HYD}HYD_{jnt} + \beta_{WQ1}WQ1_{jnt} + \beta_{WQ2}WQ2_{jnt} \\ &+\beta_{WQ3}WQ3_{jnt}\beta_{CHA}CHA_{jnt} + \beta_{BIR}BIR_{jnt} + \beta_{FISMUS}FISMUS_{jnt} \\ &+\beta_{PRICE}PRICE_{jnt} + (1-SQ)\eta n + \varepsilon_{jnt} \end{aligned} \qquad (4.2)$$

where η_n is a random normal error component with zero mean associated with the policy scenarios (the non-status quo alternatives) and SQ is the status quo.

Table 4-1 Attribute descriptions

HYD	Percentage of success in preventing hydrilla cover (0%, 35%, 70% and 100% success levels)
CHA	Percentage of success in preserving charophytes cover (0%, 7%, 14% and 21% success levels)
BIR	Number of shags species visiting the lake (0,1, 2 and 4 species)
FISHMUS	Number of fish species and mussels retained (0, 1, 2 and 3 species)
WQ1, WQ2, WQ3	Effects coding for 4 levels of water quality (significant, moderate or slight deterioration, or same condition from current quality and clarity of water)
PRICE	The money attribute was set at 6 levels: \$0, \$10, \$20, \$40, \$80, \$160 and presented as the cost to the respondent's household each year for the next 5 years.

Given β_{kn} and η_n the probability of observing alternative i to be selected from the J alternative in the choice task is a logit and the sequence of t choices made by a respondent is a joint logit or:

$$\Pr(i_1, i_2, i_3, \ldots, i_t \mid \beta_n, \eta_n) = \prod_t \Pr(i_t \mid \beta_n, \eta_n) = \prod_t \frac{\exp(\beta_n' x_{jnt} + \eta_n)}{\sum_{j=1}^{J} \exp(\beta_n' x_{jnt} + \eta_n)} \quad (4.3)$$

To obtain the unconditional probability, the random components need to be integrated out over their respective ranges:

$$\Pr(i_1, i_2, i_3,\ldots, i_t) = \int_{\beta_n}\int_{\eta_n}\prod_t \frac{\exp(\beta_n' x_{jnt} + \eta_n)}{\sum_{j=1}^{J}\exp(\beta_n' x_{jnt} + \eta_n)} f(\beta_n, \eta_n \mid \boldsymbol{\mu}, \boldsymbol{\Omega}) d\beta_n d\eta_n \quad (4.4)$$

The assumed distributions are normal with mean vector $\boldsymbol{\mu}$ and variance covariance $\boldsymbol{\Omega}$, only the mean of η_n is restricted to zero. In the maximum simulated likelihood estimation these integrals were approximated by weighted probability averages based on quasi-random draws from prime numbers, known as Halton draws (Train, 2003). Halton draws have good coverage properties thus allowing a reduction in the number of necessary draws to achieve high precision.

Equation 4.4 provides the basis for the estimation of the random parameter or mixed logit model where respondent characteristics are included as a random parameter in a distribution rather than as a fixed variable as in the simpler multinomial logit model. This is a significant advance towards reality as it allows for respondent heterogeneity rather than assuming all respondents will respond in the same way.

Using this model the marginal value of a change for a single attribute is the ratio of the negative of the coefficient of the environmental attribute divided by the money attribute as per equation 4.5. This ratio is termed a part-worth (PW) and is the marginal rate of substitution between the income change and the environmental attribute. It is one of the key outputs of a choice modelling experiment and is used to estimate compensating surplus which is the estimate of welfare change as a result of the change in the environmental values (see equation 4.7).

$$PW = -1x\beta_{attribute} / \beta_{money} \tag{4.5}$$

4.5 Design

Having defined the economic problem and hypothetical question, the first step in the survey design is to determine the important attributes of Rotoroa Lake and their levels. This was done using focus groups arranged by a professional market research agency. Groups were convened in Wellington and Hamilton in April 2008. Participants did not know the purpose of the study until they arrived at the meeting. Prior to this the focus group presentation was tested with a group from Biosecurity New Zealand to ensure the technical aspects were accurate.

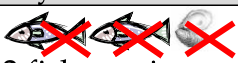
The first part of the focus group session was a presentation to introduce the concepts of freshwater biodiversity, the threats to lake biodiversity and biodiversity protection and control measures. Next, the case study lake was introduced and its features described using slides to depict the various attributes of the lake including natural and man-made aspects. Then participants were asked to make a choice between two different states of the lake with variable attribute specificities. The object was to determine which features of the lake people valued most highly. Aspects of the lake that were tested included water with and without surface plants, board walk versus natural lake edge, ducks versus pukeko (exotic vs. native), oxygen weed versus charophytes (exotic vs. native), a scene with boats on the lake versus birds on the lake, and a scene of the lake side with introduced trees versus native trees.

The next stage introduced hydrilla, the potential invasive weed, its characteristics and likely impacts. Participants were asked to indicate how acceptable different states of the environment would be to them. Water quality and clarity, presence of hydrilla, presence of native water plants (charophytes), presence of native fish and mussels, native birds, water sports and lake side recreation were tested. Finally, participants were asked to consider various increases in their annual household rates bill for different control mechanisms resulting in different outcomes taking into account other demands on the home budget.

On the basis of the information collected from BNZ and the two focus group meetings, the key attributes and their levels were selected for the choice experiment. This was tested on a convenience sample of 12 people in June 2008 drawn from colleagues and friends. The results were analysed and used as the priors to assist in the experimental design of the survey (discussed later in this section).

Figure 4.1 shows an example choice set. The rows represent the attributes; for example, water quality and clarity, coverage of native submerged plants etc. and the columns represent the options or scenarios, which are described by a set of attribute levels including the cost to the participant's household.

Figure 4-1 Example of a Choice Set

Question 1: Options A, B and C Please choose the option you prefer By ticking ONE box			
Attributes	Option A (status quo)	Option B (Alt1)	Option C (Alt2)
Extent of hydrilla	 100% coverage	 30% coverage	No hydrilla
Water quality and clarity	Significant deterioration	OK Same as now	OK Same as now
Coverage of native submerged plants	Eliminated from lake	Eliminated from lake	 Same as now at 21% cover
Number of native bird species	 All 4 shag species do not visit the lake anymore	 3 shag species do not visit the lake anymore	 3 shag species do not visit the lake anymore
Fish and mussels	 2 fish species and mussels disappear from the lake	 Mussels disappear from the lake	 1 species of fish and mussels disappear from the lake
Cost to your household each year for 5 years	$0	$20	$160
I would choose ☑	☐ A	☐ B	☐ C

The money attribute was “the cost to your household each year for 5 years.” The payment vehicle was a household rate levied to fund hydrilla control, as provided for under the Biosecurity Act (1993).

Money values were chosen to cover the range of payments likely to be acceptable based on the focus group results. The selected values were $0, $10, $20, $40, $80 or $160.

The status quo is the "do nothing option" with a payment of zero dollars and with each environmental attribute at its worst level. The status quo is presented as Option A in all choice situations. Two alternatives to the status quo (Alt1 and Alt2) are presented as Option B and C, respectively, in the survey questionnaire.

Efficient design of surveys results in reliable parameter estimates characterised by small standard errors. The experimental design is Bayesian in nature using the normal distribution for the coefficients of all environmental attributes and the money attribute. In essence Bayesian designs allow the analyst to utilise existing knowledge about the attributes to assist in designing an experiment that results in a good explanation of the model with a small sample size.

As discussed in Ferrini and Scarpa (2007), a Bayesian efficient design is less sensitive to misspecifications of the priors than a point efficient design. This is because Bayesian designs recognise the uncertainty in existing knowledge where as a point estimate doesn't. So by choosing a point estimate, if it is wrong, the design will not achieve a good model fit or the sample will be larger than need be, thus adding to sampling cost.

The MNL estimates of the parameters from the convenience sample (see Table 4.2) were used as priors when they were significant at 95% confidence level (otherwise a theoretical prior was used for the experimental design). The estimates were assumed to be normally distributed with standard deviation equal to the estimated standard errors.

The criterion to be minimized for the efficient design was the sum of the variances of the marginal WTP of each attribute, as suggested in Scarpa and Rose (2008). As a result, the design is specific to WTP estimation (C-efficiency), rather than to estimation of parameter estimates (D-efficiency). Scarpa and Rose (2008) review these efficiency criteria and conclude that the right criteria are case dependent and need to be selected in light of the objectives of each study.

The recent release of Ngene (ChoiceMetrics, 2009), an experimental design software for stated choice experiments, allowed the evaluation of the survey design for efficiency. The evaluation using Ngene showed that the design is efficient with an S estimate 4.156, where S is the minimum theoretical sample size and a D-error of 0.022, where a small D indicates design efficiency (Hensher, Rose, and Greene, 2005).

While the S estimate implies that the minimum sample size required is 5 respondents for the most difficult attribute to estimate (highest standard error), bias errors necessitate higher sample sizes. Bias arises from random choice behaviour and the assumption that all random components are independent (the IID assumption in MNL). However, the low S estimate achieved indicates an efficient design.

Table 4-2 MNL estimate convenience survey

Variable	Coefficient	Standard Error	P[|Z|>z]
HYDR1	0.8814*	0.5047	0.0807
HYDR2	1.1512**	0.5371	0.0321
HYDR3	2.1230***	0.5621	0.0002
WQUAL1	0.7167	0.5082	0.1584
WQUAL2	0.5628	0.5283	0.2867
WQUAL3	0.2473	0.4903	0.6140
CHAR1	1.3297**	0.5441	0.0145
CHAR2	2.3927***	0.6032	0.0001
CHAR3	3.1035***	0.5812	0.0000
BIRDS1	-0.1871	0.5544	0.7358
BIRDS2	0 .2586	0.4947	0.6011
BIRDS3	1.5754***	0.5149	0.0022
FISH1	0.3807	0.5470	0.4864
FISH2	1.3063**	0.5114	0.0106
FISH3	1.7579***	0.4870	0.0003
PRICE	-0.0206***	.0044	0.0000
LL		-64.545	
Pseudo-R^2		0.382	
AIC (Akaike information criterion)		1.134	
BIC (Bayesian information criterion)		1.467	

*** Significant at 99% confidence level
** Significant at 95% confidence level
* Significant at 90% confidence level

The optimal design comprised 60 choice sets. These were randomly divided into five groups resulting in a manageable grouping of 12 choice sets per respondent. The five groups of choice sets were uniformly distributed in each survey sample resulting in each group of choice situations being (more or less) uniformly represented. Appendix 1.1 sets out the complete experimental design, while Appendix 1.2 presents the master coding table of levels for the environmental and cost attributes.

4.6 Data collection

Typical methods for data collection include mail-out surveys, telephone surveys, internet surveys and personal paper or computeraided design interviews. Telephone surveys involve considerable cognitive burden as each questionnaire typically involves 8-16 choice sets with three options across five to six attributes per choice set. Impersonal mail-out surveys are unable to convey richness of information to a similar level achieved in a personal interview (Kerr and Sharp, 2003). Personal interviews ensure respondent understanding of the survey and allow the use of visual aids to convey information, but it is the most expensive form of data collection particularly in multiple locations.

This study used a hybrid community meeting approach to obtain the benefits of face to face briefings while saving on costs and time. It has the advantage of bringing the assembled group of respondents to a uniform level of understanding of the issue and administering choice questionnaires to multiple respondents in one sitting. Community service groups (e.g. primary school, dragon boating association, Lions, Rotary) were contacted to organise the meetings using a promotional flyer (Appendix 1.3), a $50 donation to the service group per person recruited and $20 petrol voucher to the participant. The community service groups were requested to obtain 50-60 participants making up a cross-section of adults in the community with a gender balance, and a range of ages, educational qualifications, incomes and ethnicity.

The meeting started with a 40 minute powerpoint presentation (Appendix 1.4) to introduce the survey (see Appendix 1.5 for speech notes). Topics covered included: freshwater biodiversity, biodiversity protection, the case study lake, the hypothetical hydrilla incursion, the range of impacts that hydrilla could have on the ecosystem and how to answer the choice questions. Participants were asked to consider the cost to their household in relation to other budgetary demands. They were informed that the personal information they provided would be treated in confidence and was required to check for representativeness of the sample against census information. Along with the usual questions on gender, age, education, profession, income and ethnicity they were asked to indicate if they were a member of a conservation organisation. The presentation was followed by distribution of the survey form, which took 20-30 minutes to answer the 12 choice questions and provide personal information. Each questionnaire was checked for completeness before handing over the thank you voucher. The meeting ended with a light supper.

The survey samples were drawn from four locations with varying proximity to Lake Rotoroa. The four samples are Rotoroa (local, sample beside or near the lake), Rototuna (within district, sample in Hamilton - same city as the lake), Morrinsville (within region, sample in Waikato – same region as the lake) and Karori (out of region, sample in Wellington – a distant urban location). The four locations were chosen to observe the effect of distance on the WTP for any of the attributes.

4.7 Modelling and results

The survey gathered a total of 225 respondents but twelve under-age participants in the Rotoroa sample (under 18 years old) were excluded as they would be unlikely to be a party to household budget decisions. This resulted in a total of 213 respondents distributed among Rotoroa (44), Rototuna (40), Morrinsville (65) and Karori (64). Overall, the analysis consisted of 2,556 observations.

The community meeting approach does not purport to generate a statistically representative sample of each community. However, it does provide a cross-section of informed opinion from the community such as that would exist following a community awareness campaign and debate about management options for a hydrilla incursion (Kerr and Sharp, 2007).

The sample respondents were generally representative of the relevant population (refer to Table 4.3 below) for a number of aspects (e.g. gender in Rototuna and Karori; young and mid-age in Morrinsville and Karori; low income in Rotoroa and high income in Rototuna, European/Asian ethnicity and high/low skills in Rototuna). In terms of gender, male was over-represented in Morrinsville. Polytech and degree qualifications were generally over-represented in all samples. The old and young age groups were generally under-represented except in Karori (where old was over-represented). Except in Rototuna, the European ethnicity was over-represented. The Maori and Pacific ethnicities were over-represented in Rotoroa and Rototuna but under-represented in others. Asian (except in Rototuna) and other ethnicity were generally under-represented. The high income group and high-skill occupation group were generally over-represented except in Rototuna.

Membership of a conservation group resulted in positive responses for Rotoroa (23 %), Rototuna (8 %), Morrinsville (14 %) and Karori (16 %). These results compare with the national average of 8% (DOC 2008).

Table 4-3 Survey demographics

	Sample				Population Census				Lower Limit				Upper Limit			
	Rotoroa	Rototuna	Morrinsville	Karori	Rotoroa	Rototuna	Morrinsville	Karori	Rotoroa	Rototuna	Morrinsville	Karori	Rotoroa	Rototuna	Morrinsville	Karori
GENDER																
Male	40.9%	42.5%	66.2%	51.6%	48.3%	48.5%	49.1%	47.7%	41.2%	41.0%	43.2%	41.8%	55.4%	56.0%	55.1%	53.5%
Female	59.1%	57.5%	33.8%	48.4%	51.7%	51.4%	50.9%	52.3%	44.1%	43.5%	44.7%	46.0%	59.3%	59.3%	57.0%	58.7%
QUALIFICATION																
No Qual	0.0%	0.0%	4.6%	1.6%	14.7%	16.3%	31.2%	7.8%	12.5%	13.8%	27.4%	6.9%	16.9%	18.8%	35.0%	8.8%
Fifth	9.1%	10.3%	4.6%	1.6%	9.5%	12.8%	16.5%	7.1%	8.1%	10.8%	14.5%	6.3%	10.8%	14.8%	18.5%	8.0%
Sixth	20.5%	12.8%	6.2%	1.6%	22.8%	24.3%	18.4%	25.2%	19.5%	20.6%	16.2%	22.1%	26.2%	28.0%	20.6%	28.3%
Polytech	38.6%	33.3%	56.9%	34.4%	19.3%	21.5%	17.1%	15.1%	16.5%	18.2%	15.0%	13.3%	22.1%	24.8%	19.1%	17.0%
Degree	31.8%	43.6%	27.7%	60.9%	24.8%	19.5%	6.4%	40.4%	21.2%	16.5%	5.6%	35.5%	28.5%	22.5%	7.1%	45.3%
AGE																
Young	22.7%	11.4%	35.4%	17.2%	35.2%	19.3%	21.8%	18.2%	30.0%	16.3%	19.1%	16.0%	40.4%	22.2%	24.4%	20.4%
Mid-age	77.3%	75.0%	46.2%	57.8%	47.9%	58.5%	51.8%	62.5%	40.9%	49.5%	45.5%	54.9%	55.0%	67.5%	58.0%	70.2%
Old	0.0%	2.3%	18.5%	25.0%	16.9%	22.3%	26.5%	19.3%	14.4%	18.9%	23.3%	16.9%	19.3%	25.8%	29.7%	21.6%
INCOME																
High income	31.8%	35.0%	43.1%	57.8%	22.1%	32.6%	13.7%	37.0%	18.9%	27.6%	12.1%	32.5%	25.3%	37.5%	15.4%	41.5%
Low income	68.2%	65.0%	56.9%	42.2%	62.3%	55.4%	72.1%	52.0%	53.2%	47.0%	63.4%	45.7%	71.4%	63.9%	80.8%	58.3%
ETHNICITY																
NZ European	70.5%	67.5%	90.8%	89.1%	60.3%	68.4%	72.4%	72.6%	51.4%	57.9%	63.6%	63.8%	69.2%	78.9%	81.2%	81.5%
NZ Maori	22.7%	12.5%	3.1%	0.0%	13.2%	7.2%	12.2%	5.0%	11.3%	6.1%	10.8%	4.4%	15.2%	8.3%	13.7%	5.6%
NZ Asian	0.0%	10.0%	1.5%	6.3%	13.7%	11.7%	2.7%	14.6%	11.7%	9.9%	2.4%	12.8%	15.7%	13.5%	3.0%	16.3%
NZ Pacific	4.5%	2.5%	0.0%	0.0%	2.6%	0.6%	1.0%	4.0%	2.2%	0.5%	0.8%	3.6%	3.0%	0.7%	1.1%	4.5%
Others	2.3%	7.5%	4.6%	4.7%	10.2%	12.1%	11.7%	3.8%	8.7%	10.2%	10.3%	3.3%	11.7%	13.9%	13.1%	4.3%
OCCUPATION																
High skill	38.6%	48.7%	27.7%	42.2%	45.5%	45.7%	36.0%	56.1%	38.8%	38.6%	31.6%	49.2%	52.2%	52.7%	40.3%	62.9%
Low skill	61.4%	51.3%	72.3%	57.8%	50.1%	52.3%	57.5%	39.9%	42.8%	44.3%	50.6%	35.0%	57.5%	60.4%	64.5%	44.8%

Source: Statistics New Zealand, 2006 Census area unit and territorial unit data (lower and upper limits represent the 95% confidence interval)
Definitions:

Old	Over 60 years
Young	Under 30 years
Mid-age	30-60 years
High Income	Household income > $100,000 pa
High Skill	Occupation manager or professional
Lower/Upper Limit	Sample estimate should be with in the range for 95% confidence

Relevant population:

Rotoroa	Hamilton Lake area unit
Rototuna	Rototuna area unit
Morrinsville	Matamata-Piako District
Karori	Karori North, Karori Park, Karori East and Karori South area units

4.8 Coding of attributes

The coding of the attributes for analysis reflects the change in the various levels for a particular attribute. For example, there is success in removing 35% of hydrilla cover in level 1 relative to the status quo (from 100% to 65% coverage, see Master Table in Appendix 1). Level 1 numeric coding is then 35 (see Table 4.4). Level 3 coding of 100 reflects total success in removing hydrilla.

Table 4-4 Numeric coding

Attribute	Level 0	Level 1	Level 2	Level 3	Description
HYD	0	35	70	100	Total success in removing hydrilla
CHA	0	7	14	21	Total success in preserving 21% charophytes cover
BIR	0	1	2	4	Total success in preserving 4 shags
FISHMUS	0	1	2	3	Total success in preserving 2 fish and 1 mussel (2+1=3)

Water quality utilised effects coding in order to account for non-linear effects in the attribute levels. The non-linear effects arise from differences in utility between any two consecutive attribute levels as demonstrated by Hensher, Rose and Greene (2005, pp. 119-121). The four levels are coded into three variables as shown in Table 4.5.

Table 4-5 Effects coding

Water quality	WQ1	WQ2	WQ3
Level 0 - significantly worse than now	-1	-1	-1
Level 1 – moderately worse than now	1	0	0
Level 2 – slightly worse than now	0	1	0
Level 3 – OK, same as now	0	0	1

4.9 Pooling test

Tests were undertaken to determine whether samples from different locations were significantly different, as to preclude pooling (e.g. pooling the samples from the Waikato region namely, Rotoroa, Rototuna and Morrinsville). The two tests involved interaction variables and the unobserved error.

Interacting the location variable with the environmental attributes (e.g. hydrilla, charophytes, birds, fish-mussels and price) reveals whether location is significant in accounting for the variance in taste intensities. Interaction variables account for the interaction effect where the preference for the level of one attribute is dependent upon the level of a second attribute (Hensher *et al.*, 2005, p.116). Rotoroa, as the sample nearest to the affected lake, was used as the baseline location in creating the interaction variables. The interaction variables showed that there is no significant difference accounted for by location in terms of the environmental attributes of hydrilla, water quality, charophytes, birds and fish-mussels. But, the interaction with the price attribute shows the Wellington interaction being significantly different from the Rotoroa, Hamilton and Morrinsville.

A complementary test for pooling is testing whether the unobserved error accounts for significant differences (Rose, 2009 pers. comm.). This test determines whether there is an error variance linked to choosing the status quo against the alternatives. Using this test for the Waikato region samples showed a significant error term at 99% confidence level. This indicates that the different locations are different due to the unobserved error.

4.10 Models

In choice experiments, the choices made by individuals, the attributes of the alternatives they choose and the characteristics of the individuals are observed. Assuming utility maximising individuals, choice models represent the true but partially observed decision rule adopted with a probability of selecting that alternative which maximises relative utility.

The simple Multinomial Logit (MNL) model was used to initially analyse the responses from each sample. The standard MNL model assumes that respondents have similar preferences (i.e. unexplained error terms are independent and identically distributed (IID)). The standard MNL model resulted in all attributes except for water quality being significant at the 99% level for the four locations. WQ (water quality attribute) is considered significant if any one of the three WQ variables has a significant p value.

To increase explanatory power, the panel version of the Random Parameters Logit (RPL) model (also known as Mixed Logit model) was utilised. The RPL model relaxes the most restrictive assumptions of the MNL model (i.e. respondents have similar preferences) by allowing for heterogeneity of individual utility for the attributes. In addition, correlation between attributes and variance in choosing among alternatives (Alt1 and Alt2 vs. SQ) can also be investigated in RPL modelling. The latter introduces a normally distributed random error term associated with alternatives. Intelligent Halton draws were used to derive the estimates as this process only required one-tenth the number draws compared with simple pseudo-random draws (Bhat (2001) cited by Hensher *et al.*, 2005, pp. 614 - 616). A total of 150 draws were used in the estimation.

The RPL model with normal distribution for the environmental attributes and random parameters for the two alternatives and the status quo yielded the best model fit with adjusted McFadden's R^2 for Rotoroa (0.468), Hamilton (0.390), Morrinsville (0.389) and Wellington (0.439). However, this model did not always perform well for willingness to pay. Specifically the range for the 95%

confidence interval resulted in some attributes with lower limits that were illogical (i.e. negative WTP).

To address the WTP issue, the standard deviations of the attributes are constrained to be a function of the mean (Hensher, *et al.*, 2005, p.614). The triangular distribution is constrained to value of 1 for the environmental attributes (which forces the mean to equal to the spread of the distribution). This resulted in only a slight deterioration, but still a good level, of model fit with adjusted McFadden's R^2 for all four locations ranging from 0.356 (Morrinsville) to 0.464 (Rotoroa). All attributes were significant for the four locations except for STATQUO in the Waikato region locations. The additional specification of random parameters for the alternatives showed that the error term is not significant for Rotoroa and Hamilton.

The results of the models for each of the four locations are summarised in Table 4.6 - 4.9 (Rotoroa, Hamilton, Morrinsville and Wellington models).

These tables present the coefficient mean and standard deviation of estimates and p-values of the parameters. The bottom part of the tables shows several tests of model fit. McFadden's pseudo-R^2 cannot be interpreted in the same way as the R^2 in a linear regression model. Pseudo-R^2 values between 0.3 and 0.4 represent acceptable model fit in a discrete choice model as these are translated as an R^2 of between 0.6 and 0.8 for the linear model equivalent (Hensher *et al.*, 2005, pp 338-339). The model has better fit the higher the LL (log likelihood; i.e. less negative number or closer to zero).

Table 4-6 Rotoroa model coefficients and p-values

	MNL		RPL1		RPL2	
Variable	Estimates	p-values	Estimates	p-values	Estimates	p-values
HYD μ	2.2082***	.0000	3.4253***	.0000	3.4306***	.0000
WQ1 μ	-.1199	.5728	-.2294	.3359	-.2198	.3149
WQ2 μ	.3945*	.0663	.4659*	.0704	.4660*	.0745
WQ3 μ	.3546*	.0919	.5897**	.0145	.5852***	.0084
CHA μ	1.8003***	.0000	2.7479***	.0000	2.7613***	.0000
BIR μ	1.4810***	.0000	2.1998***	.0000	2.1956***	.0000
FISHMUS μ	1.1657***	.0000	1.9046***	.0000	1.9064***	.0000
σ_η	-	-	-	-	.6797	.9264
STATQUO	-1.7094**	.0375	-1.0248	.2179	-2.1245	.7538
PRICE	-.0084***	.0000	-.0136***	.0000	-.0101***	.0000
LL	-328.547		-311.010		-310.961	
Pseudo-R^2			.464		.464	
AIC	1.279		1.212		1.216	
BIC	1.351		1.285		1.297	

*** Significant at 99% confidence level, ** at 95% level, * at 90% level
Note: Standard deviation is the same as the mean, μ indicates that it is the mean value of the parameter

Table 4-7 Hamilton model coefficients and p-values

	MNL		RPL1		RPL2	
Variable	Estimates	p-values	Estimates	p-values	Estimates	p-values
HYD μ	1.3898***	.0000	2.1486***	.0000	2.0933***	.0000
WQ1 μ	.4250**	.0274	.4824*	.0712	.6042***	.0065
WQ2 μ	.4209**	.0369	.5441**	.0382	.6147**	.0115
WQ3 μ	-.0935	.6231	-.0222**	.9343	-.1356	.5003
CHA μ	1.3857***	.0000	2.0340***	.0000	1.8907***	.0008
BIR μ	.9064***	.0000	1.2856***	.0000	1.2185***	.0000
FISHMUS μ	1.1795***	.0000	1.6978***	.0000	1.6521***	.0000
σ_η	-	-	-	-	3.0854	.1184
STATQUO	-1.0823**	.0469	-.4893	.3571	-3.2378	.2519
PRICE	-.0078***	.0000	-.0115***	.0000	-.0112***	.0000
LL	-342.849		-333.007		-330.213	
Pseudo-R^2			.369		.374	
AIC	1.466		1.425		1.412	
BIC	1.544		1.503		1.505	

Table 4-8 Morrinsville model coefficients and p-values

	MNL		RPL1		RPL2	
Variable	Estimates	p-values	Estimates	p-values	Estimates	p-values
HYD μ	1.5211***	.0000	2.4480***	.0000	2.1631***	.0000
WQ1 μ	.0373	.7977	-.3652**	.0315	-.1778	.2326
WQ2 μ	.1909	.1949	-.0053	.9741	.0708	.6884
WQ3 μ	-.0722	.6193	.5377***	.0008	.3272**	.0300
CHA μ	.8252***	.0000	1.4771***	.0000	1.1502***	.0016
BIR μ	.8608***	.0000	1.3834***	.0000	1.2037***	.0000
FISHMUS μ	.7745***	.0000	1.2037***	.0000	1.0296***	.0000
σ_η	-	-	-	-	3.2231***	.0000
STATQUO	-1.1508***	.0037	-.6220	.1223	-3.7056***	.0071
PRICE	-.0063***	.0000	-.0100***	.0000	-.0087***	.0000
LL	-576.387		-552.016		-542.192	
Pseudo-R^2			.356		.367	
AIC	1.501		1.439		1.416	
BIC	1.555		1.492		1.476	

Table 4-9 Wellington model coefficients and p-values

	MNL		RPL1		RPL2	
Variable	Estimates	p-values	Estimates	p-values	Estimates	p-values
HYD μ	1.5534***	.0000	1.9835***	.0000	2.0265***	.0000
WQ1 μ	.2377	.1394	.3775	.1049	.4301***	.0027
WQ2 μ	.4242**	.0108	.6777***	.0036	.7119***	.0000
WQ3 μ	.0303	.8487	-.0924	.6946	-.1379	.3684
CHA μ	1.3512***	.0000	1.6643***	.0000	1.6170***	.0001
BIR μ	1.3190***	.0000	1.6551***	.0000	1.6531***	.0000
FISHMUS μ	1.0511***	.0000	1.3147***	.0000	1.3350***	.0000
σ_η	-	-	-	-	2.5003***	.0000
STATQUO	-1.3340***	.0035	-.9760**	.0287	-2.8340**	.0337
PRICE	-.0107***	.0000	-.0129***	.0000	-.0130***	.0000
LL	-525.588		-514.412		-509.801	
Pseudo-R^2			.390		.396	
AIC	1.392		1.363		1.354	
BIC	1.447		1.417		1.414	

The AIC (Akaike information criterion) and BIC (Bayesian information criterion) are also tests of model fit that trade off improvements in LL with increasing number of parameters (i.e. a higher LL or a lower number of parameters leads to better AIC and BIC). The smaller the AIC and BIC, the better the model fit.

4.11 Willingness to pay and Marginal Rate of Substitution

The WTPs and confidence interval for the four locations are shown in Table 4.10 and Figure 4.2. Willingness to pay (WTP) is generated from the parameter estimates of the environmental and price attributes. As this results in a WTP per unit change, the result has been normalised to represent total success in removing hyrdilla (x 100), preserving charophytes cover (x 21), preserving 4 shags (x 4) and preserving 3 fish/mussel species (x 3).

The calculation utilises equation 4.5, for example, the WTP by Rotoroa residents (Table 4.6) for 100% removal of hydrilla is -2.2082/-0.0084 = $262.88 per household per annum for five years. Note that the estimates of WTP in Table 4.10 may vary slightly from the above calculation (which uses mean values as the WTP estimates) as Table 4.10 estimates of WTP are the result of simulation runs.

The 95% confidence interval for the WTP is also generated. The WTP confidence intervals for the MNL models in the four samples have been calculated using the delta method (Greene, 2000). The delta method creates a linear approximation of the variance for functions of maximum likelihood estimates (Xu and Long, 2005).

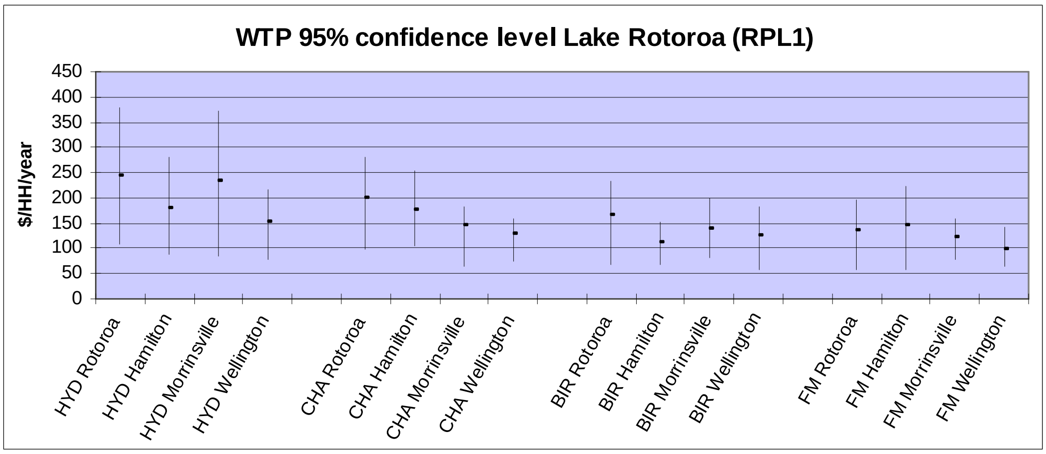
Table 4-10 Willingness to pay and 95% confidence interval (lower, upper), $ per HH/ year over 5 years

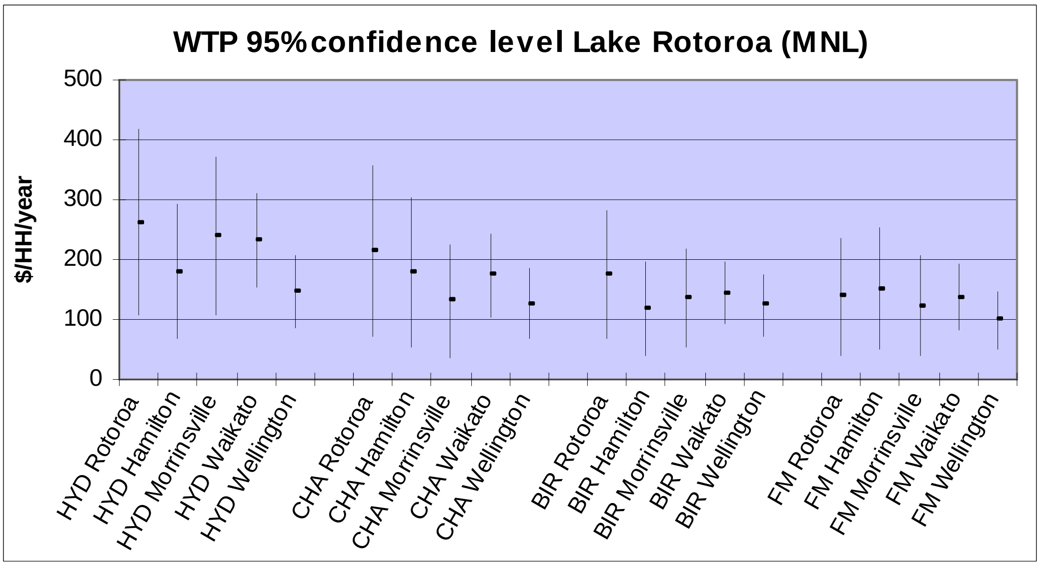
	M N L				R P L 1			
Attribute	**Rotoroa**	**Hamilton**	**Morrinsville**	**Wellington**	**Rotoroa**	**Hamilton**	**Morrinsville**	**Wellington**
HYD	$262.46	$178.70	$240.56	$145.71	$243.71	$178.61	$233.81	$151.05
	(107,418)	(66,291)	(108,373)	(86,206)	(110,378)	(89,280)	(86,372)	(77,215)
WQ1	-$14.25	$54.65	$5.90	$22.30	-$16.91	$42.67	-$35.95	$29.38
	(-64,35)	(-2,111)	(-39,51)	(-8,53)	(-20,-15)	(33,52)	(-51,-29)	(24,35)
WQ2	$46.89	$54.12	$30.18	$39.79	$33.92	$47.06	-$0.51	$51.83
	(-10,104)	(-5,114)	(-18,79)	(6,74)	(26,40)	(37,60)	(-1,0)	(36,73)
WQ3	$42.15	-$12.03	$11.42	$2.84	$43.04	-$1.91	$52.79	-$7.13
	(-12,97)	(-60,36)	(-35,57)	(-26,32)	(26,56)	(-2,-2)	(35,73)	(-8,-6)
CHA	$213.98	$178.17	$130.51	$126.74	$200.34	$176.40	$145.53	$128.52
	(70,358)	(53,303)	(37,224)	(67,187)	(100,280)	(106,252)	(64,182)	(75,158)
BIR	$176.02	$116.54	$136.13	$123.72	$164.33	$111.64	$137.91	$126.87
	(68,284)	(38,195)	(53,219)	(73,175)	(69,232)	(68,154)	(81,200)	(58,183)
FISHMUS	$138.55	$151.65	$122.49	$98.60	$135.28	$145.54	$120.16	$99.24
	(40,237)	(49,254)	(39,206)	(51,146)	(58,197)	(59,223)	(76,160)	(63,141)

The confidence intervals for the RPL models were generated using parameter estimates for each of the 44, 40, 65 and 64 choices analysed (i.e. conditional parameter means) for the Rotoroa, Hamilton, Morrinsville, and Wellington samples, respectively. The parameter estimates for each choice is not a specific individual estimate but a distribution resulting from 150 intelligent Halton draws. The mean and 95% confidence intervals were generated from this range of part worth estimates.

Except for water quality, the WTP and 95% confidence interval generated from both the MNL and RPL models are significantly different from zero and the lower limits are above zero. RPL1 has the advantage of better model fit and generally tighter confidence interval.

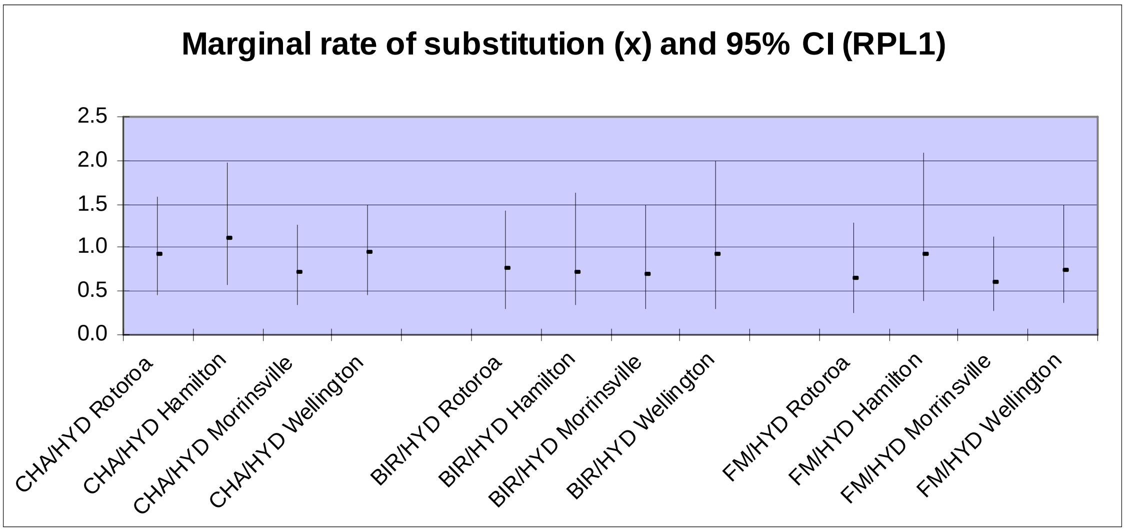
Figure 4-2 Willingness to pay confidence interval – by location





Apart from WTP, where relating the environmental attribute to the money attribute produces a dollar estimate, the marginal rate of substitution (MRS) shows the relative value of one attribute to a reference attribute. The avoidance of hydrilla, which is generally the highest valued attribute, is used as the reference. The mean MRS for Rotoroa, Hamilton, Morrinsville and Wellington and the 95% confidence interval are shown in Figure 4.3. The chart shows that the mean MRS is generally below 1x.

Figure 4-3 Marginal rate of substitution and confidence interval – by location



Note: (x) refers to the mean value and (CI) to the confidence interval

The confidence intervals for WTP and MRS by sample and by attribute show some overlaps. To assess the statistical significance of differences in WTP and MRS, the equality of the estimates is tested using the asymptotically normal test statistic (Campbell, Hutchinson, and Scarpa, 2008) equation 4.6:

$$ANTS = \left(WTP_k^{L1} - WTP_k^{L2}\right) / \sqrt{\left(Var\left(WTP_k^{L1}\right) - Var\left(WTP_k^{L2}\right)\right)} \quad (4.6)$$

where k is the attribute of interest, L1 and L2 are the two locations to be compared and WTP is the WTP or MRS mean.

In terms of WTP for the attributes, each pair of locations is not statistically different at the 95% confidence interval (see Table 4.11). By attribute, the WTP are also not statistically different across the four locations. This implies that the WTP for any particular attribute is similar across locations (e.g. near or distant from the lake).

Table 4-11 ANTS Tests for equality of WTP

	Rotoroa vs. Hamilton	Rotoroa vs. M'sville	Hamilton vs. M'sville	Rotoroa vs. W'ngton	Hamilton vs. W'ngton	M'sville vs. W'ngton
HYD	1.10	0.60	0.55	1.26	0.64	1.16
CHA	0.85	1.48	1.29	1.86	1.83	1.59
BIR	1.56	0.96	0.72	1.53	0.40	-0.26
FISHMUS	0.20	0.51	0.85	1.23	1.56	-0.76

Note: ANTS of less than 1.96 is not statistically different

Comparing each pair of locations, the MRS for the attributes are not statistically different at the 95% confidence level (Table 4.12). Similarly, by attribute the MRS are also not statistically different. This implies that the relationships between attributes are stable across locations and between attributes within a location.

Table 4-12 ANTS Tests for equality of MRS

	Rotoroa vs. Hamilton	Rotoroa vs. M'sville	Hamilton vs. M'sville	Rotoroa vs. W'ngton	Hamilton vs. W'ngton	M'sville vs. W'ngton
CHA/HYD	0.31	0.81	1.33	-0.06	0.76	0.54
BIR/HYD	0.45	0.77	-0.07	0.31	0.43	0.49
FM/HYD	0.53	0.26	-0.69	-0.87	-0.36	0.37

Note: ANTS of less than 1.96 is not statistically different

4.12 Aggregate value

The appropriate way to estimate the benefits of changes that involve multiple attributes is to estimate Compensating Surplus (CS) (Hanemann, 1984). CS is estimated from the benefits people receive from environment conditions both before and after the proposed change as shown in equation 4.7.

$$\text{CS} = \frac{-1}{\beta_{\text{M}}} * (V_0 - \alpha_m V_1) \qquad (4.7)$$

where β_M is the coefficient for the monetary attribute and is interpreted as the marginal utility of income, V_0 represent the utility of the initial state, V_1 the utility of the subsequent state and α_m is the coefficient of the inclusive value (Morrison and Bennett, 2006).

Typically non-market valuation surveys focus on households (HH) and respondents are asked to state their preferences on behalf of their household. Equation 4.8 follows equation 4.7 assuming CS is unadjusted for socio-demographic characteristics.

$$CS_{HH} = -1/\beta_m \left(\sum_i \beta_i \Delta_i \right) \tag{4.8}$$

where:

CS_{HH}	=	compensating surplus per household
β_m	=	coefficient of the money attribute
β_i	=	coefficient of the i^{th} environmental attribute; and
Δ_i	=	change in the quantity of the i^{th} attribute

The aggregation of the changes in the mean WTP for the environmental attributes in the case study results in the CS illustrated in equation 4.9 (2006). In this case CS does not take into account the socio-economic characteristics of the respondents.

$$CS = -1/\beta_{PRICE} * (\beta_{HYD} * \Delta\, HYD + \beta_{CHAR} * \Delta\, CHAR + \beta_{BIR} * \Delta\, BIR + \beta_{FISHMUS} * \Delta\, FISHMUS) \quad (4.9)$$

where conditional parameter means ($\beta_{attribute}$) is a summation for each sample and Δ represent total success in removing hydrilla (HYD), and preserving current levels of charophytes cover (CHA) and species of birds (BIR) and fish/mussels (FISHMUS).

The aggregation uses the 2006 census household (HH) population of Rotoroa (1,479 HHs near the lake), Hamilton (45,726 HHs), Waikato (138,336 HHs), and New Zealand (1,454,175 HHs). The components of CS are derived as follows using Hamilton as an example:
WTP in Hamilton for Hydrilla based on the RPL1 model is $178.61 /HH (see Table 4.10). This is multiplied by the number of HHs in Hamilton minus the number of HHs in Rotoroa (45,726-1,479), which equals $7.9 million (see Table 4.13). The Present Value (PV) for 5 years for Compensating Surplus is calculated at $348 million for the Waikato region and $3 billion for New Zealand (aggregating relevant columns in Table 4.13). These PVs have been estimated using a discount rate of 8% with a sensitivity analysis on the discount rate at 6%. Intuitively, these unadjusted values seem very high. Later in this section adjustments are made for differences between the sample and population characteristics for income and membership of a conservation organisation, both of which reduce the aggregate values significantly.

These estimates of CS are based on estimates of community WTP to have a hydrilla-free lake with current (status quo) levels of charophytes, birds, fish and mussels. CS is a conservative estimate of the value of the lake's natural environment as encapsulated by the four attributes because there is a portion of utility that is unexplained, although in this case the high level of explained utility gives confidence in the results.

Table 4-13 Annual and present value of WTP

Annual value (NZ$m)	**Rotoroa**	**Hamilton**	**Waikato**	**New Zealand**
RPL1				
HYD	0.4	7.9	21.7	198.8
CHA	0.3	7.8	13.5	169.1
BIR	0.2	4.9	12.8	166.9
FISHMUS	0.2	6.4	11.1	130.6
Compensating surplus	1.1	27.1	59.0	665.4
Present value for 5 years				
CS @ 8% discount rate	4.4	108.2	235.7	2,656.8
CS @ 6% discount rate	4.6	114.1	248.7	2,803.0

Notes:
1. Hamilton is Hamilton households less Rotoroa households (i.e. rest of Hamilton)
2. Waikato is Waikato households less Hamilton households (i.e. rest of Waikato)
3. New Zealand is New Zealand households less Waikato households (i.e. rest of New Zealand)

Aggregation bias is caused by three main factors (Morrison, 2000): response rate, similarity of preferences of respondents and non-respondents, and correlation between preferences and socio-demographic characteristics (SDCs). As non-response is not applicable to the survey method used, the correlation between preferences and SDCs, specifically income (i.e. high income and low income) and membership in conservation groups were investigated. Interaction variables of each SDC with the various attributes showed no significant effect on preferences except for income and price attribute in Wellington and membership in conservation group and price in Wellington, Morrinsville and Hamilton.

Despite the lack of significant effect for some attributes, Tables 4.14 and 4.15 show adjustments for income and membership in a conservation group. Methods for adjusting the mean values include adjusting the sample mean, using weighted regression analysis, and the weighted average approach (Morrison, 2000).

Table 4.14 shows the mean household income between the sample and the population in each location. As the mean household income is higher in the sample, mean WTPs were adjusted by factors ranging from 0.72 to 0.85. The impact is a 28% reduction in the PV for New Zealand.

Table 4-14 Annual and present value of WTP (adjusted for income)

Annual value - Adjusted for household income

(NZ$m)	Rotoroa	Hamilton	Waikato	New Zealand
RPL1				
HYD	0.3	5.9	15.5	142.3
CHA	0.3	5.8	9.7	121.1
BIR	0.2	3.7	9.1	119.5
FISHMUS	0.2	4.8	8.0	93.5
Compensating surplus	0.9	20.1	42.3	476.3
Present value for 5 years				
CS @ 8% discount rate	3.7	80.1	168.8	1,901.8
CS @ 6% discount rate	3.9	84.5	178.1	2,006.4

Notes:
1. Hamilton is Hamilton households less Rotoroa households (i.e. rest of Hamilton)
2. Waikato is Waikato households less Hamilton households (i.e. rest of Waikato)
3. New Zealand is New Zealand households less Waikato households (i.e. rest of New Zealand)

Mean household income

(NZ$)	Rotoroa	Hamilton	Morrinsville	Wellington
Sample	$ 73,068	$ 77,250	$ 77,154	$ 79,141
Population	$ 61,767	$ 57,184	$ 55,248	$ 56,651
Adjustment	0.85	0.74	0.72	0.72

Note: Population mean based on Statistics New Zealand 2006 census household income for Hamilton, Waikato and New Zealand.

Table 4.15 illustrates the adjustment for membership in a conservation group. The samples' ratio of membership in conservation groups is compared with the ratio reported by the Department of Conservation in its national survey (DOC, 2008). As the ratio of membership is generally higher in the sample, mean

WTPs were adjusted by factors ranging from 0.39 to 1.13. The impact is a 41% reduction in the PV for New Zealand.

Table 4-15 Annual and present value of WTP (adjusted for membership in conservation group)

Annual value - Adjusted for conservation group membership

(NZ$m)	Rotoroa	Hamilton	Waikato	New Zealand
RPL1				
HYD	0.1	8.9	13.9	111.8
CHA	0.1	8.8	8.7	95.1
BIR	0.1	5.6	8.2	93.9
FISHMUS	0.1	7.2	7.2	73.5
Compensating surplus	0.4	30.5	37.9	374.3
Present value for 5 years				
CS @ 8% discount rate	1.7	121.7	151.5	1,494.4
CS @ 6% discount rate	1.8	128.4	159.9	1,576.7

Notes:
1. Hamilton is Hamilton households less Rotoroa households (i.e. rest of Hamilton)
2. Waikato is Waikato households less Hamilton households (i.e. rest of Waikato)
3. New Zealand is New Zealand households less Waikato households (i.e. rest of New Zealand)

Membership in conservation group

	Rotoroa	Hamilton	Morrinsville	Wellington	New Zealand
Sample	23%	8%	14%	16%	
Population					9%
Adjustment	0.39	1.13	0.64	0.56	

Note: Population based on Depatment of Conservation survey of people involved in conservation outside the home (DOC Annual Report 2008)

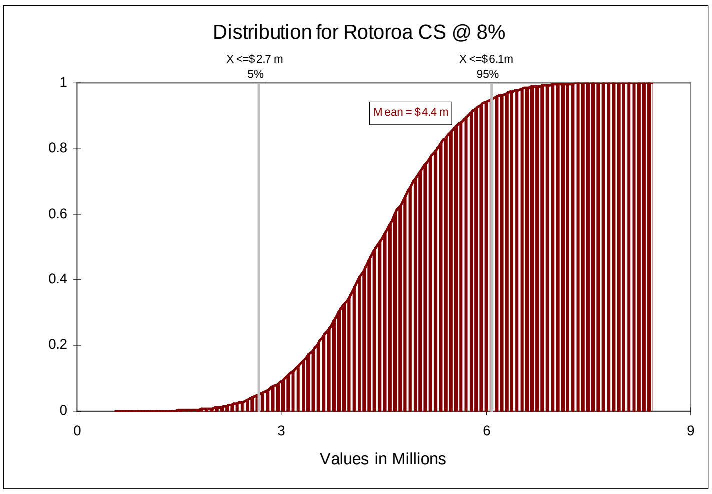
4.13 Accounting for uncertainty

The uncertainty in the mean WTP estimates can be incorporated in the PV analysis using the risk simulation technique QuRA™ (Bell, 2000). Combining estimates to determine the overall uncertainty needs to account for the relationships between the uncertain estimates (i.e. correlation). The environmental attributes exhibit a moderate degree of positive correlation with correlation coefficients ranging from 0.6 to 0.7. Using @RISK, the Excel add-in, the probability distribution of the PV has been estimated by incorporating the means, standard deviations and correlation coefficients between the uncertain WTP variables in the cashflow and simulated over 5,000 iterations. The expected PV results for the four locations are shown in Table 4.16. A sample PV distribution is also shown for Rotoroa with an expected NPV of $4.4 million (8% discount rate) and a 90% chance that the NPV is between $2.7 million and $6.1 million.

Table 4-16 Expected present value of CS (with risk simulation)

Compensating surplus - Expected PV 5 years (NZ$m)	Rotoroa	Hamilton	Waikato	New Zealand
CS @ 8% discount rate	4.4	108.1	236.1	2,659.2
CS @ 6% discount rate	4.6	114.2	248.3	2,804.1

Figure 4-4 Probability distribution for PV of Rotorua



Note that the estimates of CS using risk simulation (Table 4.16) are very similar to the estimates without risk simulation (Table 4.13). This is because mean values are used in each case. The risk simulation demonstrates the uncertainty surrounding the mean. For example, Figure 4.4 shows that there is a 90% chance CS for Rotoroa residents will be within the range of $2.7 million to $6.1 million based on uncertainty in the estimates of WTP for the attributes included in the model.

The above discussion only deals with uncertainty in the estimates of WTP which result from the sampling method and the models used. When these estimates are combined with other information in a CBA of a response there are other uncertainties that should be taken into account as well, for example, the uncertainty around the cost of the response and the likelihood that the response will be successful.

When point estimates are used the impact of such uncertainties are usually explored using sensitivity or 'what if' analysis. More advanced analysis would attempt to estimate the uncertainty around such issues using probability distributions and incorporating these into a risk simulation of the overall net present value.

4.14 Discussion and conclusion

The aim was to elicit quantitative estimates of key environmental values of a freshwater system that could be used for benefit transfer primarily under a situation of extreme time pressure such as in the early days of a pest response. The survey design, which was subsequently evaluated using Ngene (ChoiceMetrics, 2009), required a minimum sample size that was less than 10% of the actual sample size per location. This gives confidence that the experimental design was suitable even for the relatively small sample size used.

The preferred RPL1 model (environmental attributes truncated, triangular distributions and price fixed) had an excellent model fit for all locations equivalent to a linear R^2 of 70-80% and all attributes, except water quality, statistically significant at the 99% level of confidence. Water quality proved somewhat troublesome with lower levels of statistical significance due to the different interpretations people could place on the levels provided (significantly worse, moderately worse and slightly worse and no change).

Overall people were willing to pay more to avoid hydrilla infestation than to protect individual existing attributes of the environment. This is in line with the expected large negative impact of the weed and the likelihood that once in the lake there would be a high probability of it spreading to other waterways. Of the existing environmental attributes charophytes, which are of international significance and at high risk from hydrilla, rated highest followed by birds and fish and freshwater mussels.

There was a generally high degree of consistency in the ranking of WTP for different attributes within each location. While there appears to be a decline in WTP from close to the lake to more distant locations, tests for the confidence interval at 95% confidence level show that there is no statistical difference among locations for the environmental attributes. This may be explained by heterogeneity of preferences within each sample causing overlapping WTP confidence intervals.

Pooling tests to indicate significant difference between the different locations were inconclusive. The first test which tested whether there was a preference for the level of one attribute (environmental) being dependent on another variable (location) showed there was no significant difference for the Waikato region sub-samples, but

Wellington was significantly different. The second test looked at the error variance between alternatives and found that there was a significant difference at the 99% level and it was due to the unobserved error. Taken together these tests indicate there is a significant difference between values held in the region and outside the region (Waikato and Wellington), but there is no significant difference between values held within the region i.e. between residents in Rotoroa, the rest of Hamilton and the rest of the region.

Morrison (2000) notes that distance effect may not exist in all cases and it may be more relevant for use values rather than passive use values and it may be that many factors apart from distance may affect WTP, such as environmental preferences in general. In another study investigating distance effects on environmental values, there was no strong decreasing utility with distance and that the distance effect is variable depending on the type of attribute (Concu, 2007). As this study focused on biodiversity, the lack of a distance effect is consistent with Morrion's view that passive use values (the aesthetic value of the environment) may not exhibit strong distant effects. On the other hand, the relatively high value on the eradication of hydrilla is likely to be due to the threat that it can easily spread to distant sites.

Aggregating the mean WTP for the environmental attributes to the 2006 census household population resulted in a Present Value for 5 years for Compensating Surplus (CS) for all environmental attributes of $348 million for the Waikato region and $3 billion for New Zealand using a discount rate of 8%. Analysis of aggregation bias using interaction variables of income and membership in conservation group SDCs with the various attributes showed no significant effect on preferences. But, making direct adjustments for income and membership in conservation group resulted in a reduction of 28% and 41% in PV respectively. On the assumption that membership of a conservation group is independent of household income, combining these adjustments would result in a CS of $930 million. This compares with the unadjusted CS of $3 billion.

Despite the lack of a statistical distance effect, on-going work on aggregation issues may suggest a lower value for compensating surplus possibly due to such factors as non-attendance (where respondents may ignore a particular attribute such as cost in stating their preferences). Thus, aggregation based on mean WTPs needs to be treated with caution. There is also the issue of mental account, which is the point that people would not be willing to pay for every lake in New Zealand at the same amount as one lake. This casts doubts on the sense of aggregating values beyond the local or district level (Marsh, pers. comm., 2009). On the other hand biosecurity issues represent a special case. It may be that respondents outside the region are thinking that stopping the spread of a pest at the local level means that it will not spread to their region. This may explain their willingness to pay amounts similar to those at the local level. Decision makers need to apply judgement and common sense to such estimates and depending on the situation restrict aggregation of values to the appropriate level, be that local, district, region or national.

Incorporating uncertainty in the mean WTP estimates resulted in a 90% probability that the PV for Rotoroa (local level) would be between $2.7m and $6.1m. Similar levels of uncertainty exist for the other results. By incorporating uncertainty into the analysis decision makers are made more aware of the uncertainty embodied in estimates, which is not apparent when point estimates alone are presented.

The choice experiment to estimate environmental values for a freshwater lake has provided statistically significant WTP values that could be used in a CBA. By sampling communities at varying distances from the lake it was shown that WTP declined the further away from the environmental asset in question; however, this was not statistically significant at the 5% level. This is in line with intuition and gives credence to the aggregated values.

The results are presented as distributions of WTP which gives analysts and decision makers an improved understanding of the uncertainty embodied in the estimates. This uncertainty can be placed alongside other uncertainties, such as the estimates of physical damage from a pest incursion, the cost of the response and likelihood of a successful response when constructing and reporting on the costs and benefits of different response options.

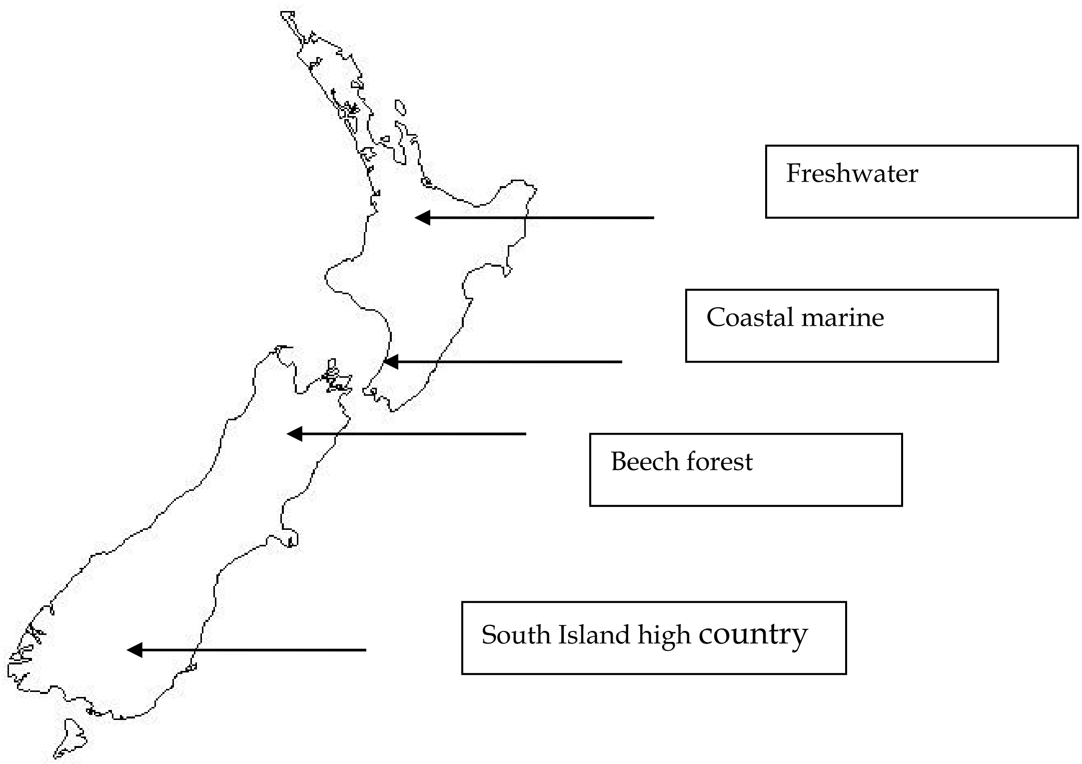
By extending quantitative CBA beyond market impacts to include impacts on environmental values, decision makers are likely to make better decisions on resource allocation.

Chapter 5 : Building the Biodiversity Valuation Database

5.1 Introduction

In the previous chapter the methodology used to estimate biodiversity values for a specific habitat, a freshwater lake, was set out in detail. This case study is one of four that were undertaken during 2007 and 2008, chosen in conjunction with MAFBNZ, so that the biodiversity values could form the foundation of a systematic Biodiversity Valuation Database (BVD). A diverse range of locations (Figure 5-1), habitats and pests were chosen and evaluated using a common methodology to produce comparable values specifically for benefit transfer during an actual pest response analysis.

Figure 5-1 Location of case studies



Each significant ecosystem is threatened by a serious potential exotic pest with a relatively high likelihood of spreading to other areas. In each ecosystem the environmental attributes chosen for analysis included several elements of biodiversity subject to local loss or extinction. Taken together all these aspects mean that the BVD should have relevance to a wide variety of potential biosecurity threats.

In this chapter the process of building the BVD is described starting with a description of the key features of each of the case studies. This is followed by comparative analysis of the biodiversity values. The chapter ends with a review of the key features of the BVD and an outline of how the values can be applied using benefit transfer for biosecurity decision making. The goal was to develop a robust database of values that could be applied quickly and relatively simply.

5.2 Summary of the case studies

The case studies are the building blocks of the BVD. Together they provide a range of values of key attributes of high priority ecosystems that are considered by MAFBNZ to be subject to significant invasive threats. The objective of each case study was to estimate the dollar values of marginal changes to indigenous biodiversity and hence quantify an important component of value for use in cost benefit analysis. The threat and key attributes for each study are outlined below with the detail of each study provided in separate reports.

5.2.1 Coastal marine

The study uses the European Shore Crab (*Carcinas maenas*) as the example alien invasive species (Bell, 2008). The study location is the Pauatahanui Inlet 30 kilometres north of Wellington on the west coast being representative of New Zealand's coastal marine environment.

This crab is particularly aggressive threatening indigenous species in the estuary and sheltered coastline, including: possible elimination of species e.g crabs and shellfish; predation of a range of fish and shellfish species that could result in significant reductions in customary and amateur catch; die back of coastal vegetation through burrowing and subsequent salt intrusion; and restrictions on recreational activity such as children paddling along the waters edge.

The crab has spread from its native habitat in Northern Europe to a number of places around the world and has established itself along the south east coast of Australia. Biosecurity New Zealand considers it a matter of when, not if, it will arrive in New Zealand waters. The Pauatahanui Inlet is one of 350 estuaries around the coast of New Zealand that would provide a suitable habitat for the European Shore Crab.

Four sub-samples of around 50 people were surveyed representing populations adjacent to the inlet (local), Wellington (within region) and different ends of the country (out of region) at Dunedin and Auckland (the Auckland sub-sample was subsequently dropped through lack of participants).

Initial surveying used reallocation of government expenditure as the payment mechanism, but this failed to give significant results and so the local sub-sample (Pauatahanui) was resurveyed using a special tax on households with much improved and statistically significant results. In order to ensure the results would be consistent with the initial survey, the only change made was the money variable which became a special tax with values of zero, $25, $50 and $100 per household per annum for three years instead of zero or $2 million one-off reallocation of government expenditure.

This resulted in a total WTP per household over the four attributes of $185 per annum, with the greatest value placed on loss of shellfish species at $57. Interestingly, loss of ability for children to paddle along the water's edge had only a marginally lower value at $54. Loss of recreational fishing had a value of $37 per annum and loss of vegetation around the estuary $36.

In terms of relativity, loss of shellfish species and loss of paddling for children had approximately 50% higher value than loss of recreational fishing or loss of marginal vegetation.

When these figures are extrapolated across the whole of the community of 3,372 households (based on Statistics NZ 2006 Census data) around the estuary and over the three years of the payment period then the present value of the total loss (discounted at 10%) is $1.7 million with $530,000 of this due to loss of shellfish species. Three years was chosen as the payment period as this was the expected length of a likely biosecurity eradication programme should the crab be detected in the estuary. In retrospect there should not have been a link between payment and response programme.

Calculating marginal rates of substitution (MRS) between attributes enables comparisons to be made between attributes based on relative values and without reference to money. In this study the relative value for the loss of biodiversity (at 1.0) is slightly higher than loss of children's ability to paddle along the water's edge 0.95 and significantly higher than the loss the loss of recreational fishing had a MRS of 0.65, loss of vegetation 0.63.

Coastal marine case study

Threat – Estuarine system threatened by alien aggressive crab

Key insights

- A special tax on households provided superior results compared with reallocation of government expenditure as the payment mechanism

- WTP results per HH per annum over 3 years
  - Loss of shellfish species $57
  - Loss of paddling by children $54
  - Loss of recreational fishing $37
  - Loss of shoreline vegetation $36

- The range of views on value were much wider for loss of shell fish species and no paddling compared with loss of recreational fishing and loss of shoreline vegetation.

5.2.2 South Island high country

The invasive species considered is wilding pines and the study location the tussock grasslands of South Island high country, Mackenzie Basin (Kerr and Sharp, 2007).

Environmental attributes including landscape, endangered flora (*Hebe cupressoides*) and fauna (robust grasshopper and the white bait species bignose galaxias).

Wilding trees threaten to invade large areas (private land, pastoral leases and conservation and unoccupied Crown land) of the South Island. Wilding pines, with seeds mostly dispersed by wind, have demonstrated an ability to spread from an initial 250 hectares to cover an area over 100,000 hectares. The impacts of wilding pine incursion into the high country range from smothering indigenous biota (plant communities, plant and animal species) to merely visual effects (landscape). Intervening impacts include recreational (changing character of recreation sites) and economic (compete with pastoral land use and reduce water availability). With the range of impacts of wilding pines, controlling its spread delivers benefits in the form of avoidance of reduction of amenity values (recreation and tourism) from the high country. The Mackenzie Basin was chosen as the case study site as it hosts several species at risk due to wilding pines and is the area of research focus for the extent and effects of wilding pines.

The location of the four survey groups were from varying distances from the Mackenzie Basin: local - Twizel, within district - Fairlie, within region - Timaru and out of region - Christchurch.

People prefer less wilding pine coverage and for any given amount of coverage, people do not favour large, contiguous blocks. People value the continued existence of the three endangered species and prefer lower personal costs. Of the environmental attributes, bignose galaxias had the highest value at $110 per year for 5 years followed by robust grasshopper ($95/year) and hebe ($58/year). Households were willing to pay $60 per year for 5 years to prevent large blocks of wilding pines rather than scattered plots over the next 20 years.

South Island high country case study

Threat – Sub alpine tussock grasslands threatened by wilding pines

Key insights

- WTP results per HH per annum over 5 years
    - Prevent extinction of
        - Bignose galaxias $110
        - Robust grasshopper $ 95
        - Hebe $ 58
    - Prevention of large blocks of wilding pines compared with scattered plots over the next 20 years $ 60

- Different value preferences can occur that are not systematically related to particular sectors of the community e.g. irrespective of socio-economic level people may have a strong or a weak affinity for the environment

- The survey results represent the views of "informed citizens" in a scenario that would exist following a community awareness campaign and debate during an actual response rather than the views of the community at large.

5.2.3 Freshwater

The invasive considered is the submerged aquatic weed hydrilla (*Hydrilla verticillata)* and the study location Lake Rotoroa (Hamilton

Lake) an urban lake highly modified, but retaining native species (Bell, Cudby, and Yap, 2009a).

Local loss of native species including submerged meadow grass (charophyte species), birds, and fish and mussels, and restriction of recreational activities.

Hydrilla was chosen as the case study invasive as it is MAFBNZ's top priority weed. Although restricted to only three lakes in Hawkes Bay area, it has the greatest potential for negative impacts on New Zealand's freshwater systems. Hydrilla is a submerged freshwater perennial plant that is characterised by prolific growth and tolerance of a wide range of freshwater habitats from clear, murky, still or flowing water; temperature between 0 and 35^{o}C; water depths from a few centimetres to 9 meters; low light to full sun; and a wide range of acidity and nutrient levels.

Potential negative impacts of hydrilla span the range of environmental, economic and social conditions. Hydrilla can dominate freshwater systems displacing indigenous biodiversity (charophytes, pond weeds, milfoils, shags, smelts and common bully), necessitate chemical use for control and increase flooding and erosion risk by clogging waterways. Water quality is reduced by lowering water circulation, reducing light and oxygen availability and the carbon uptake can cause quite large pH fluctuations. Economic impacts include clogging of irrigation and hydro power systems, increased costs for fishers, reduced tourism and increased eradication, control, surveillance, monitoring and public awareness costs to managers of water systems. Social impacts include reduced recreational activity, and negative impacts on public health and Maori cultural and spiritual matters.

Freshwater case study

Threat – Urban lake threatened by the alien competitive submerged aquatic weed hydrilla

Key insights

- WTP results per HH per annum over 5 years, local sample
  - Avoidance of hydrilla $244
  - Loss of charophytes $200
  - Loss of a native bird species $164
  - Loss of a fish /mussel species $135

  (Ref. Table 4.10, p. 110)
- People are willing to pay more to avoid hydrilla getting into the system than to protect existing biodiversity once it is there. This is due to the high chance hydrilla will spread to other water ways
- Loss of the native submerged water plant (charophytes) which are of international significance had the highest biodiversity value, followed by birds, fish and freshwater mussels
- The further people live from the lake the less they are WTP for biodiversity, but this is a relatively weak relationship
- While uncertainties exist for biodiversity values these are not out of line with physical uncertainties.

Lake Rotoroa was chosen in conjunction with MAFBNZ as it has a higher risk of hydrilla invasion, has a long history of management, the lake itself is still largely in a native state, but has a highly modified shoreline and a profile due to surrounding housing and

recreational use. This lake has features typical of many lakes in New Zealand that make it useful to extrapolate from.
Overall people were willing to pay more to avoid hydrilla infestation than to protect individual existing attributes of the environment. This is in line with the expected large negative impact of the weed and the likelihood that once in the lake there would be a high probability of it spreading to other waterways. Of the existing environmental attributes charophytes, which are of international significance and at high risk from hydrilla, rated highest followed by birds and fish and freshwater mussels.

5.2.4 Beech forest

The invasive considered is the European wasp and the study location is the Beech forest at Lake Rotoiti, Nelson Lakes National Park (Kerr and Sharp, 2008).

The objective of this case study is to estimate community preferences and values associated with the impact of wasps and/or their management on indigenous species in the South Island. In the case of wasps, the aim is to measure the change in utility associated with changes in indigenous biodiversity particularly the abundance of birds and insects.

Invertebrates are particularly successful in gaining entry into New Zealand (often as stowaways) and this threat is expected to increase with the volume of trade. While most of these exotic species have no adverse impact, social wasps' impacts include: alteration of indigenous biodiversity food chains resulting in reduction in native birds (e.g directly competing for food (honeydew and invertebrates) and preying on pollinators (hover and bristle flies)); affecting commercial agriculture through reduction in bees; and recreational activities through wasp stings. With the significant impact of wasps, management strategies to control wasps have value in terms of the damages avoided.

Beech forest case study

Threat – beech forest birds and insects threatened by European wasp

Key insights

- Overall people value native species and the avoidance of stings.

- For both birds and insects, there is a higher value attached to preventing native species becoming virtually absent (i.e. extinct) relative to maintaining a very healthy population.

- WTP results per HH per annum over 5 years, regional sample
    - Preventing too few birds $325
    - Preventing too few insects $198
    - Prevent a 1% increase in wasp stings $ 5

- There were similar WTP between Nelson (regional) and Christchurch (national) households with no statistical distance decay effect.

- While the survey was not aimed to be representative of each community or to be representative of the whole of South Island, the results provide an understanding of the likely magnitude of values people hold for attributes which will be useful for cost-benefit analysis of species protection programmes.

Overall people value native species and the avoidance of stings. For both birds and insects, there is a higher value attached to preventing native species becoming virtually absent (i.e. locally extinct) relative to building a bigger very healthy population. Of the existing environmental attributes, birds have a higher value (Nelson WTP of $325 per year for too few birds) compared with insects (Nelson WTP of $198 per year for too few insects). Nelson households were willing to pay $5.25 per year to prevent a 1% increase in the probability of wasp stings. Lastly, there were similar WTP between Christchurch and Nelson households.

5.2.5 Key insights from the case studies

This section summarises the key insights from the case studies. Using schools and community service groups to recruit community members for group meeting-based surveys was quick and cheap. The process conveyed high quality background information and had educational benefits for biosecurity as well. The results represent the response of "informed citizens" in a scenario that would exist following a community awareness campaign and debate about control pest management options rather than the views of the community at large.

The statistical models used were able to explain a large proportion of the variance in people's choices. Statistical power was enhanced significantly by the use of models that allowed for respondent heterogeneity. The models showed that different tastes can occur that are not systematically related to particular sectors of the community. For example, irrespective of socio-economic level people may have a strong or weak affinity for the environment.

The approach highlighted the uncertainty embodied in the estimates of WTP and while significant is not out of line with the uncertainties inherent in the estimates of physical damage from a pest incursion. Decision makers are often presented with point estimates of key variables, with little indication of the robustness of the estimate. Quantifying the uncertainty surrounding the estimates provides an additional dimension for decision makers to weigh up options and improve the quality of decisions.

Respondents were willing to pay more for the avoidance of pests than for the local preservation of indigenous biodiversity. This is likely to be related to the high chance of pests spreading from the initial incursion to the rest of New Zealand.

Results support previous studies, which show active use values, such as boating, tend to reduce much more significantly with distance than passive use values, such as loss of biodiversity. In general the case studies show no statistically significant distant decay effect for indigenous biodiversity values.

While the surveys were not aimed to be representative of each community the results provide an understanding of the likely magnitude of values of the attributes which will be useful for cost-benefit analysis of species protection programmes.

5.2.6 Using the case studies for benefit transfer

The results presented in the case studies are hypothetical in the sense that they are not the focus of actual response programmes. In two cases the pests are not present (crabs and hydrilla) and in the other two (wilding pines and wasps) the pests are already established. To be useful to MAFBNZ the results of the case studies need to be able to be transferred to the analysis of new incursions using the benefit transfer method as outlined in Chapter 3. Benefit transfer is still an evolving discipline, and to date there has been minimal experience using values derived from choice modelling. Given the uncertainties and, recognising the scientific uncertainties involved as well, it is prudent that analysts should focus on the orders of magnitude for key values rather than precise point estimates.

Key issues to be taken account of in the transfer process include differences in site and population between the study and transfer sites, framing differences in the choice experiment including scope, scale, seasonality and welfare measure aspects (Rolfe and Bennett, 2006).

Most of the work undertaken to compare methods of benefit transfer and evaluate the accuracy of the methods has been conducted on CV studies. Usually the original studies were undertaken without the thought of benefit transfer in mind so differences in experimental design, data collection and econometric model make comparisons very difficult and large transfer errors have been found. Obviously the quality of the analysis for the study site or sites is a major determinant of the quality of the value estimates at the policy site.

5.3 Building a BDV using CM results

5.3.1 Existing databases

Databases of previous studies are held in New Zealand and also in Canada. Enough basic information is held about each study in the database for an analyst to assess whether a particular study might be useful for benefit transfer. An important piece of information is usually a link to a journal article or working paper that describes each study in full including the data and methodology adopted. The case studies that underlie this thesis will be submitted to these databases.

Geoff Kerr of Lincoln University maintains an inventory of New Zealand non-market valuation studies (Kerr, 2009). The Environmental Valuation Reference Inventory (EVRI, 2009) developed and maintained by Environment Canada has become the major world depository of environmental valuation studies. While there are many hundreds of recreational valuation studies, particularly using contingent valuation, the number of choice modelling studies is relatively small but growing rapidly. When this research started in 2006 there were no studies related to biosecurity recorded in this database.

5.3.2 Biodiversity values for biosecurity

At the start of this research, surveys of the literature (Bell and Kaval, 2004) and (Sharp, Kerr, and Kaval, 2006) found no studies that related to biodiversity values for biosecurity. In order to fill this gap, and in consultation with MAFBNZ, the four case studies were undertaken with the objective of starting a database of biodiversity values that could be used in biosecurity decision making. A similar methodology was adopted in each case so that the resulting WTPs could be compared. As identified by Desvousges, Naughton and Parsons (1992), this is a basic requirement in order to compare source studies. This is supported by van Bueren and Bennett (2004) who state that benefit transfer is expedited by developing a specific database of values for subsequent case study applications, such that any necessary adjustments can be explicitly modelled.

Typically benefit transfer is based on transferring the information from one or more existing studies. Windle and Rolfe (2007) adopted a different approach where a series of valuation studies were undertaken specifically to build a reference database of values for benefit transfer purposes. This built on earlier work by van Bueren and Bennett (2004) and Morrison and Bennett (2004). The Windle and Rolfe approach has benefits in that the design of the non-market valuation studies and data collection is conducted specifically to ensure accurate benefit transfer and that any adjustment factors can be explicitly modelled. They undertook a series of choice modelling exercises designed to develop a benefit transfer framework for the condition of natural resources in regional areas of Queensland, Australia. They found that small differences in scope such as between region and state do not significantly affect values and so there is some promise that systematic databases can be developed for benefit transfer. In doing so, they suggest that instead of presenting broader and by implication less well defined choice sets, it is preferable to present more narrowly scoped and precisely defined trade-offs (Rolfe and Windle, 2008).

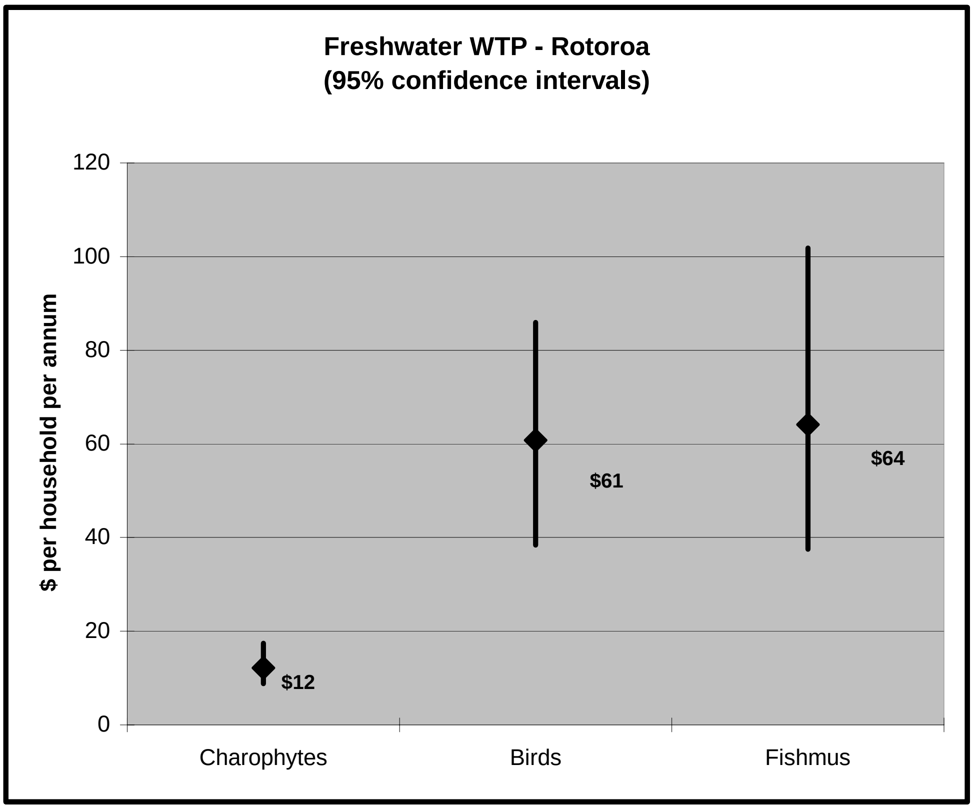
A similar targeted approach has been conducted in this research, but specifically for biosecurity issues.

In the next section, the key features of the four case studies that are common are reviewed and then final adjustments in methodology made to make them as comparable as possible. But first the key elements of the WTP estimates – the mean and confidence limits – are described.

5.3.3 Best estimates and confidence limits

Figure 5.2 shows the estimates of WTP for changes in the key attributes of the freshwater ecosystem prior to normalisation (i.e. 1% change in charophytes c.f. 21% x $12 = $252 when normalised). Note that these estimates differ somewhat from the results of the original case study as a slightly different model was adopted for comparison purposes (see section 5.3.4), the key difference being the dropping of the water quality attributes due to lack of statistical significance.

Figure 5-2 Attribute best estimates and confidence intervals: Lake Rotoroa



Note: Charophyte value is for a 1% change, Birds 1spp. and Fishmus 1 spp.

The extent of the line can be interpreted as the range over which there is 95% confidence the mean will lie somewhere on the line. The dot mark on each line shows the best estimate of household WTP.

For charophytes, a 1% change in the coverage of Charophytes in the lake is valued at $12/HH/yr for 5 years for people living in close proximity to the lake. Similarly the WTP for loss of one bird species is $61 and for loss of one fish species $64.

While the range is much wider for birds and fish compared with charophytes, relative to the size of the best estimate the ranges are similar all resulting in significance at the 1% level. This means that confidence can be taken that the values are different from zero (see Appendix 2 for the model outputs).

5.3.4 Comparing biodiversity values

When comparing the values from different studies, the more similar the studies are the better the results. This is the conclusion of Johnston (2007), who contended that as a rule, generalisation errors are smaller in cases where transfer and study sites are similar. Taking this as a guiding principle, the four case studies undertaken to form the basis of the Biodiversity Valuation Database were re-evaluated using a common model as explained below. The earlier results presented used case specific models. As the same choice modelling process had already been adopted, comparisons between studies were built on solid ground. The key features of the case study process were: use of ecologists to understand the science; focus groups to select the attributes; convenience sampling to generate priors for efficient survey design; a hybrid community meeting approach with informed respondents to collect data; and choice modelling to analyse the data.

In the original studies, varyious models were tested and the model with the best statistical fit and highest explanation of the data was chosen for the presentation of results. In the comparative analysis the panel random parameters logit (RPL) model was chosen, which assumes that preferences vary between individuals, but an individual's preferences are consistent. This follows Colombo, Calatrava-Requena and Hanley (2007) who found that utilising a random parameter approach, which includes the respondents' taste heterogeneity, significantly reduced the magnitude of the transfer error. In addition, all environmental attributes were assigned constrained triangular distributions. The constrained triangular distribution forces the mean to equal the spread of the distribution where "the density starts at zero, rises linearly to the mean and then declines to zero at two times the mean" (Hensher et al., 2005, p. 614). This specification is appealing for WTP parameters as it estimates easily and gives good results with non-negative estimates, but it does reduce heterogeneity, essentially trading-off statistical fit for a desirable behavioral property. The price attribute was fixed, which is a common practice that makes welfare measurement easier (Colombo et al., 2007).

This model specification is a compromise which allowed the four case studies to be compared with acceptable model fit and statistical significance. The RPL model with constrained triangular distributions for the environmental variables and price fixed produced better results than alternatives such as the MNL model or using normal distributions for the attributes.

When the models were run, all attributes were significant at the 1% level and the reduction in fit was small. All models had acceptable levels of explanation as shown by the pseudo-R^2 statistic (see Appendix 2).

In order to be useful, the estimated WTP values from the survey samples must be able to be transferred and aggregated up to relevant populations in policy studies. As the samples cannot be described as random samples, because of potential bias due to significant self-selection, the usual measure of adjusting for non-response (as described in section 4.2) is not applicable. For the meeting approach, aggregation bias must be done through adjusting the mean values.

Bateman (2009) demonstrated that economic theory driven transfers outperform both univariate (mean values) and best-fit function transfers. For example, adding the theory driven variable 'income' to a function transfer dramatically reduced transfer error. But adding variables which economic theory has no prior expectations (e.g. age) induced context specific error and increased error rates (p.36).

This is in line with the view that a transfer approach, which systematically adjusts the transfer values, is clearly preferable from a policy perspective compared with transferring unadjusted values and accepting some bias. When transferring values, the driving issue is the changing context between study and policy site. This can include a change in the provision of a good, a change in socio-demographic characteristics (SDCs) and changes in site characteristics. The solution is to derive models that are consistent with economic theory and focus on the factors which are likely to be consistently present in individual's utility function, irrespective of location e.g. income and whether the respondent is a member of a conservation organisation.

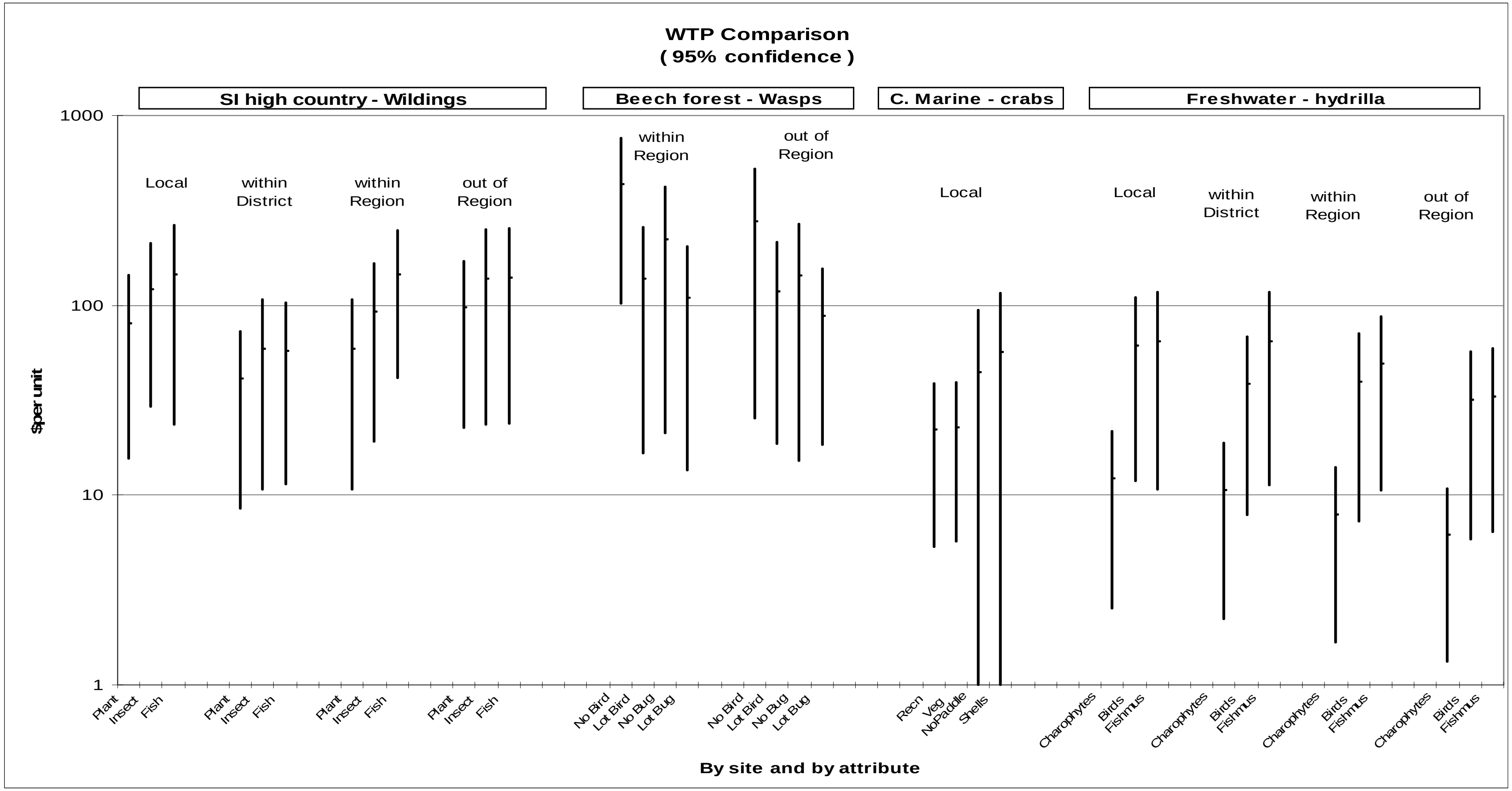
When considering what is important in transferring values Rolfe *et al.* (2006) note that tests conducted for transferring CM values by Morrison, Bennett, Blamey and Louviere *(1998)*, Rolfe and Bennett (2000) and van Bueren and Bennett (2004), show that population differences were more important than site differences. In other words, differences between the populations between study sites and policy sites explained more of the transfer errors than differences in site characteristics.

In this section recent lessons from the literature have been reviewed to ensure the best possible estimates of values are obtained from the transfer process. These lessons will be incorporated into the process of transferring biodiversity values from the Biodiversity Values Database (BVD) that will be covered in Chapter 6. The estimated WTP results for the comparative analysis of the four case studies are provided in Figure 5.3 and Appendix 2.

5.3.5 Biodiversity Valuation Database

Figure 5.3 shows the results of the case studies using the same RPL model. Together this data forms the foundation of a BVD. Appendix 3 provides the details of means, standard deviations and 95 percent confidence intervals by case study, location of sample and attribute. A key feature of this chart is the wide confidence intervals at the 95% level. This shows that people have widely differing views of the value of the environment. A log scale has been used to highlight the lower ends of the ranges. In order to be useful as transfer values the estimates must be statistically significantly different from zero.

Figure 5-3 Biodiversity Values Database - attributes across case studies



Note: $per unit refers to a change in the value of each attribute

Note that all attributes are significant at the 95% confidence level, except NoPaddle and Shells in the Crabs study, which are significant at the 90% level.

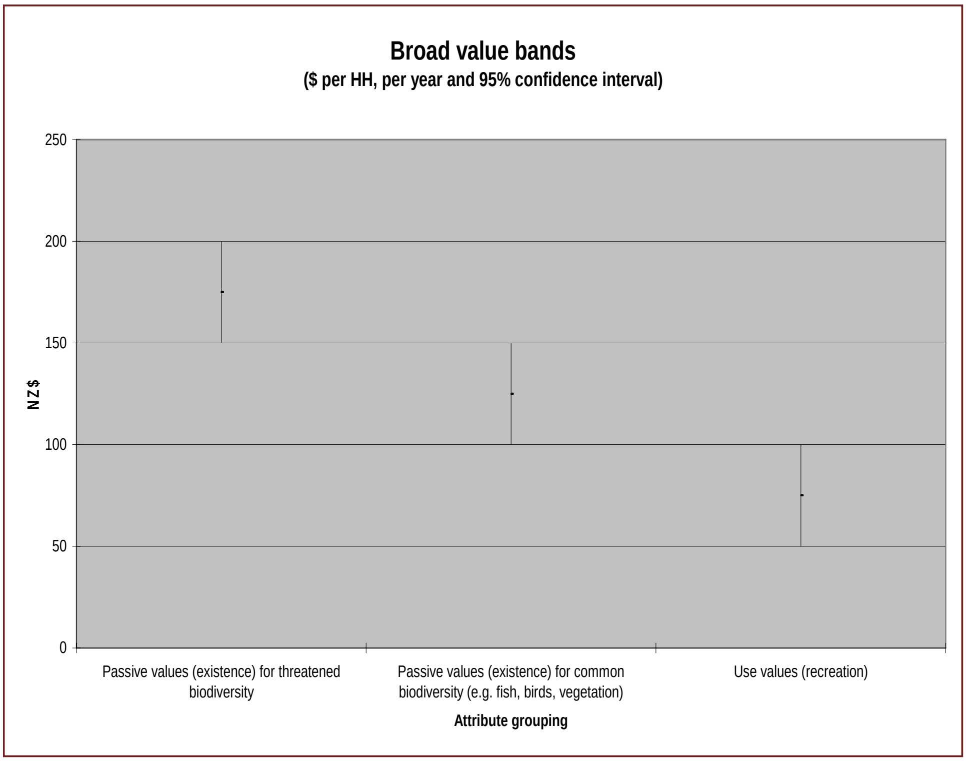
5.3.6 Value Bands

As the body of choice experiment and benefit transfer research grows, general trends are emerging in the environmental values that have been measured. For example, use values tend to reduce the further away from the study site, as opposed to existence values which have less variation with distance. There are many more examples in the literature, but most of these appear to be case specific and so are not helpful in establishing trends.

At first glance Figure 5.3 may not appear to exhibit any patterns useful for informing biosecurity decision makers. What is being looked for are patterns in the results. It is important that patterns can be recognised if the results of the study are to be used at the investigation stage of a response, where high level rapid decisions are needed to prevent the pest becoming established.

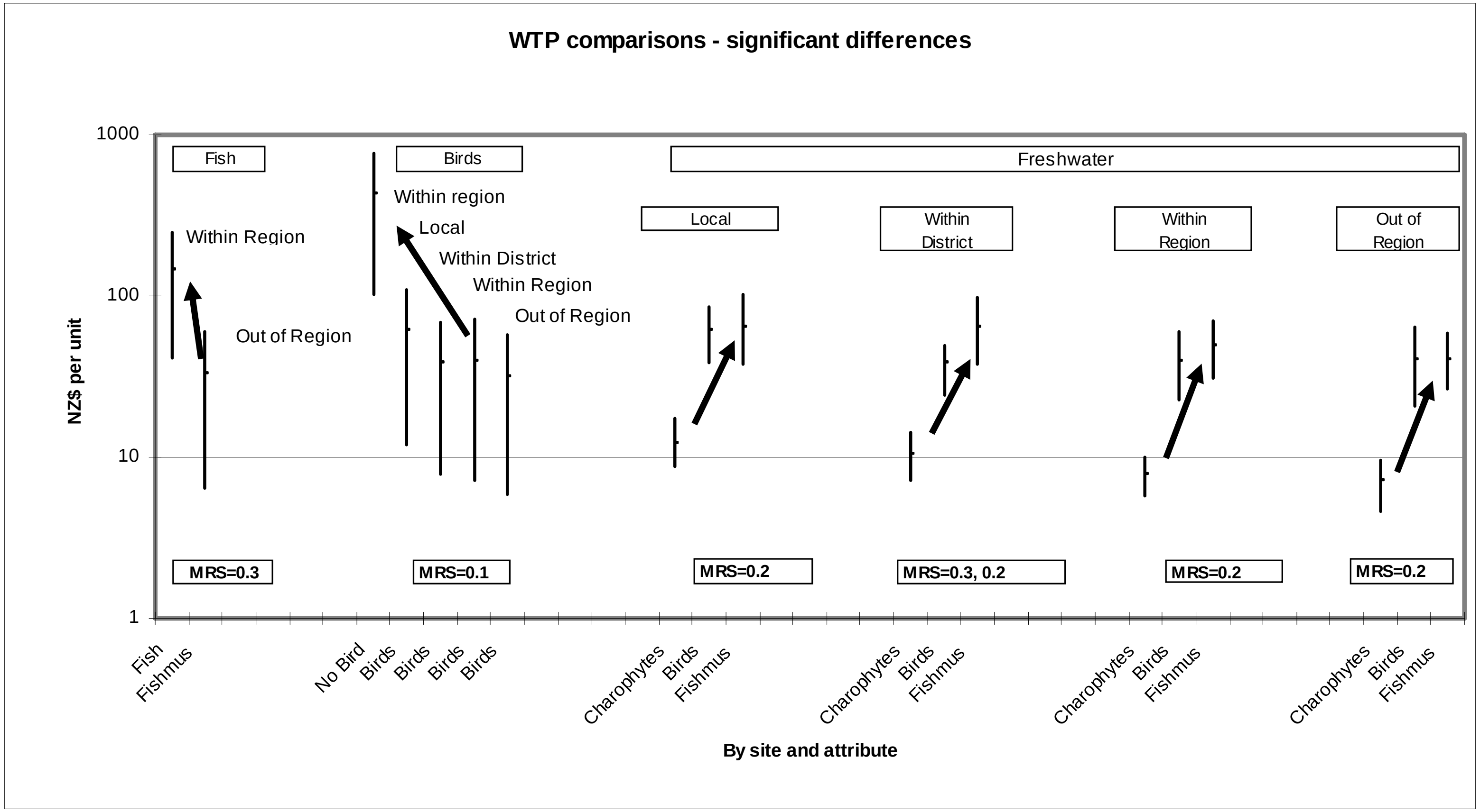
The central question is whether or not values for certain types of environmental attributes can be categorised and grouped into broad bands. Figure 5.4 shows a very simple illustration of this idea, using three categories: high, medium and low. High values may include passive use existence values for endangered species. Medium level values may include passive use existence values for common biodiversity that is locally under threat, while, low values may include indirect active use values such as recreation.

Figure 5-4 Value bands



This idea has been explored with comparative data. Figure 5.5 shows the comparisons that are statistically significant at the 5% level, that is, where there is only a 5% chance that the difference between attributes is not significant i.e. reject the hypothesis that the values are the same.

Figure 5-5 WTP comparisons with significant differences



5.3.7 Significance of WTP difference estimates

In order to determine whether the comparisons were significant, the equality of the WTP estimates was tested using the asymptotically normal test statistic (Campbell et al., 2008):

$$ANTS = \left(WTP_k^{L1} - WTP_k^{L2}\right) / \sqrt{\left(Var\left(WTP_k^{L1}\right) - Var\left(WTP_k^{L2}\right)\right)} \tag{5.1}$$

where k is the attribute of interest, L1 and L2 are the two locations to be compared and WTP is the WTP or MRS mean.

This allows unambiguous results to be estimated based on Z values and p(Z) being less than 0.05 showing that the difference is significant at the 5% level i.e. there is less than a 5% chance of there being no significant difference, where:

$$Z = \mu / \sigma \tag{5.2}$$

$$p(Z) = 2(1 - NORMSDIST\,|Z|) \tag{5.3}$$

NORMSDIST is the standard normal cumulative distribution with mean zero and standard deviation one.

As an example, consider the results for Fish from Figure 5.5. This shows that the out of region respondents are prepared to pay only 20% of the amount within region respondents are WTP to prevent the loss of a fish species (see Figure 5.5). This is shown by the marginal rate of substitution (MRS) of 0.2, which is the ratio of the out of region estimate to the within region estimate. For out of region, it was the **local** loss of a fish species in Lake Rotoroa, while for within region people it was **extinction** of a fish species in the South Island high country i.e. $33 and $145 respectively.

None of the other fish comparisons were significant at the 5% level, implying that there is no significant difference between values for fish in the South Island high country and Hamilton Lake for the samples surveyed.

Next consider Birds. The best estimate for the local loss of a bird species in Hamilton Lake was 10% of that for the loss of bird abundance in South Island beech forest for within region people (MRS 0.1). WTP of $32-$61 for Lake Rotoroa and $431 for beech forest. There was no significant difference at the national level for loss of bird abundance (out of region people at $274) compared with local loss of a bird species at Lake Rotoroa. It is the wide dispersion around the best estimate that precludes making strong (statistical) assertions about differences.

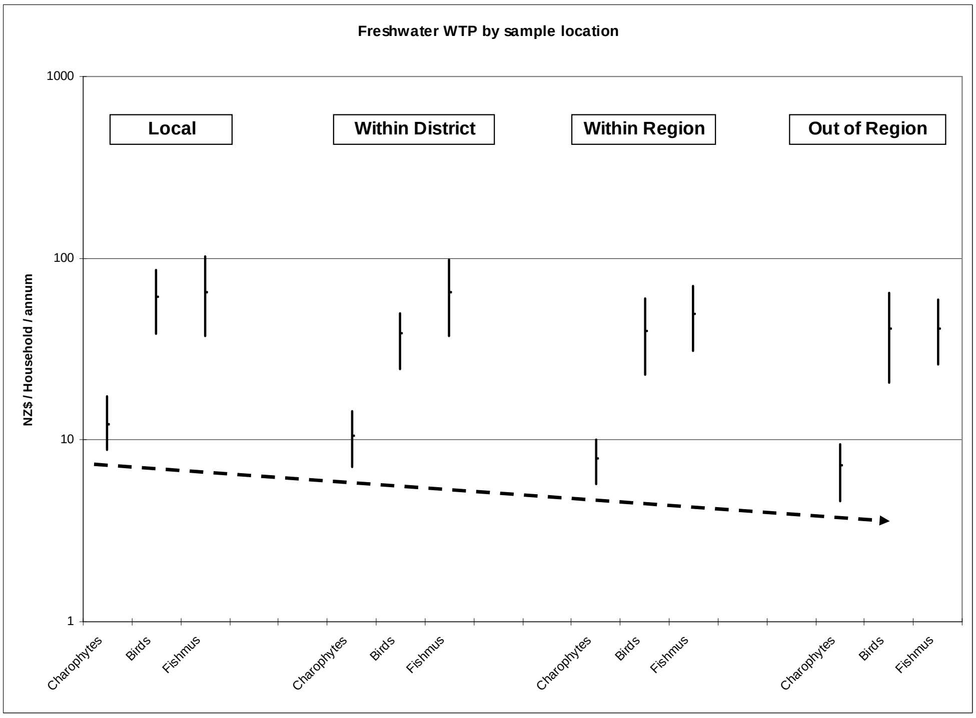
Within the case studies, the only comparison that is significant is that of charophytes with birds and fish. In each of the four samples, a 1% change in charophytes was valued at between 20% and 30% of the local loss of a bird or fish species (MRS 0.2 and 0.3).

When making comparisons between and within studies a note of caution is warranted. There is the potential for results to be confounded by differences in the studies, such as between populations, extent of species loss and proximity to the issue. In this study care has been taken to address confounding by using a common methodology for surveying and analysis, nevertheless the issue remains (see section 5.4.1 for further discussion).

It is possible to undertake further analysis for the freshwater attributes. Figure 5.6 shows the estimates of WTP across the four freshwater samples in the survey.

Populations were sampled at the local level next to the lake (Rotoroa), at the district level across the other side of Hamilton city, within region (Morrinsville) and out of region (Wellington). This is done to show whether WTP changes with distance from the resource at risk.

Figure 5-6 Comparison of Freshwater WTP by location of population



As expected the results show WTP declined with distance from the lake (from local to within district to within region and to out of region), although only modestly. WTP for a change in Charophyte cover declined from $12 for local to $11 for within district to $8 for within region and to $6 for out of region. The dashed inclined line on Figure 5.6 shows this trend, although the differences between these levels were not statistically significant.

Similar trends occurred for the other attributes: Birds: $61, $38, $39, $32 and Fish: $64, $64, $49, $33.

While WTP might be expected to decline with distance because of the wider range of alternatives available to respondents, this seems not to occur for passive use values. One observation from the literature is that active use values (e.g. recreation) decline faster with distance than passive use values (e.g. aesthetics).

The above discussion relates to comparisons of attributes between studies and within studies where analysis showed there was a significant difference at the 5% level. The majority of comparisons did not show a significant difference and therefore the attributes are statistically the same. In other words, most indigenous biodiversity WTP values are similar, both within and between studies.

5.3.8 Conclusion regarding bands

From the results so far the following conclusions can be made with regard to the studies. People consider the local loss of a bird species as equivalent to about 10% of the loss associated with the national extinction of a bird species. The welfare loss of a local freshwater fish species is equivalent to about 20% of the welfare loss associated with the extinction of a freshwater fish species. A one percent change in a rare freshwater plant at the local level is equivalent to about 20% of the welfare loss of a local bird or fish species. While people have values for indigenous biodiversity that can be quantified, these values differ widely for any particular issue and often within sample variation is much wider than between sample variation. Because of the inherent difficulties in making comparisons between studies, the widely differing views and limited data, no clear bands for different types of losses have yet emerged.

5.4 BT: utilising the database for decision support

5.4.1 Key Issues

Estimating biodiversity values involves many assumptions and there is wide scope for the analyst to make decisions that can influence the results. The lessons from this approach are as follows.

Framing. The extent to which the scale and scope of different studies are similar, is important to the conclusions that can be drawn from comparisons when transferring values from one site to another. If scale (i.e. local, district, region or national issue) and scope (e.g. narrow and based on a specific species, species abundance or more broadly with a whole ecosystem) are significantly different it is doubtful that meaningful values can be transferred. It is very important to understand how values are estimated in the underlying primary studies before attempting to transfer values to a new situation. The errors in the primary study estimates must be added to the errors that occur when transferring an estimate from one site to another. Together these errors can be very significant, but often the uncertainties in the science (e.g. physical changes) may be even larger and this will affect the uncertainty in the economic values.

Sample demographics. It is important to collect socio-demographic characteristics of the respondents, and that these are compared to census information. Where samples vary significantly from the population this should be made clear so account can be taken by decision makers. Simple adjustments such as income and/or membership of a conservation organisation should be applied when there are significant differences between the study group and the policy population. Self selection bias is still a potential issue for analysts. Even after checking against census information, participants may have other unobserved traits that make them more or less informed or environmentally conscious. This is an issue for all forms of surveying with those who hold strong views, either way, more likely to opt in.

Economic conditions. Surveys are undertaken at a point in time and there will be an underlying set of economic circumstances at play. The values that respondents give in buoyant economic conditions may be significantly different to those given during a recession. If values more than a year apart are compared, they need to be adjusted for inflation. If values are being transferred between countries then adjustments should be made using purchasing power parity exchange rates.

Informed responses. One view is that informing the respondents will bias the results. A counter view is that it will not be possible to get sensible estimates of value without an informing process first. This is because of the complex nature of biodiversity and biosecurity issues, which few people will have considered in detail. In the studies reported here, respondents were provided with the information that it is expected MAFBNZ would provide if a real incursion were to occur. It is important that the cost of a response is not discussed as this would shift the emphasis from the value of biodiversity to the respondent to the cost of the response.

Willingness to pay. How real are stated preferences? It is very difficult to verify whether stated willingness to pay in a hypothetical situation would be backed up by actual payments in a real situation. However, there are examples of WTP studies being used to justify special taxes for environmental policies that are then accepted by communities. For example, a special tax to improve water quality in Lake Rotorua (Bell and Yap, 2004). Overseas, stated preference studies help inform environment court decisions, particularly when environmental degradation is involved. These studies go back at least to the famous case of the Exxon Valdez oil spill (Arrow et al., 1993).

Marginal dollar values. Valuing indigenous biodiversity involves marginal values rather than absolute values. It is the magnitude of the values which communities place on changes in the environment that is of interest, not the absolute value of the environment *per say*. It is not possible to say that a particular ecosystem is worth $x, but it can be said that people are WTP $y to improve the environment or avoid a deterioration.

Non-Monetary values. A major advantage that choice modelling (CM) has over contingent valuation (CV) is that CM also provides relative (non-monetary) values of environmental attributes. This overcomes a major objection that some people have of the use of economic tools to value biodiversity. An output of a CM study is the marginal rates of substitution (MRS) of one attribute for another. It can be said, for example, that people suffer a welfare loss from extinction of a species about 10 times the welfare loss of a local species.

Core cultural values. The closer values are to core faiths or beliefs and peoples' commitment to these the less relevant dollar values become. This is because core faiths are held tightly and are less likely to be traded off for money. That said, social and cultural values change over time as circumstances change and a value held in a by-gone era may not hold today and today's value may not hold in a future era.

Art or science. There is both art and science in estimating biodiversity values. Lying behind the estimates is a huge body of peer reviewed literature embodying theoretical sound concepts, such as utility theory and statistical significance. However, how the science is applied is just as important to the final result. The role of the analyst is to strive to minimise bias and provide independent and objective advice to inform decision makers.

5.4.2 Key findings

At this point it is useful to pause for a stock take on what has been accomplished for estimating indigenous biodiversity values for biosecurity response.

The non-market valuation tool, choice modelling, enables the estimation of specific environmental attribute values that are representative of particular ecosystems. These models can capture the values people and communities regard as important, and these values can be transferred to new situations.

Newly released software (Ngene) enables the design of efficient experiments that produce statistically significant results using relatively small samples of the populations that might be affected by pests and subsequent responses. Ngene was used to validate the efficiency of the designs used for the case studies.

The models that can be used to estimate biodiversity values are continually improving and it is now possible to model the taste differences of groups and individuals rather than assume everyone is identical in their views. Latent class (LC) and random parameter logit (RPL) models can do this using Nlogit software taking over from the more restrictive assumptions underlying the multinomial logit model (MNL).

The foundations of a systematic Biodiversity Values Database (BVD) have been laid. The BVD contains four case studies across very different ecosystems that have been undertaken using similar methodology. These studies provide 36 estimates of environmental attributes across 11 different populations throughout New Zealand. This forms a rich information base about the values people hold for various attributes ranging from the extinction of plants and animals to local removal only.

Noting that confounding influences are a concern, the analysis undertaken has shown that there are significant differences between the values of some attributes. But the majority of values are not significantly different implying that people generally hold much the same values for the loss of indigenous biodiversity. They are willing to pay similar amounts to protect different types of indigenous biodiversity and these values do not alter significantly with distance from the incursion site.

The concept of bands of value is at an early stage of testing. Currently there are only a few comparisons on which to draw in order to establish whether there are bands of value for different environmental attributes. As more studies are undertaken to estimate biodiversity values, these can be added to the BVD enhancing its usefulness.

A key principle in transferring values is that better results are obtained when circumstances are similar. This applies to both the environmental attributes being transferred and the populations at both the study and policy sites. Educating the public and decision makers on the limits of the estimates and the uncertainty that surrounds them is an important role of the analyst.

The BVD forms a base on which new biodiversity attributes values can be added. As this happens and the database is refined it will become increasingly useful to analysts. If new primary studies are undertaken with a view to adding to the database, using compatible methodologies, then the value of the database will be maximised. If the BVD becomes a living thing that is built on and refined over time it will become an increasingly valuable tool, not only to MAFBNZ, but more widely to decision makers in other natural resource management areas. The challenge is to begin using the BVD so that strengths are built on and weaknesses minimised. The ultimate test of the BVD is whether its use improves decision making on responses to exotic pests that impact on both industry and native plants and animals.

5.4.3 Summary

Establishing the systematic biodiversity valuation database is a major step forward in developing a decision support system for MAFBNZ. There is now a pool of biodiversity values for a diverse range of ecosystems in New Zealand that have been derived from community preferences based on changes induced by exotic incursions of pests and diseases. The values are thus specifically related to biosecurity. Because the same methodology was used to estimate the values, they are directly comparable with each other. Building on the BVD will enhance its value for biosecurity and allied natural resource decisions.

The next chapter describes the process for utilising these biodiversity values in the CBA of response options.

Chapter 6 : Transferring Biodiversity Values

6.1 Introduction

In this chapter the process of transferring biodiversity values from the BVD to the CBA of an actual incursion situation is demonstrated. The method outlined has been designed specifically for MAFBNZ to use during an actual biosecurity response as outlined in Chapter 2. In this situation time constrains the analyst to using existing information rather than conducting primary data gathering.

6.2 The benefit of a response: compensating surplus

Information on economic values can improve decision making at both the investigation and CBA stages of a response. In each case, the aim is to utilise values of indigenous biodiversity to improve the estimate of the benefit of a response. For this the concept of compensating surplus (CS) is utilised as set out in Section 4.12 and Equation 4.7.

The WTP estimates are multiplied by the estimated changes in the quantities of the environmental attributes to estimate CS for each household (Equation 4.8).

The estimate of CS for each household is then multiplied by the number of households affected to provide an estimate of the total benefit. Care must be taken to ensure that the estimate applies to the correct population. For example, a WTP estimate for the local area should be multiplied by the number of households in the local area. If there is a within district estimate of WTP then this should be multiplied by the district population minus the estimate for the local population, similarly for within regional and out of region estimates of WTP. The separate estimates are then summed to obtain a total.

The estimate of the total relates to one year, but the WTP question respondents usually answer relates to a number of years, typically five. As a consequence, the annual estimates need to be discounted over the relevant number of years to obtain a Present Value (PV) based on a chosen discount rate. A common rate should be used so that comparisons can be made between biodiversity studies. For MAFBNZ the rate is likely to be in the range of 6% to 10% and the actual rate determined in consultation with The Treasury. Which ever rate is used it should be subject to a sensitivity analysis because discount rates have a material impact on values and there is no agreed 'correct' rate.

The key difference between Investigation and CBA is the amount of time available to do the analysis and thus it influences the rigor that can be applied to the analysis.

6.3 Investigation

The time for investigation is typically extremely limited. As a rule, in order to be considered, values would be required in about a week. This obviously precludes undertaking a primary non-market valuation study, which may take 6 - 12 months. However, the results of this study provide the beginnings of a database for indigenous biodiversity transfer values that can be drawn on. In order to demonstrate how such a database can be used in a response situation, a simple example using the direct transfer approach is demonstrated.

6.3.1 Example using point estimates

Assume an exotic weed is discovered in a central North Island lake. On investigation it is found that if nothing is done the incursion is likely to reduce the cover of indigenous meadow grass by 20%, and one species of native bird would no longer frequent the lake through loss of food supply. Further assume there are 100 households around the lake and 1,500 households in the district that would be affected by this change to the environment.

An initial decision is required on a response strategy, together with an estimate of the possible benefits associated with alternative responses.

There is insufficient time to conduct a primary study; hence any analysis must draw on information from previous work. The BVD has estimates of the mean WTP values for both these environmental attributes (see Table 6.1).

Scientific and biosecurity advice is taken and it is decided that these values can be transferred without adjustment to the policy site. In addition, comparisons of the SDCs at the study site (Lake Rotoroa) are similar so that no adjustment needs to be made for differences in the populations between the study and policy sites in this case.

Table 6-1 WTP estimates from the BVD ($/HH p.a.)

Lake Rotoroa	Local	District
Charophytes (1% Δ)	12	11
Birds (1 spp)	61	38

The annual benefit (CS) of preventing the impact on the environment is therefore the sum of the local and within district WTPs as per Equation 6.1.

$$CS = (\$12 * 20 + \$61 * 1) * 100\ HH + (\$11 * 20 + \$38 * 1) * (1{,}500 - 100)HH \qquad (6.1)$$

$$= (\$301) * 100 + (\$258) * 1400 = \$391{,}300$$

The PV of the sum of benefits over 5 years (the timeframe of the special tax) at a discount rate of 8% is

$$PV_{CS} = \$1{,}562{,}000 \qquad (6.2)$$

Adjustments could include factors to modify the biodiversity values if the site characteristics and or the population socio-demographic characteristics in the policy site were to vary significantly from the study site. If average HH income varies significantly between the study and policy sites then a simple way to adjust for this is to multiply the WTP estimate by the ratio of the HH income at the policy site to the HH income at the study site. An alternative to adjusting for differences in income is to adjust for differences in a characteristic such as membership of a conservation organisation. This information has been collected in the surveys. It can be checked against the national average of 8% (DOC, 2008). If there is a significant difference, then WTP could be weighted by the ratio of the national percentage to the sample percentage.

Note that the values in the BVD were estimated in surveys carried out in 2007 and 2008. The analysis carried out here is in 2009 dollars so the WTP estimates in the BVD should be indexed up into 2009 dollars using the Consumers Price Index (CPI). As time goes by this will become a more important issue, but for the above example inflation has not been taken into account. The cashflows for this analysis are set out in Table 6.2.

If values are transferred from another country then an adjustment should also be made for differences in purchasing power parity (World Bank, 2008).

Based on information provided by MAFBNZ, assume that eradication of the weed is likely to cost around \$500,000 and take one year.

From this information the expected Net Present Value (NPV) and Benefit to Cost ratio (B/C) of eradication is:

$$\text{NPV} = \$1{,}562{,}000 - \$500{,}000 = \mathbf{\$1{,}062{,}000} \tag{6.3}$$

$$\text{B/C} = \$1{,}562{,}000/\$500{,}000 = \mathbf{3.1} \tag{6.4}$$

As well as presenting the expected NPV and B/C ratio additional information would also be provided. This should include a sensitivity analysis on key variables including the discount rate (e.g. NPV @ 6% = \$1,648,000 and B/C = 3.3, NPV @ 10% = \$1,483,000 and

B/C = 3.0); and some "what if" analysis (e.g. what if eradication costs are double expected, in this example this would reduce the NPV to $562,000 and B/C to 1.6).

Based on this analysis and assuming there are no other benefits or costs the initial advice to decision makers would be to proceed with eradication. As the response is rolled out better information is likely to become available and this provides the opportunity to update the analysis.

6.3.2 Risk Simulation

The above analysis has been carried out using point estimates of WTP. Using point estimates overlooks the significant uncertainty around these figures. They were generated from relatively small samples of people from the community who, as individuals, expressed varying degrees of WTP. The BVD captures this through estimates of the standard deviation and upper and lower limits given 5% confidence i.e. there is a 5% chance the best guess will be outside the range.

Ignoring uncertainty assumes away important information that is relevant to the decision, this is, the variability inherent in the estimates of biodiversity values and the risk that the point estimate is misleading.

Uncertainty should be explicitly addressed using risk simulation by applying the following standardised approach developed by Nimmo-Bell called QuRA™ (Quantitative Risk Analysis). This utilises the Excel add-in @RISK to generate distributions of key risky variables and incorporates these into a distribution of the NPV of the project (Bell, 2000). The process is undertaken as follows.

Once the analysis has been carried out using point estimates, a sensitivity analysis will show which of the key variables contribute most to the NPV - these will be the variables that cause the NPV to change most when changed by a set percentage e.g. 5%. Once the variables that contribute most to the NPV have been identified the next step is to select which of these have the most uncertainty by comparing the coefficients of variation (standard deviation divided by the mean) for each variable. Usually there will be three or four key variables that meet the above criteria (high impact on the NPV and high uncertainty).

Table 6-2 A hypothetical example of benefit transfer

Year			(PV)	0	1	2	3	4	5
Discount rate		8%							
Benefits									
Local HH		100							
	WTP	No.							
Char	12	20	$95,825	0	24,000	24,000	24,000	24,000	24,000
Birds	61	1	$24,356	0	6,100	6,100	6,100	6,100	6,100
Sub-total			$120,181						
District HH		1400							
Char	11	20	$1,229,755	0	308,000	308,000	308,000	308,000	308,000
Birds	38	1	$212,412	0	53,200	53,200	53,200	53,200	53,200
Sub-total			$1,442,167						
Total Benefits			$1,562,347						
Costs			$500,000	500,000					
NPV			**$1,062,347**						
B/C			**3.1**						
Input cells									
Sensitivity Analysis									
Discount rate	10%	NPV	$983,335						
		B/C	3.0						
	6%	NPV	$1,148,298						
		B/C	3.3						
What if Analysis									
Cost double		NPV	$562,347.0						
		B/C	1.6						

Note: Estimates of mean values – assuming point estimates

In the example, it is clear that the WTP estimates meet the criteria of high impact on the NPV and high levels of uncertainty. Consulting the BVD provides the following estimates of the standard deviations (SD) associated with the mean values (see Table 6.3).

Table 6-3 Example WTP estimates – means and standard deviations ($/HH)

Lake Rotoroa	Local sample		District sample	
	Mean	SD	Mean	SD
Charophytes (1% Δ)	12	2	11	2
Birds (1 spp)	61	14	38	7

When combining estimates to determine their overall uncertainty the relationships between the uncertain estimates should be taken into account using the correlation coefficients. These are derived from the VARB matrix which is an output of the RPL model. VARB is the variance-covariance matrix of the attributes that have been specified as random parameter distributions. In this case, all the biodiversity attributes have been specified as constrained triangular distributions with the mean equal to the spread, notated as (t,1). The correlation coefficients $\left(\tau_{ij}\right)$ can be estimated as

$$\tau_{12} = \sigma_{12}^2 / (\sqrt{\sigma_{11}^2} * \sqrt{\sigma_{22}^2}) \qquad (6.5)$$

This information has been estimated for the 36 variables in the BVD and is contained in Appendix 4. For Lake Rotoroa there is a low level of positive correlation between Charophytes and Birds with a correlation coefficient of 0.2 for both local and district populations (see Table 6.4 and Appendix 4). We assume the distributions and correlations will be the same for the policy site and transfer them directly.

Table 6-4 Correlation matrix: Charophytes / Birds

	Charophytes	Birds
Charophytes	1.0	0.2
Birds	0.2	1.0

Using @RISK, the probability distribution of the NPV can be estimated. This is done by replacing the point estimates of the key uncertain variables in the cashflows with the means and standard deviations and specifying the correlation coefficients between the uncertain variables to ensure the dependency relationships are taken into account. @RISK will then simulate the uncertainty, by drawing from the distributions, usually over 5,000 or more iterations and combine the results into a distribution of the NPV.

The cashflows and output for the risk simulation are shown on Table 6.5. Note that the NPV has remained the same as in Table 6.2. This is because the distributions of the environmental values are symmetrical around the mean.

The additional information now available to decision makers from risk simulation as compared with point estimate analysis is contained in the chart of the NPV (Figure 6.1). This shows that after taking into account the uncertainty in the WTP estimates, the NPV of the response has a 100% chance of being positive. Also there is a 90% chance the value will be between $0.65 million and $1.46 million.

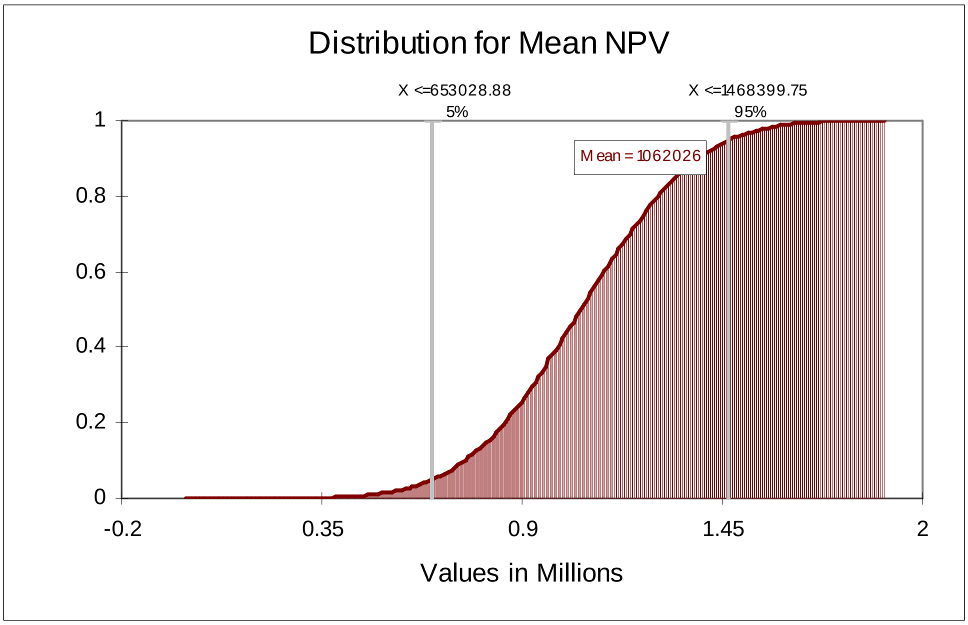
Table 6-5 Hypothetical example using risk simulation

Year						(PV)	0	1	2	3	4	5
Discount rate			8%									
Benefits												
	Local HH		100									
			WTP	SD	No.							
		Char	12	2	20	$95,825	0	24000	24000	24000	24000	24000
		Birds	61	14	1	$24,356	0	6100	6100	6100	6100	6100
		Sub-total				$120,181						
	District HH		1400									
		Char	11	2	20	$1,229,755	0	308000	308000	308000	308000	308000
		Birds	38	7	1	$212,412	0	53200	53200	53200	53200	53200
		Sub-total				$1,442,167						
	Correlation Coefficient		0.2									
	Total Benefits					$1,562,347						
Costs						$500,000	500,000					
NPV	Mean					**$1,062,347**						
	Confidence limits			5%		$653,000						
				95%		$1,468,000						
B/C						**3.1**						
Input cells												

Note: assumes the WTP estimates are normally distributed

By reference to the cumulative probability distribution of the NPV (Figure 6.1), it is possible to estimate the likelihood of any value being exceeded. The expected NPV is $1.06 million. This gives a much more realistic view of the likely outcome of the response. As the chances of the response being positive are very high this gives decision makers confidence to proceed with the response.

Figure 6-1 Probability distribution of NPV ($ million)



Experience with using different types of distribution (other than normal) to describe uncertain variables has shown that often the distribution of the NPV changes little. However, assumptions about correlation can have a significant impact. Table 6.6 shows the impact on the NPV of different assumptions about the correlation between Char and Birds. With a correlation coefficient of 0.2 the range of the NPV (5% and 95% confidence) is $0.65m - $1.47m. If the correlation coefficient is assumed to be -1 then the range narrows to $0.74 -

$1.38m. On the other hand, if the correlation is assumed to be +1 then the range widens to $0.59m - $1.53m. This confirms the expectation that negative correlation reduces the range of the NPV, while positive correlation increases the range.

Table 6-6 Impact of correlation assumption on the NPV range

Correlation coefficient	NPVs ($m) at the 5% & 95% limits	
	x<= 5%	x<=95%
-1	0.74	1.38
0.2	0.65	1.47
+1	0.59	1.53

Of course in real life there are other risks and uncertainties that need to be taken into account when assessing the economics of a response. Also, decisions are much more complex and may be much less clear cut, but the example shows how the database could be used to aid in making quick high level decisions.

6.4 Cost Benefit Analysis

At the investigation stage it is likely that enough information will be available to assess whether the response will proceed to the development of a business case requiring Cabinet approval. It is also likely that enough information will be available to assess whether environmental and social benefits will be material to the decision on a major response.

Having completed a high level benefit assessment using benefit transfer values, it is time to assess whether or not a primary non-market valuation survey and analysis using choice modelling should be undertaken. Factors that would be taken into account include whether the quantum of non-market benefits would influence the decision on the response, whether the study site and the policy site (from both environmental and the socio-demographic points of view) are similar enough to have confidence in the transfer values, and whether there is enough time to conduct a primary study. If a primary survey is to be undertaken then the case studies demonstrate the process.

Such decisions require value judgements by management as to how representative the values are for the site in question and how similar the characteristics are between the database population and the policy population.

If passive use values are likely to form a critical component of the decision on a response, then steps need to be taken early to obtain the information to estimate those values. Initially, utilising benefit transfer should provide approximate estimates. If these are not satisfactory then a primary choice modelling study needs to be initiated immediately recognising this would normally take 6 to 12 months. With urgency and cooperation between scientists, ecologists, biosecurity specialists, management and economists it may be possible in some cases to reduce this to 4 to 5 months, or even three months.

6.5 Pest management

Once a weed or pest has become established ongoing decisions are required on appropriate management. A pest of national significance requires the development of a National Pest

Management Strategy and a pest of regional significance, requires the development of a Regional Pest Management Strategy. Both such strategies require a CBA to be undertaken.

As with responses to incursions, currently analysis of pest management programmes usually focus on quantifying active use values i.e. those benefits pertaining to industry that can be quantified using market prices. The methodology outlined in this chapter is equally applicable to decisions on pest management where passive use values are important to such decisions.

6.6 Incorporating biodiversity values into the DSS

The key steps required to incorporate biodiversity values into the MAFBNZ DSS are summarised as follows:

Stage	Action
1.	Assess the biodiversity values at risk from the incursion
2.	Refer to the database for relevant biodiversity values
3.	Estimate the benefits of a response using Compensating Surplus by taking the sum of WTP values weighted by the quantitative change of each multiplied by the relevant number of households affected
4.	Discount the future compensating surplus values to obtain a PV of benefits
5.	Estimate the PV of the cost of a response that relates to the benefits
6.	Subtract the PV of the costs from the PV of the benefits to obtain the NPV
7.	Divide the PV of the benefits by the PV of the costs to obtain

Stage	Action
	the Benefit/Cost ratio
8.	Conduct a sensitivity analysis on key variables including the discount rate
9.	Conduct 'what if' analysis on key sensitive variables
10.	Carry out risk simulation using QuRA™ to highlight the uncertainty in the estimate of the NPV and likelihood of a positive NPV
11.	Provide initial recommendations on the appropriate response
12.	Revise estimates as new information comes to hand
13.	If there are no relevant values in the database assess whether a primary study is required
14.	If the answer to the previous point is yes, and there is time and money then conduct a primary study.

A key output of this research is a manual for MAFBNZ which integrates and synthesises all the elements that have been brought together in this thesis in a practical way for implementation (Bell, Cudby, and Yap, 2009c).

The final chapter evaluates and critiques the additions to knowledge and innovation that this thesis has contributed.

Chapter 7 : Evaluation, critique, conclusions

7.1 Evaluation

7.1.1 Biodiversity values

Some of the most pressing problems facing New Zealand are at the margin between the economy and the environment. Once basic needs have been met, other aspects of well being such as the environment assume an increasing importance, examples include climate change, water quality in lakes and waterways and loss of native plants and animals due to pests and diseases. Valuing biodiversity is a critical step in reaching sound community decisions about the use of environmental resources.

The techniques developed in this thesis allow community values to be incorporated in an analysis of the threat of an invasive species.

7.1.2 Tools

Among the stated preference techniques, choice modelling is preferred over contingent valuation because it allows the multiple attributes of an ecosystem to be valued together and enables analysts to place dollar values on biodiversity changes. As a result there is the ability to make direct comparisons between benefits and costs in the economy and the environment.

Deliberative methods and mediated modelling are intuitively appealing, but in New Zealand constraints on budgets and time mean that they will have much more limited application than the decision support system that builds on the existing systems.

In this thesis a number of recent advances in the CM tool have been incorporated for the estimation of indigenous biodiversity values. The panel version of the random parameters logit model enables the expression of heterogeneous individual tastes. This has significantly increased the explanatory power of the model over the multinomial logit model.

7.1.3 Hybrid surveys

The community surveys have utilised advances in experimental design using Bayesian methods to incorporate prior information improving efficiency and thus reducing sample size while retaining high levels of significance and minimal standard error.

A hybrid community survey methodology has been developed and refined. It incorporates elements of personal interview face-to-face one-on-one surveys, but with major savings in time and cost. By bringing together groups of around 50 respondents, time and cost savings of the order of ten fold can be made while retaining high quality information. Personal surveys allow the surveyor to convey complex information to respondents and thus obtain high quality information in return. The survey purpose, biodiversity and biosecurity issues, and instructions can all be conveyed at one sitting. In addition, at the end of the session, each questionnaire can be checked and omissions or errors rectified thus eliminating issues to do with partially completed questionnaires.

These hybrid surveys are not based on random sampling so cannot be taken as statistically representative of the whole community. However, endeavours are made to assemble a cross section of the community with respect to gender balance, age, education, incomes, professions, ethnicity plus affiliation to a conservation organisation. Socio-demographic information is collected from them and this can be calibrated against census information to test representativeness. At the benefit transfer stage, adjustments can be made to WTP values based on differences between the characteristics of respondents and the policy population.

The WTP values obtained from this process are indicative of an informed group of people from the community similar to that which would attend a community meeting held by MAFBNZ during an actual incursion. The values obtained by this method are community estimates with associated variation around the estimate.

7.1.4 Application of biodiversity values to biosecurity decisions

There are two key points in the response process when changes to biodiversity values can aid decision making. The first is the investigation phase during the early stage of a response when basic information on the pest and its likely impact is being collected. The second is later when formalising the Cost Benefit Analysis - when response options are analysed in detail.

An early assessment of biodiversity values at risk utilising benefit transfer techniques can greatly assist the decision on whether to undertake a primary non-market valuation study for the CBA. If an early decision is made to undertake a primary study there is likely to be enough time before a decision is required on a response for the results of the study to be incorporated into the CBA provided that the analysis is given urgency and priority.

Routine pest management decisions can also utilise biodiversity values. Whether benefit transfer values or a full primary study is utilised will depend on the applicability of the values in the database to the new situation and the importance of the pest.

7.1.5 Building a database of values

Most primary non-market valuation studies have been undertaken without a thought to benefit transfer. As a consequence there are usually multiple deficiencies that limit their use in other situations. At the start of this research, a literature search found no studies of indigenous biodiversity values in the biosecurity space. For this reason this study has involved the development of a systematic biodiversity valuation database specifically for biosecurity purposes. This is the first of its kind, world wide, with features not present in the international environmental values database (EVRI) or Lincoln's New Zealand specific environmental valuation database.

The existing databases (EVRI and Lincoln) record basic information about individual unrelated studies including directions as to where the detailed valuation information can be sourced, which is usually in journals or working papers.

The biodiversity valuation database (BVD) has been developed specifically to provide biodiversity values as input into CBA on responses to incursions of exotic pests and diseases. Unlike information held in existing databases, the values were generated using a consistent approach and methodology so that the values would be comparable. The estimates cover four different high impact pests in four diverse high at-risk ecosystems. Sample estimates were obtained from local, within district, within region and out of region populations so that the effect on WTP values of distance from the hypothetical incursion could be gauged. This is important when considering the level of extrapolation that is applicable for specific biodiversity values in the CBA.

When the biodiversity values held in the database are not relevant to a biosecurity response, consideration should be given to undertaking a primary study. As the number of primary studies builds up the relevance of the database will increase. This is the start of a process that could in the future result in a comprehensive database of comparable biodiversity values for input in CBA.

To ensure that new non-market valuation studies will be useful for benefit transfer there are a number of key requirements that need to be met. Firstly, the process should be thoroughly documented. This should include details of the valuation methodology including the frame (scale and scope), and experimental design. Secondly, the primary data should be made accessible including basic information on the site, pest and ecology. Thirdly, the questionnaire should be made available including: the choice questions; socio-demographic characteristics of respondents; supporting questions on beliefs and attitudes; and responses. Fourthly, the spreadsheet recording respondent answers to choice questions and the SDCs of each respondent should be available. Fifth and lastly, the results should be documented including WTP mean values, standard deviations and correlations, MRS plus discussion on interpretation.

7.1.6 Transferring values

Unlike changes to the economy where market prices guide values, changes to the environment require indirect survey methods incorporating non-market values that can be estimated through stated preferences. The WTP concept is the equivalent of the market price. It is real, but cannot be estimated directly and therefore the context from which it is derived is important. Chapter 5 highlighted the key issues, such as framing effects that need to be taken into account when interpreting and transferring WTP values. Transferring values from one site to another or one environmental attribute to another can lead to significant errors. The guiding principle for greater reliability and validity of benefit transfer is "more similar is better", for both the attributes being transferred and the populations as well.

Improved modelling has generated a rich vein of information about valuation and values. Such improvements include: incorporating into the models the multiple attributes of the environment and SDCs of the people in the survey samples using choice modelling; the use of Bayesian experimental design to reduce sample sizes while retaining the explanatory power of the models by incorporating "prior" knowledge about environmental attributes; and taking into account the individual preferences of people taking part in surveys (using the panel RPL model) rather than assuming they are all alike.

Almost all cost benefit analyses that have been undertaken up till now have used point estimates to generate NPVs, but the RPL model generates standard deviations and correlations, in addition to mean values. When these are incorporated into the analysis through risk simulation, richer information about the uncertainty surrounding key input variables is conveyed to decision makers.

By incorporating these improvements into stated preference techniques of environmental valuation the process of transferring values will be greatly improved. The more robust and transparent primary surveys are the greater the opportunity there will be to adjust their outcomes for benefit transfer to new situations. This thesis has utilised these advances for the valuation of indigenous biodiversity and demonstrated how they can be incorporated into decision making on biosecurity response.

7.1.7 Analysis of Risk and Uncertainty

Up till now, the estimates MAFBNZ has incorporated into the analysis of costs and benefits of a response have been largely based on market prices of active use values. For example, the benefits of protecting the Kauri tree from die-back disease were quantified by estimating the market value of the timber. Point estimates were used in the analysis to derive the Net Present Value with sensitivity analysis and "what if" analysis used to highlight the possible variability in the results. Unfortunately this approach gives no indication of the likelihood of a positive Net Present Value or its likely range.

In this thesis a method has been introduced to incorporate passive use values into the analysis and also how to highlight the uncertainty that is inherent in these and other risky variables in a CBA by applying risk simulation using QuRA™. This is the first time that the uncertainty in passive use values has been incorporated into an analysis. It is a major step forward in improving the information available to decision makers.

7.1.8 Total Economic Value and marginal change

This thesis has focused on incorporating biodiversity values into the cost benefit analysis of response. The biodiversity values estimated fall into the category of passive use values. Not considered are the Indirect Use Values and Option Values, which include, for example, valuing the recreation and ecosystem services provided by biodiversity. While often these elements are not considered, they may be potentially important.

Ecologists and conservation enthusiasts are often concerned about attempts by economists to place dollar values on whole ecosystems. When it is explained that economic analysis is concerned primarily with valuing marginal changes, much of the objection dissipates. It is quite a different thing to estimate the value of a change to an ecosystem in contrast to valuing the whole system. Cost benefit analysis attempts to quantify the "with" minus the "without" project benefits and costs.

Choice modelling attempts to isolate the key attributes of an ecosystem containing indigenous biodiversity that are most important to people, and to value changes to these attributes. As with all models this is an abstraction from reality, which cannot capture the full complex dynamic nature of ecosystems.

7.1.9 Wider Application

The estimates of biodiversity values developed here have application beyond biosecurity. Any resource allocation question that requires biodiversity values can utilise the values and the methodology outlined above. For example, the values of indigenous biodiversity are likely to be of interest to the Ministry for the Environment, Department of Conservation (DoC) and regional councils.

7.1.10 Synthesis

The contribution of this thesis to knowledge and innovation is the integration and synthesis of a number of independent economic strands into a coherent and practical manual for MAFBNZ personnel to incorporate biodiversity values into biosecurity decisions on response. The challenge was to extend the current decision support system by developing a theoretically robust component for incorporating biodiversity values into CBA on response that could be implemented by MAFBNZ economists and biosecurity specialists. This has been achieved by integrating recent advances in choice modelling including Bayesian methods in experimental design with innovation in surveying through the hybrid method, utilising the panel random parameters logit model, conducting four case studies to populate a systematic biodiversity valuation database of comparable values, demonstrating how these values can be transferred to new situations through direct univariate transfer and utilising risk simulation to quantify for decision makers the uncertainty in key variables generating likelihood values in the CBA of response options. The total package has application beyond biosecurity to wherever biodiversity values are required for natural resource management.

Choice modelling, benefit transfer and risk simulation provide a way of incorporating biodiversity values into CBA that is quick and relatively simple. Concerns about bias particularly in aggregating WTP values can be reduced by making adjustments to transferred values and by decision makers applying judgement and common sense to the level of aggregation that is relevant.

7.2 Critique

Non-market valuation is a rapidly advancing area of economics. This is in part a reflection of changes in the values society holds. Increasingly greater store is placed on the aspects of well-being that are not traded in markets. Concerns have risen over human induced climate change, intensification of pasture based livestock systems causing nutrient and effluent contamination of lakes and water ways, and increasing risk to indigenous biodiversity from exotic pests and diseases. Decision makers are demanding better analysis of these, and other stresses on the environment.

Until now MAFBNZ has relied on quantifying the benefits to New Zealand's agriculture, forestry and fishing industries to justify spending public money on response programmes. Under moves recently negotiated with industry organisations to share the cost of response that benefits them, justification for spending public money is shifting to a focus on public benefits. Among these, protecting indigenous biodiversity is paramount. This thesis makes accessible the tools to quantify such benefits and a decision support system to integrate the tools into decision making. By quantifying biodiversity benefits in the same metric as industry benefits, a more comprehensive and balanced analysis is obtained.

While distribution issues are often put aside by economists, equity and fairness are important considerations in the development of policies whose successful implementation will need the support of a wide range of stakeholders. The cost sharing initiative where industry pays for a share of the cost of a response is much more likely to be supported by industry if they perceive that the public is carrying its share of the burden as well. Quantifying the public benefits in a way that is comparable with industry benefits will make transparent the rationale for cost sharing and should lead to more sustainable policies.

7.3 Conclusions

The primary objective of this thesis has been to develop a system for quantifying and incorporating non-market values into cost benefit analyses of response options to exotic pest and disease incursions. The thesis has achieved this objective.

The biodiversity values and the process of incorporating them into decisions through the decision support system have much wider application. Any decision requiring estimates of the value of indigenous biodiversity when allocating resources provides the potential for application of the tools developed in this thesis.

This work represents an initial step; having set a foundation there is much work to do to realise the full benefits. More primary choice modelling surveys are needed to build up and enrich the Biodiversity Valuation Database. Such studies should broaden the range of ecosystems and biodiversity values in them. The DSS needs to be implemented by MAFBNZ so that analysts can refine and further develop the processes of transferring values. Decision makers need to be educated into the interpretation of the outputs of the analysis and the value such analysis can add to decisions. They need to understand the limitations as well as the benefits.

There are many pressing issues that confront society where the techniques brought together here can add value to decision making. Better analyses mean more informed decisions, which in turn lead to improved well-being for all.

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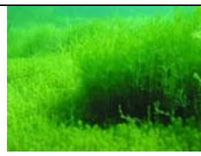
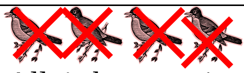
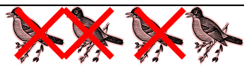
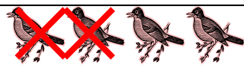
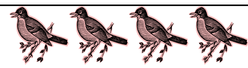
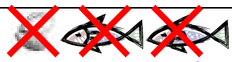
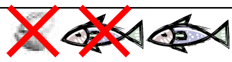
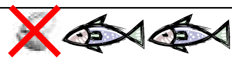
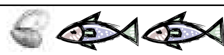
Appendix 1 Freshwater case study

1.1 Experimental design

Choice situation	alt1.price	alt1.hydr	alt1.wqual	alt1.char	alt1.birds	alt1.fish	alt2.price	alt2.hydr	alt2.wqual	alt2.char	alt2.birds	alt2.fish
5	40	0	2	1	3	3	20	3	1	3	0	0
13	20	3	0	0	1	1	40	1	3	0	1	2
15	160	1	3	0	1	1	160	0	1	1	0	0
23	10	1	0	1	3	1	80	3	3	2	0	0
28	0	0	2	1	0	2	160	1	1	1	3	2
37	0	3	2	0	3	1	160	1	1	3	1	3
39	80	2	3	2	2	0	0	1	0	1	2	2
43	40	2	3	0	1	1	20	0	0	0	2	3
44	40	1	2	2	3	0	10	3	1	2	0	3
46	20	2	3	2	1	0	40	0	0	3	2	3
59	80	1	1	2	2	0	10	2	2	0	1	3
60	0	2	3	1	0	0	160	2	0	1	3	2
2	80	0	1	3	2	2	10	3	1	1	1	2
6	80	3	1	0	1	2	10	0	2	3	2	1
7	40	0	2	3	0	3	10	3	1	2	3	0
8	20	3	0	1	1	0	40	1	3	0	2	2
10	40	3	1	1	1	3	20	0	2	3	1	0
14	10	1	0	0	0	3	80	2	3	0	3	1
19	0	0	0	0	2	1	160	1	3	1	2	1
21	80	1	0	3	3	2	10	1	3	2	0	2
27	160	3	3	2	2	2	0	1	0	2	1	2
35	40	0	2	3	3	0	20	3	1	0	0	3
52	40	1	1	3	0	3	20	3	2	3	3	0
58	160	2	1	1	2	3	0	1	2	1	1	0
1	0	1	2	3	0	3	80	2	1	3	3	1
4	160	3	1	3	2	1	0	0	2	2	1	2
17	20	3	1	3	1	0	40	0	2	1	2	3
20	160	3	2	1	0	3	0	2	1	1	2	0
22	80	1	3	2	2	3	10	3	0	3	1	0
24	0	0	3	2	0	2	80	1	0	3	3	0
26	160	2	0	1	1	2	0	0	3	0	1	1
30	0	2	0	2	1	1	160	3	2	0	3	3
36	160	1	1	2	3	3	0	2	2	2	0	0
41	80	2	3	3	0	2	10	0	0	3	3	1
42	40	0	1	3	0	0	20	2	2	0	3	3
54	20	1	0	1	2	2	20	1	3	0	2	2
11	10	0	2	2	1	3	80	2	2	2	2	0
16	80	3	2	0	3	1	20	0	1	3	0	3
18	20	0	1	3	3	0	40	2	2	0	0	3
25	20	0	3	2	2	2	40	3	0	2	0	1
31	20	0	2	1	3	1	40	3	0	2	0	2
33	10	2	0	1	0	3	80	1	3	1	3	0
45	0	2	0	0	1	2	160	3	3	0	3	1
47	80	1	3	2	3	1	10	2	0	3	0	1
48	10	2	0	1	2	1	80	2	3	1	2	1
50	10	3	0	2	1	1	80	2	3	3	2	1
53	0	1	0	0	3	1	160	1	3	2	0	2
56	40	2	0	2	2	3	20	2	3	2	1	0
3	20	1	2	0	3	1	80	1	1	1	0	1
9	160	2	3	2	2	2	0	3	0	1	1	1
12	10	0	3	0	2	2	80	2	0	0	1	3
29	10	3	1	0	1	0	160	0	3	1	2	3
32	160	2	1	3	3	2	0	0	2	3	1	2
34	40	3	1	0	0	0	20	0	2	0	3	2
38	10	0	2	3	2	0	40	3	1	0	1	3
40	10	0	1	3	3	3	40	3	2	3	0	0
49	0	3	3	0	0	0	160	2	0	2	2	2
51	160	2	2	1	1	3	0	0	1	1	2	1
55	80	3	2	3	0	0	10	0	1	2	3	3
57	20	1	3	1	0	2	40	1	0	2	3	1

Note: see Table 4-1 p.87 for the attribute descriptions (hydr etc) and p.90 and Figure 4-1 p.91 for a description of the alternatives (alt1 and alt2)

1.2 Coding master table

	Level 0 (status quo)	Level 1	Level 2	Level 3
Hydrilla	 100% coverage	 65% coverage	 30% coverage	No hydrilla
Water quality and clarity	Significantly worse than now	Moderately worse than now	Slightly worse than now	OK Same as now
Native submerged plants	Eliminated from lake	 Reduced to 7% cover	 Reduced to 14% cover	 Same as now at 21% cover
Native birds	 All 4 shag species do not visit the lake anymore	 3 shag species do not visit the lake anymore	 2 shag species do not visit the lake anymore	 All 4 shag species happy to visit the lake
Mussels and native fish	 Mussels and 2 fish species disappear from the lake	 Mussels and 1 species of fish disappear from the lake	 Mussels disappear from the lake	 Mussels and all fish species remain in the lake

	Level 0 (status quo)	Level 1	Level 2	Level 3	Level 4	Level 5
Cost to your household each year for five years	$0	$10	$20	$40	$80	$160

Appendix 1.3 Flyer invitation for survey participants

June 2008

Research on a freshwater ecosystem affecting you

You are invited to take part in an evening aimed at finding out the values you place on a local freshwater environment. Your involvement will assist your community through a donation to the Waikato Waka Ama and Dragon Boating Association who are arranging the meeting.

New Zealand is continually exposed to foreign pests and diseases that can potentially threaten our freshwater systems. Some of the values threatened are difficult to place monetary values on as there are no market prices. Nevertheless these nonmarket values are important to people. Placing values on them will help Biosecurity New Zealand provide more informed advice to the government than is currently possible.

The evening will take the form of a group exercise where information will be given out prior to a series of questions. These questions will relate to values of freshwater including potential changes to what you see and experience, loss of species including plants, fish and birds and the value to you of changes to the environment.

No particular qualifications or experience are necessary. We are interested in your personal views. The information you provide will be kept confidential, will not be disclosed in any way that will identify you and you may withdraw at any time. No preparation is necessary just come along.

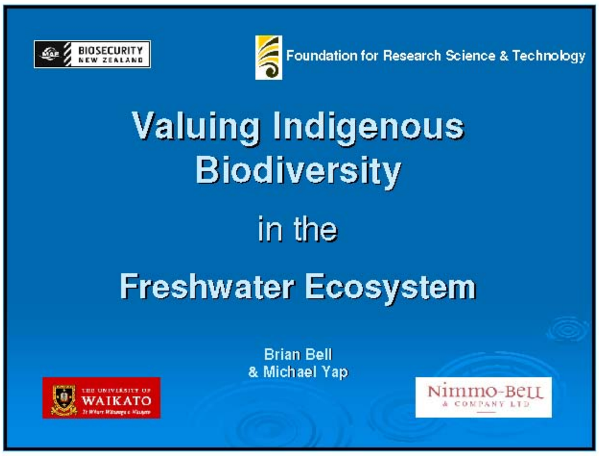
Supper will be served at the end of the session. In addition to the donation to the Association, we will give you a $20 voucher as a way of a personal thank you.

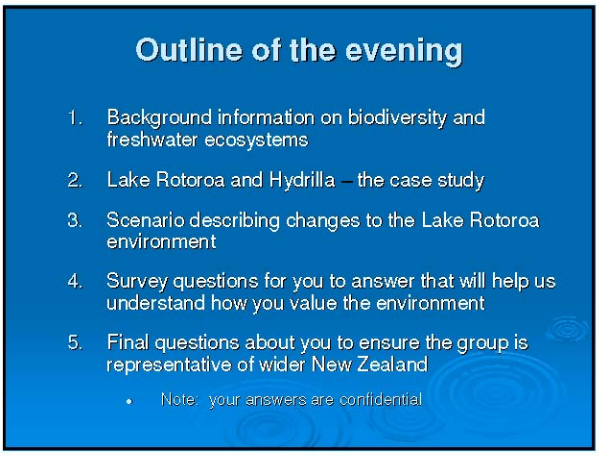
We can guarantee an interesting evening and look forward to seeing you. If you would like further information please contact me.

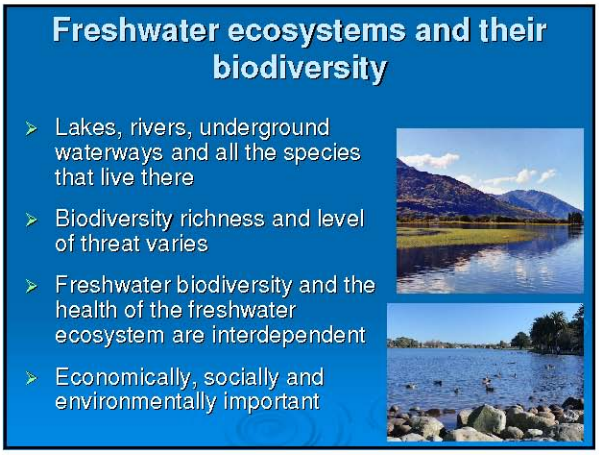
Date:	1 July 2008
Time:	<insert time>, followed by supper
Place:	<insert location; school hall for example>.

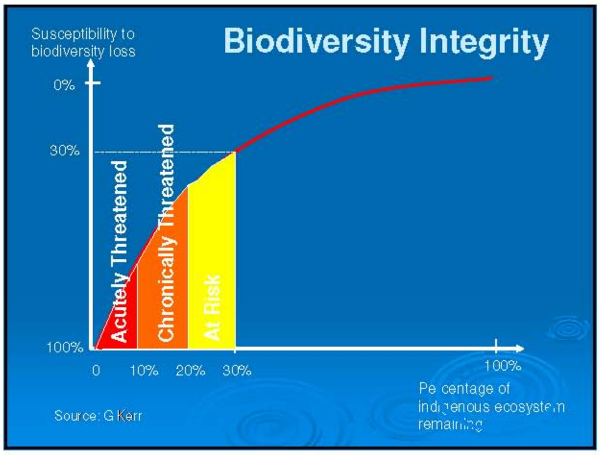
Brian Bell
Research leader
Phone: 04 472 4629
Email: brian@nimmo-bell.co.nz

Appendix 1.4 – Presentation to respondent groups









Lakes

Source: Nathan 2007

- New Zealand has very diverse lake types
- Water can stay in lake basins for very long periods, making lakes vulnerable to human activities
- Lake quality is affected by many interconnected factors
  - land use in a catchment
  - stability of lake sediment
  - the composition of submerged flora and fauna

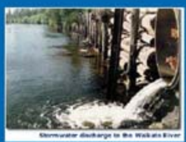
Threats to lake biodiversity

- Many freshwater lake species have a significantly reduced distribution
- The ecosystems they depend on are severely under threat
- New Zealand has internationally recognised communities of submerged plant species under threat
- There are very few totally indigenous lakes left, and many are ecologically degraded

Source: MfE 2006
(monitored lakes only)

Threats to lake biodiversity

- **Habitat loss and degradation**
  - Urban pressures
  - Rural land use
  - Lack of vegetation along water ways
- **Pests**
  - Causing further habitat loss, and displacement of native species



Stormwater discharge to the Waikato River

Freshwater ecosystem protection

- Lowland freshwater ecosystems are least protected
- Not as much conservation effort in freshwater compared to other ecosystems
- Submerged invasive plants have caused considerable damage
- The spread of invasives depends on the level of access
- Preventing ecosystem damage cheaper than repairing damage afterwards
- There are still sizable gaps in our knowledge

Freshwater pest management

- Many organisations and groups are involved
- Who is involved and what they do depends on the particular pest, its impacts and the options to control it

www.weedbusters.org.nz

Our research aims to find out the values <u>you</u> place on changes to freshwater indigenous biodiversity

Lake Rotoroa
(also known as Hamilton Lake)

Hydrilla

(hydrilla verticillata)
An invasive weed that grows underwater

One of the worlds' worst because:

- It is very difficult to get rid of – spread by dormant compressed shoots, chemicals not very effective
- It can spread quickly among a variety of habitats
- It excludes other plant species
- It can cause significant economic, social and environmental damage

Hydrilla's threat

- Present in three Hawkes Bay lakes; activity controls have prevented its spread
- Not present in Lake Rotoroa
- High risk weed species
- It spreads by stem cuttings, and uniquely by dormant compressed shoots (persist for years and survive adverse conditions)
- Boating and fishing activities are the key risk pathway

Hydrilla

Potential impacts in New Zealand

- Displacement and loss of biodiversity
- Reduced water quality and clarity
- Blocked irrigation systems and hydroelectric power generation equipment
- Reduced fish availability and damaged fishing equipment
- Restricted recreation (boating, rowing, swimming)
- Radical change to the scenic value of lakes, affecting tourism
- Loss of taonga species for maori, and loss of / restrictions on mahinga kai
- Increased flooding and erosion risk leading to loss of / damage to infrastructure
- Increased costs of surveillance and control

Hydrilla

The impact on a freshwater ecosystem can be very prolific

Scenario

We will now outline a scenario which will show changes to the Lake Rotoroa environment from a hypothetical invasion of hydrilla

You will then be presented with 12 sets of questions about the scenario which will help us understand how you value the environment

Hydrilla

(exotic underwater weed)

- Currently there is no hydrilla in Lake Rotoroa
- There are other weeds – mainly Egeria (an oxygen weed). This is under control at present.
- If hydrilla invaded it could potentially overcome all the vegetation in the lake (i.e. 100% cover)
- Lake Rotoroa is a high use lake – if it established there, the risk of hydrilla spreading to other lakes in New Zealand would be significantly increased

Charophytes

(native underwater plants)

- Currently 21% lake bed cover
- They absorb nutrients and help to stabilise bottom sediments
- Charophytes are internationally recognised as a species group under threat
- A hydrilla invasion will reduce or even eliminate the Charophytes from Lake Rotoroa
- If destroyed, Charophytes can recover from seed banks, but this would take many years
- A sustained hydrilla invasion may degrade the Charophyte seed banks eventually preventing recovery

Charopytes vs. Hydrilla

Thanks go to NIWA for the photos

Water quality and clarity

- Lake water quality and clarity are critical for life-supporting capacity
- Lake Rotoroa is high in nutrients and algae
- Water quality is improving, but not suitable for drinking and swimming is discouraged (not banned)
- Lake flora and fauna can survive happily
- If hydrilla invaded water quality and clarity in the lake may deteriorate – e.g. by
  - Displacing the charophytes which stabilise bottom sediments
  - Exclude light from the lake due to its dense growth

Native shags

- 90% of lake birds are exotic mallard ducks
- Four species of native shag (1.5% of total birds)
- Shags encouraged by Council because they feed on pest fish
- Protected by law, though they are common throughout NZ
- If hydrilla invaded, shags would not visit the lake as frequently, because:
  - The clear water they need for diving in and seeing their prey (fish) would be reduced
  - Available food (fish) in the lake would be reduced

Native fish and mussels

- Four native fish species
  - long and short finned eels – these can tolerate weedy environments; therefore hydrilla would not affect them
  - common bully and common smelt
  - None of these species are threatened nationally
- Native freshwater mussels were part of habitat before humans, and reintroduced in 2001. Mussels are rare in the Waikato lakes.
- If hydrilla invaded, the mussels, bullies and smelt would be affected by:
  - competition for space
  - deterioration of the sediment
- There are also six exotic fish species in the lake

Survey questions

We will now explore
your willingness to pay
to avoid certain impacts that hydrilla may have
on Lake Rotoroa

You will now be presented with 12 sets of choices to complete

Please answer every question

Sample choice question

These dollar amounts represent what YOU are willing to pay

Question 1:

Options A, B and C
Please choose the option you prefer
By ticking ONE box

	Option A	Option B	Option C
Extent of hydrilla	100% coverage	30% coverage	No hydrilla
Water quality and clarity	Significant deterioration	OK Same as now	OK Same as now
Coverage of native submerged plants	Eliminated from lake	Eliminated from lake	Same as now at 21% cover
Number of native bird species	All 4 shag species do not visit the lake anymore	3 shag species do not visit the lake anymore	3 shag species do not visit the lake anymore
Fish and mussels	2 fish species and mussels disappear from the lake	Mussels disappear from the lake	1 species of fish and mussels disappear from the lake
Cost to your household each year for 5 years	$0	$160	$20
I would choose ☑	☐ A	☐ B	☐ C

'Illogical' choice questions

You may feel that some questions include an illogical mix of attributes

For example:

Question 16:

Options A, B and C
Please choose the option you prefer
By ticking ONE box

	Option A	Option B	Option C
Hydrilla	100% coverage	No hydrilla	100% coverage
Native submerged plants	Eliminated from lake	Eliminated from lake	Same as now at 21% cover
I would choose ☑	☐ A	☐ B	☐ C

Please answer all questions as best you can

Some final questions about you

To help us determine how well the returned questionnaires reflect people living in New Zealand

Your answers are confidential

Thank you

Appendix 1.5 Speech notes to accompany the Freshwater presentation

Welcome intro to the project and the team.

[2]First part of the evening (items 1 – 3) are for informing you of the issue. The second part of the evening is to test our survey before it 'goes live'… note that the survey content (i.e. the selected attributes of the environment and their levels) have been pre-tested using focus groups and technical experts.

[3] Ecosystems bounded by water catchments
Richness and level of threat varies - For example, wetlands contain a high proportion of biodiversity, and over 90% of New Zealand's original wetland habitat has been destroyed; there is accordingly a high level of threat to biodiversity. On the other hand, New Zealand is considered to have very few freshwater fish and plant species compared to other temperate land masses, but a very high proportion of these are endemic. Within this group, whole species groups are threatened (e.g. charophytes, bryophytes) both in New Zealand and even more so internationally – so protection efforts for these species are internationally significant.

[5] Few countries of comparable size have lakes of such diverse origins as New Zealand; there are over 775 lakes of various types all with vastly different characteristics, including those formed by glaciers, rivers, dunes, landslides, volcanoes, coastal barriers, and many others (including Waikato's peat lakes e.g. Lake Rotoroa)
Land-use in a catchment therefore affects the amount of water, nutrients, sediment and other contaminants that enter a lake. Nutrient increases can help algae growth which reduces water clarity and impacts of the availability of light throughout the lake affecting the flora and fauna that depend on that light. Oxygen levels can decline severely affecting the life supporting capacity of the lake. Other factors affecting lake water quality are numerous and include sediment being stirred up into the water (which results in further nutrient re-leases), and the composition of submerged plants and fish (e.g. displacement of native species by pests). All of these factors are interconnected, and one change can lead to multiple flow on effects in the ecosystem.

[6] New Zealand has internationally recognised communities of species under threat; charophytes and deep water bryophytes, many species in these broader groups and these are under threat from pests as well as declines in lake water clarity. These species groups are under threat internationally.
A 2005 review of the New Zealand Biodiversity Strategy (Green & Clarkson 2005) noted that overall there has been a serious decline in the quality of many freshwater ecosystems. Hardly any of our lakes are not affected in a

major way by pest fish or plants. All freshwater habitats still free from alien plants or fish can be regarded as endangered (MfE 2002a). Many lakes (especially those in lowland and coastal areas) have been ecologically degraded by pests, sediment / contaminent flows and altered water currents. Small, shallow lakes surrounded by farmland have the poorest water quality of all our lakes; many of these are degraded by eutrophication – some so degraded they are incapable of sustaining fish life

[7]Note that picture at bottom is of didymo

[8] The existing network of protected areas includes some freshwater bodies, but this is far from representative of the full range of freshwater ecosystems and habitats; lowland lakes and rivers, floodplain wetlands, mid-altitude wetlands, and geothermal systems are all poorly represented in terms of ecosystem protection
Compared with efforts in other ecosystems (e.g. marine) and in particular parts of the freshwater system (e.g. wetlands), the freshwater environment has received proportionately less biodiversity conservation effort in New Zealand over the last few years. The extent of invasive plant impacts in freshwater ecosystems has been undetectable to most people and relatively few resources have been allocated to eradication or control
There are still sizable gaps in our knowledge of aquatic species and of and the extent and condition of their habitats.

[9] Many organisations and groups are involved, ranging from central government, regional and local councils, industry, community groups and the general public.
Who is involved and what they do depends very much on the particular pest, its impacts and the options to control it.
Ultimately <u>you</u> fund pest management, whether through:
Central government via your taxes
Regional government via your regional rates
Local government via your city rates
Other organisations such as Fish and Game (a crown entity funded through hunting and fishing levies) or community groups (funded by donations and involving volunteer work).
That is why we are talking to you today. Our research aims to find out what you - as the ultimate beneficiary of pest management efforts – are willing to pay to deal with invasive freshwater pests that affect our indigenous biodiversity.

[10] Our research aims to find out what <u>you</u> are willing to pay to deal with invasive freshwater pests that affect our indigenous biodiversity
To do this we are using a case study involving:
Lake Rotoroa as the freshwater ecosystem
Hydrilla as the freshwater invasive weed

[12] The lake is located in Hamilton city, and is surrounded by parkland, residential housing and other urban structures (e.g. roads, hospital). The lake is one of 31 shallow, peat lakes concentrated around the Waikato and Waipa districts and Hamilton City. They represent the last of the formerly extensive peat bogs of the region and are a relatively unique type of lake in New Zealand (the earlier map of New Zealand's lake types does not show peat lakes, which are grouped with a range of other lake types under the 'other' category….)

[13] The lake is ringed with walkways that are used for walking, cycling and observing wildlife

[14] There are many lakeside recreational activities around the lake; facilities include a playground, green spaces, sports grounds, wildlife. Many community events are held around the lake Domain. People can also fish from the lake, and feeding the ducks is a popular activity.

[15] There is a variety of native and introduced wildlife in and around the lake;
Native = Shags (four species), Pukekos, bullies, smelt, eels, freshwater mussels, charophytes, raupo (bulrush)
Exotic = Mallard ducks, six exotic fish species including Rudd (pictured; coarse fish), oxygen weed (egeria), water lilies etc

[16] Recreational activities on the water are common and include yachting, dragon boating, waka ama (outrigger canoes), model boats, paddle boats, wind surfing and many others. The Lake Domain is the headquarters for the Hamilton Yacht Club (pictured bottom left).

[17] Hydrilla is a submerged, freshwater perennial plant that is characterised by prolific growth and tolerance of a wide range of freshwater habitats:
It is considered one of the world's worst aquatic weeds because of its persistence and ability to spread quickly among a variety of aquatic habitats, and exclude other plant species. Its dormant compressed shoots that are very difficult to get rid of (including with chemicals) distinguish it from other weeds.
Note that Hydrilla would dominate and ultimately take over the Egeria (oxygen weed) that is currently in Lake Rotoroa (the Egeria is considered under control at the moment). Prior to the Egeria, there were other oxygen weeds present… but eventually Egeria dominated.

[18] In New Zealand, hydrilla was first recorded in the 1960's and at present it is limited to three Hawkes Bay lakes. It has recently been eradicated from privately owned lake in Hawkes Bay after about two decades of effort; use of carp was the solution.

Hydrilla is far more problematic than any other aquatic weed species currently present in New Zealand because it can also propagate via special growths call turions and tubers which can persist for long periods, sprouting years later and surviving ice-cover, drying, ingestion and regurgitation by waterfowl, and herbicide use.
Hydrilla's sale and distribution has been prohibited since 1982, and it is ranked as one of the highest risk aquatic weed species in New Zealand. In 2006 it was made a 'notifiable organism'. This places a duty on any person aware of hydrilla in a new location to notify Biosecurity New Zealand
Submerged aquatic weed distribution is significantly correlated with boating and fishing activities rather than spread through wildlife or other natural means (e.g. wind). This is because stem fragments, turions and/or tubers can be transferred between and within water bodies by boating and fishing equipment and establish in new places. Activity controls on the four Hawkes Bay lakes containing hydrilla (particularly the bans on motorised boats and commercial eeling), and associated public awareness campaigns have no doubt prevented its spread throughout New Zealand
[19]

- Displacement of and/or loss of biodiversity, especially native aquatic flora and fauna due to alteration and loss of habitat (e.g. removing clear water and/or food for fish; reducing water quality)
- Native aquatic plants such as charophytes, pondweeds and milfoils are at risk in the 1-5m depth zone; Shallow water ways are particularly at risk. Existing weed species are also out-done by hydrilla which is very concerning as those other weeds can be effectively controlled with diquat, but hydrilla cannot. Shoreline flora is at risk from floating mats of dislodged hydrilla being driven into bays or against shorelines by wind where hydrilla overwhelms resident vegetation
- Hydrilla can remove essential habitat for fish and birds by reducing water quality and reducing the available clear water habitat. The impact on both native fish and birds will depend on the bird species present in the lake and what their use of the water ecosystem is (e.g. whether for food, shelter or both)

[20] As illustrated by the list on the previous slide, Hydrilla's potential impact on our freshwater ecosystems is likely to be very prolific.
These photos give you an idea of what those impacts might look like:
Hydrilla caught in a boat propeller
Surface reaching weed mats of hydrilla which restrict boating and cause damage to equipment and economic activity (e.g. irrigation, hydro-power)

[23] Charophytes are an important part of the ecosystem because they absorb nutrients from the water and help to stabilise bottom sediments.
New Zealand has internationally recognised communities of species under threat; charophytes and deep water bryophytes, many species in these

broader groups and these are under threat from pests as well as declines in lake water clarity. These species groups are under threat internationally.

[24] This slide gives you an idea of the impact of invasive weeds on charopyte meadows.

[25] Lake water quality and clarity are critical for life-supporting capacity
Underwater plants need light for photosynthesis
Fauna need to see their prey
Flora and fauna need the right balance of nutrients to survive; if essential nutrients (e.g. oxygen) go out of balance, so does the ecosystem
Water quality has improved over recent years as nutrient levels decline and aquatic plants re-establish. There was a sudden vegetation decline in the late 1980's; such plants absorb nutrients from the water and help to stabilise bottom sediments.
Hydrilla would reduce water clarity and quality by (for example):
- Displacing the charophytes which stablise bottom sediments
- Exclude light from the lake due to its dense growth

[26] Hamilton City Council management plan aims to encourage more shags around the lake,
Other birds are typical exotic city birds (e.g. pigeons, sparrows). Black swans once lived at the lake but left after their food source - the submerged weed beds – collapsed in the early 1990's. That said, along with the return of egeria, there have been some sightings of black swans (HCC 2006).

[27] Lake Rotoroa has diverse fish fauna by New Zealand standards; six exotic fish species (goldfish, rudd, tench, brown bullhead catfish, mosquitofish and perch) and four native species (shortfinned and longfinned eels, common smelt and common bully) are found. Exotic fish were stocked for recreational fishing (Clayton & de Winton 1994; HCC 2006).
Freshwater mussels were part of the original habitat before humans altered it… In the year 2001, 3000 mussels were re-introduced to the lake. In 2007, 65 mussels found in the sandy shallow substrates in Scooter Boat Bay and it is hoped they will continue to survive and breed, particularly as filter feeders they can have a positive impact on water quality. But, the arsenic in the lake sediment (from weed control in the 1950s) may be preventing the mussels from thriving.

Eels can tolerate weedy environments and it is unlikely they would be affected.

[29] If there is one variable among these that is much more important to you than any of the others, its very important that you don't 'protest vote' against the other variables. Please weigh up your willingness to pay for the chosen

option against other demands on your budget. In a real world situation this would be an addition to your regular rates bill.
After the survey questions, we will revisit this. You will have the opportunity to make any comments on variables which dominate your thinking and why.

Appendix 2 Random parameter logit (RPL) comparative analysis: results of individual case studies

Coastal Marine

Scope	**Local**	
Sample location	(Pauatahanui)	
	Coefficient	SE
RECN	-0.888***	0.183
VEG	-0.919***	0.149
SHELLS	-2.583***	0.248
NOPADDLE	-2.229***	0.268
MONEY	-0.417***	0.004
Model statistics		
RPL model with panel groups of	47	
Number of observations	564	
No. of observations per group	12	
Log likelihood function	-469.549	
Info. Criterion: AIC	1.682	
Info. Criterion: BIC	1.721	
Info. Criterion: HQIC	1.697	
Restricted Log Likelihood	-619.617	
McFadden Pseudo R^2	0.242	
Chi square	300.135	
Degrees of freedom	5	

Note: ***=P<0.01

South Island high country

Scale	Local		Within District		Within Region		Out of Region	
Sample location	(Twizel)		(Fairlie)		(Timaru)		(Christchurch)	
	Coefficient	SE	Coefficient	SE	Coefficient	SE	Coefficient	SE
PLANT	0.754***	0.157	0.845***	0.159	1.080***	0.213	0.969***	0.137
INSECT	1.109***	0.150	1.223***	0.163	1.657***	0.231	1.410***	0.152
FISH	1.374***	0.165	1.176***	0.150	2.560***	0.304	1.420***	0.151
MONEY	-0.009***	0.001	-0.020***	0.002	-0.017***	0.002	-0.010***	0.001
Model statistics								
RPL model with panel groups of	37		41		35		52	
Number of observations	589		651		559		832	
Observations per group	16		16		16		16	
Log likelihood function	-410.938		-434.292		-212.725		-485.012	
Info. Criterion: AIC	1.419		1.355		0.786		1.182	
Info. Criterion: BIC	1.471		1.403		0.840		1.222	
Info. Criterion: HQIC	1.439		1.374		0.807		1.197	
Restricted Log Likelihood	-647.082		-715.196		-614.124		-914.045	
McFadden Pseudo R^2	0.364		0.392		0.653		0.469	
Chi square	472.288		561.809		802.796		858.065	
Degrees of freedom	7		7		7		7	

Note: ***=P<0.01

Freshwater

Scale	Local		Within District		Within Region		Out of Region	
Sample location	(Rotoroa)		(Hamilton)		(Morrinsville)		(Wellington)	
	Coefficient	SE	Coefficient	SE	Coefficient	SE	Coefficient	SE
HYD	0.025***	0.0028	0.019***	0.002	0.024***	0.002	0.020***	0.001
CHA	0.084***	0.014	0.083***	0.013	0.065***	0.010	0.076***	0.010
BIR	0.429***	0.056	0.307***	0.046	0.336***	0.038	0.446***	0.044
FISHMUS	0.454***	0.071	0.520***	0.070	0.415***	0.051	0.438***	0.052
PRICE	-0.006***	0.001	-0.007***	0.001	-0.008***	0.001	-0.010***	0.001
Model statistics								
RPL model with panel groups of	44		40		65		64	
Number of observations	528		480		780		768	
Observations per group	12		12		12		12	
Log likelihood function	-356.250		-362.305		-575.637		-565.046	
Info. Criterion: AIC	1.368		1.530		1.488		1.484	
Info. Criterion: BIC	1.408		1.573		1.5182		1.514	
Info. Criterion: HQIC	1.384		1.547		1.500		1.496	
Restricted Log Likelihood	-580.067		-527.333		-856.917		-843.734	
McFadden Pseudo R^2	0.386		0.312		0.328		0.330	
Chi square	447.634		330.056		526.561		557.3763	
Degrees of freedom	5		5		59		5	

Note: ***=P<0.01

Beech forest

Scale	Within Region		Out of Region	
Sample location	(Nelson)		(Christchurch)	
	Coefficient	SE	Coefficient	SE
STINGS	-0.083***	0.006	-0.069***	0.005
NO BIRDS	-6.233***	0.717	-2.645***	0.249
LOT BIRDS	2.057***	0.167	1.129***	0.132
NO BUGS	-3.401***	0.317	-1.381***	0.174
LOT BUGS	1.664***	0.155	0.815***	0.131
COST	-0.014***	0.001	-0.009***	0.001
Model statistics				
RPL model with panel groups of	91		75	
Number of observations	1812		1499	
Observations per group	20		20	
Log likelihood function	-986.812		-1162.627	
Info. Criterion: AIC	1.096		1.560	
Info. Criterion: BIC	1.118		1.585	
Info. Criterion: HQIC	1.104		1.569	
Restricted Log Likelihood	-1990.685		-1646.820	
McFadden Pseudo R^2	0.504		0.294	
Chi square	2007.747		968.386	
Degrees of freedom	7		7	

Note: ***=P<0.01

Appendix 3

Biodiversity Valuation Database

(WTP: means, standard deviations, 95% confidence intervals)

Sample Location		Attribute	WTP	SD	Z	P(Z)	95% confidence Lower	Upper
South Island high country - Wildings								
Local	Twizel	Plant	80	33	2.43	0.01	16	144
	Twizel	Insect	121	47	2.58	0.01	29	213
	Twizel	Fish	144	62	2.34	0.02	23	265
Within district	Fairlie	Plant	41	17	2.47	0.01	8	74
	Fairlie	Insect	59	25	2.39	0.02	11	107
	Fairlie	Fish	57	23	2.45	0.01	11	103
Within region	Timaru	Plant	59	25	2.40	0.02	11	107
	Timaru	Insect	93	38	2.47	0.01	19	166
	Timaru	Fish	145	53	2.75	0.01	42	249
Out of region	Riccarton	Plant	97	38	2.55	0.01	23	172
	Riccarton	Insect	138	58	2.36	0.02	23	253
	Riccarton	Fish	139	59	2.36	0.02	24	254
Beech forest - wasps								
Within region	Nelson	No Bird	431	167	2.57	0.01	103	759
	Nelson	Lot Bird	138	62	2.23	0.03	17	259
	Nelson	No Bug	222	103	2.17	0.03	21	423
	Nelson	Lot Bug	109	49	2.24	0.03	14	204
Out of region	Riccarton	No Bird	274	127	2.16	0.03	25	523
	Riccarton	Lot Bird	118	50	2.33	0.02	19	217
	Riccarton	No Bug	142	65	2.19	0.03	15	269
	Riccarton	Lot Bug	87	35	2.48	0.01	18	156
Coastal marine - crabs								
Local	Pauatahanui	Recn	22	9	2.58	0.01	5	39
	Pauatahanui	Veg	23	9	2.62	0.01	6	39
	Pauatahanui	NoPaddle	44	26	1.72	0.09	-6	95
	Pauatahanui	Shells	56	31	1.84	0.07	-4	116

Contd.

Freshwater - hydrilla

Local	Rotoroa	Charophytes	12	2	5.97	0.01	9	17
	Rotoroa	Birds	61	14	4.29	0.01	38	86
	Rotoroa	Fishmus	64	15	4.42	0.02	38	102
Within district	Hamilton	Charophytes	11	2	5.57	0.01	7	14
	Hamilton	Birds	38	7	5.16	0.01	24	50
	Hamilton	Fishmus	64	16	4.04	0.01	38	98
Within region	Morrinsville	Charophytes	8	1	7.31	0.01	6	10
	Morrinsville	Birds	39	9	4.57	0.01	23	60
	Morrinsville	Fishmus	49	9	5.61	0.01	31	71
Out of region	Wellington	Charophytes	7	1	6.82	0.01	5	10
	Wellington	Birds	41	11	3.65	0.01	21	65
	Wellington	Fishmus	41	8	4.91	0.01	26	59

Notes

- WTP is the mean estimate of willingness to pay
- SD is the Standard Deviation of the WTP, a measure of the variation in the estimate
- Z is WTP divided by SD
- P(Z) is the probability that the WTP is not significantly different from zero. The standard test is that P(Z) < 0.05 indicates the estimate is significantly different from zero
- All WTP values are significantly different from zero at the 95% level except NoPaddle and Shells for Coastal marine, which are significant at the 90% level P(Z) < 0.10
- The upper and lower bounds at the 95% confidence level indicate the levels of the WTP at which there is a 5% chance the value will lie outside these bounds
- The WTP estimates are annual estimates over 5 years, except for Coastal marine which are over 3 years.

Appendix 4 Correlation coefficients

Coastal Marine

Local (Pauatahanui) Correlation coefficients

	Recn	Veg	Shells	NoPaddle
Recn	1.0			
Veg	0.0	1.0		
Shells	0.4	0.2	1.0	
NoPaddle	0.4	0.3	0.5	1.0

South island high country

Local (Twizel) Correlation coefficients

	Plant	Insect	Fish
Plant	1.0		
Insect	0.1	1.0	
Fish	0.2	0.3	1.0

District (Fairlie) Correlation coefficients

	Plant	Insect	Fish
Plant	1.0		
Insect	0.2	1.0	
Fish	0.3	0.2	1.0

Region (Timaru) Correlation coefficients

	Plant	Insect	Fish
Plant	1.0		
Insect	0.4	1.0	
Fish	0.5	0.5	1.0

National (Riccarton) Correlation coefficients

	Plant	Insect	Fish
Plant	1.0		
Insect	0.3	1.0	
Fish	0.2	0.4	1.0

Freshwater

Local (Rotoroa) Correlation coefficients

	Cha	Bir	FishMus
Cha	1.0		
Bir	0.2	1.0	
FishMus	0.4	0.3	1.0

District (Hamilton) Correlation coefficients

	Cha	Bir	FishMus
Cha	1.0		
Bir	0.2	1.0	
FishMus	0.3	0.3	1.0

Region (Morrinsville) Correlation coefficients

	Cha	Bir	FishMus
Cha	1.0		
Bir	0.1	1.0	
FishMus	0.3	0.2	1.0

National (Wellington) Correlation coefficients

	Cha	Bir	FishMus
Cha	1.0		
Bir	0.2	1.0	
FishMus	0.3	0.2	1.0

Beech forest

Region (Nelson) Correlation coefficients

	Stings	No birds	Lot birds	No bugs	Lot bugs
Stings	1.0				
No birds	0.1	1.0			
Lot birds	-0.3	0.0	1.0		
No bugs	0.2	0.0	-0.4	1.0	
Lot bugs	-0.4	-0.3	0.2	0.0	1.0

National (Riccarton) Correlation coefficients

	Stings	No birds	Lot birds	No bugs	Lot bugs
Stings	1.0				
No birds	0.1	1.0			
Lot birds	-0.2	0.1	1.0		
No bugs	0.1	0.0	-0.2	1.0	
Lot bugs	-0.2	-0.2	0.0	0.4	1.0