Valuing Indigenous Biodiversity

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This paper outlines the rationale and hypothesis for a Masters thesis proposal in Applied Economics at Massey University. Comments, criticism and suggestions are welcome and should be forwarded to the principal author:

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Summary

This paper explores the effect of an individual's knowledge of biodiversity on the nature of his or her preferences for its preservation. Previous research suggests that individuals have a limited understanding of the concept of biodiversity and that some may be unwilling to trade-off changes in biodiversity against income.

We hypothesize that the way in which individuals understand biodiversity is such that meaningful preferences for biodiversity preservation are more likely to be expressed for large scale non-marginal changes (i.e. a regional or greater scope geographically and at a genus or greater scope genetically). Similarly we suggest that individuals can express preferences for different management regimes or policies at a large scale but are limited by a lack of technical expertise at the species or site scale. Many of the methodological constraints relating to non-market valuation of biodiversity at the species or genetic scale, are less critical at the larger scale. Similarly, the degree of uncertainty about functional relationships at the species level are less critical when considering an individual's willingness to pay for an aggregate measure of biodiversity preservation.

A discrete-choice contingent ranking valuation study is proposed to identify willingness to pay to preserve biodiversity and preferences for different management strategies. The study will address the value of endemic biodiversity in a lowland ecosystem in one region of New Zealand.

1. Introduction

The indigenous plants and animals of New Zealand are diverse, distinctive and vulnerable. Long isolation and a unique bioclimatic history have produced a special biota. Protecting indigenous species and communities, and assessing and developing sustainable use of biota are the responsibilities of policy and conservation agencies under the Conservation Act (1987) and the Resource Management Act 1991 (RMA). Under section 6 of the RMA, Territorial Local Authorities are responsible for protection of areas of significant indigenous vegetation and significant habitats of indigenous fauna outside the Conservation Estate. Recent proposed District Plans that imposed use controls and, by implication, opportunity costs of protection over such areas have met with significant resistance and political backlash. In many cases land owners retain existing use rights over areas of indigenous vegetation under the RMA.. Territorial Local Authorities are now faced with a significant policy issue: how to fulfill the requirements of section 6 in a politically acceptable and economically efficient manner. This proposed investigation of the values attributed to biodiversity and individual's willingness-to-pay (WTP) to conserve biodiversity will make a contribution to that issue.

2. Biodiversity in New Zealand

New Zealand may have 80 000 species of native animals, fungi and plants, only about 30 000 of which have been described and named (Ministry for the Environment 1997). Plants and large animals account for barely 5 000 native species in total (see Table 1).

New Zealand's biodiversity is more primitive in character than that of many other countries, having a limited representation of higher plants and animals (e.g., angiosperms and mammals), but a high representation of older plants and animals (e.g., mosses, liverworts, ferns, flatworms, snails, spiders, wingless crickets, solitary bees, leiopelmid frogs, sphenodon reptiles and ratite birds). Many species are endemic.

Since human settlement, indigenous species and communities have become increasingly fragile. For its size, New Zealand has a disproportionate share of the officially recognised extinct and threatened plant and animals species on the globe (Ministry for the Environment 1997). In only 700–800 years, humans and their accompanying animals have eliminated, among others, 32 percent of the endemic land and freshwater birds (43 species and 9 subspecies), 18 percent of the endemic seabirds (4 species out of 22), 3 of the 7 frogs, possibly 11 of the 2 300 vascular plants, and at least 12 invertebrates, such as snails and insects (Ministry for the Environment 1997). Nearly 1 000 animals, plants and fungi have been identified as threatened (Some 800 species and 200 subspecies). Among these are more than 200 fungi (5 percent of known species); nearly 200 vascular plants (10 percent); 85 non-vascular plants (8 percent); 150 vertebrate animals (58 percent); and at least 285 invertebrate animals (1–2 percent). One of the worst affected groups is our endemic land and freshwater birds, three-quarters of which (37 out of 50 species) are now threatened. The numbers of most other surviving species and subspecies have also been heavily reduced.

The viability of many species continues to be threatened by mammalian predators and herbivores, invasive weeds, and habitat loss and fragmentation due to conversion, drainage or overuse. Most of the New Zealand landscape is now ecologically hostile to many native species. Developed land now claims 63 percent of the total land area and more than 90 percent of the lowland area (see Table 2). Although nearly 30 percent of the land area is protected in the conservation estate, most of this is on steep and mountainous land.

Lowland forests, dune lands, and wetlands are under-represented in our protected areas. In most areas they have been reduced to fragments and will need considerable expansion if the biodiversity within them is to be sustained (Saunders *et al.* 1991). Most of the surviving indigenous lowland forests are not in Crown ownership (except for 150 000 hectares set aside for timber production on the West Coast). These forests are unprotected from conversion to other land uses, but timber production from them is subject to the sustainable management provisions of the Forests Act 1949 (except for 60 000 hectares set aside for economic purposes under the South Island Landless Maoris Act 1906).

3. Right and responsibilities

Protecting indigenous species and communities, and assessing and developing sustainable use of biota are the responsibilities of policy and conservation agencies under the Conservation Act (1987) and the Resource Management Act (1991). New Zealand also has obligations under international agreements (e.g., Agenda 21, Convention on Biological Diversity).

New Zealand society's responses to the pressure from habitat loss have included:

a) the preservation of nearly 8 million hectares of publicly owned mountain areas with several thousand hectares of lowland reserves and unoccupied offshore islands, under the Conservation Act 1987, the National Parks Act 1980 and the Reserves Act 1977

b) the preservation of approximately 100 000 hectares of habitat on private land through government-funded covenants and purchases, arranged through the Forest Heritage Fund, Nga Whenua Rahui, and the Queen Elizabeth II National Trust

c) the removal of all primary production incentive and support schemes that perversely resulted in accelerated biodiversity loss, e.g., land development incentives, forestry encouragement grants

d) requirements in the Resource Management Act 1991 that decision-makers provide for protection of areas of significant indigenous vegetation, significant habitats of indigenous fauna, and wetlands

e) a voluntary agreement, the Forest Accord, between forest development companies and conservation groups that prevents felling of mature indigenous vegetation for conversion to exotic forest.

Responses to the pressures from alien species include:

a) the Department of Conservation's more than 500 species management programmes including some 40 Species Recovery Programmes; 600 pest and weed control operations/annum, roughly equally divided between animal pests and noxious weeds, and spanning more than 1 million hectares of conservation land

b) pest control programmes, mainly in agricultural areas, by regional councils and the Animal Health Board under the Biosecurity Act 1993

c) testing and risk assessment of introducing new organisms, such as the calici virus, under the Hazardous Substances and New Organisms Act 1996.

3.1 The Conservation estate

The Conservation Act (1987) empowers the Department of Conservation (DOC) to preserve and protect natural resources held under the Act for the purpose of maintaining their intrinsic values, providing for their appreciation and recreational enjoyment by the public, and safeguarding the options of future generations. In effect, the Act guides what is to be protected, and the DOC's annual budget allocation determines how much can be protected. The magnitude of the task exceeds the financial resources available and is hampered by the lack of an agreed quantifiable measure of conservation outcome (Stephens & Lawless 1998). The establishment of the annual budget would appear to be unrelated to the value of the services provided by Conservation estate.

3.2 Indigenous habitats and biological diversity on private land

Under section 6 of the RMA, Territorial Local Authorities are responsible for protection of areas of significant indigenous vegetation and significant habitats of indigenous fauna outside the Conservation Estate:

> In achieving the purpose of the Act, all persons exercising functions and powers under it, in relation to managing the use, development and protection of natural and physical resources, shall recognise and provide for the following matters of national importance:
>
> a) The preservation of the natural character of the coastal environment, wetlands, lakes, and rivers and their margins, and the protection of them from inappropriate subdivision, use and development

b) The protection of outstanding natural features and landscapes from inappropriate subdivision, use and development
c) The protection of areas of significant indigenous vegetation and significant habitats of indigenous fauna."
(Section 6, Resource Management Act 1991.)

Under Section 31 of the RMA this responsibility falls to District and City Councils. Case law subsumes the impact of section 6 under the broader RMA goal of sustainable management. This raises the questions of what is it that is to be sustained? Alternatives could include the organisms that perform environmental functions, the functions themselves or the services provided by those functions (Hueting *et al.* 1997). Similarly there are scale issues. It is not clear whether "significant" areas or habitats include the full complement of genetic and species diversity and richness that make up one measure of biodiversity. Implementation is further complicated by existing use rights recognised under section 10 of the RMA which limits a TLA's ability to land owner's use of many areas of indigenous vegetation.

A common approach to date has been to use the recommended areas for protection (RAP) identified in the Protected Natural Areas Programme (PNA) administered by the Department of Conservation. This programme identifies areas of indigenous vegetation outside the Conservation estate with significant conservation values. The Department also maintains a register of Sites of Significant Biological Interest (SSBI) which some TLA's have used in identifying significant areas or habitats. Inclusion of such areas in District Plans has led to widespread concern among land owners. Land owners were often unaware of the scientific assessment of the significance of areas of indigenous vegetation on their property and use of these assessments to nominate areas in planning documents has generated significant resentment. At least one District Council has approached each land owner with an RAP on their property and negotiated their consent before listing the area in the District Plan.

Once registered, District Councils have proposed a range of policy measures to protect these areas. The use of rules to prevent removal or modification of such areas has been spectacularly unsuccessful in a few highly publicised cases. In at least one case, where land owners have applied concerted pressure, the proposed plan has been withdrawn. Fundamentally, it is an issue of property rights and it would appear that the will of society is not yet sufficiently strong to enforce complete protection of all remaining significant areas of indigenous vegetation.

Councils are now considering a wider policy package to meet the requirements of the RMA including:

a) Encouraging land-owner involvement and addressing any lack of knowledge or understanding of biodiversity issues through education and facilitation of information distribution
b) Facilitating implementation: Non-financial (management plans, accreditation) and financial incentives (covenants/management agreements, licences) are being considered
c) The use of regulation as a last resort to protect significant habitat in emergencies and against recalcitrants.

This wider range of policy options often requires some investment by TLA's and begs the question how much should we invest? The uncertainty surrounding the requirements of the RMA provide local Government with some flexibility in addressing this issue. The approach proposed in this paper provides one mechanism for assessing the social benefit of a local investment in biodiversity conservation. While there has been good progress in enforcing individual responsibility for environmental damage by integrating costs in the production using polluter and user pays mechanisms, the conservation of biodiversity seems to remain a community responsibility. Its perceived public good value suggests that funding for biodiversity conservation will primarily come from social charges (taxes and rates) for some time yet.

4. Biodiversity as an economic good

In a classic paper on the economics of conservation, Krutilla (1967) identifies the key economic problem associated with the conservation of natural resources:

> ...at any point characterised by a level of technology and a set of social preferences, the irreversible conversion or loss of natural resources in the production of private goods has proceeded further than it would have with future technology and the future composition of social preferences...

The primary reasons he identifies for private and social costs associated with the conservation of natural resources to diverge are:

a) The inability of private owners to appropriate the total social value of resources when they are conserved because of practical obstacles to discriminate pricing perfectly

b) The maximum social willingness to pay (the area under the demand curve) may not be adequate to compensate private owners for the preservation of the resource

c) the inability of a private owner to appropriate the social option, existence and bequest values of a natural resource because of the absence or inadequacy of a market for these values. Imperfect markets at best exist for these values because of the absence of knowledge about the present qualities and potential products available from natural resources and the public good nature of many of the benefits derived from natural resources (Krutilla 1967).

He concludes that imperfect information and the irreversibility of ecosystem or species loss mean a cautionary approach to natural resource use and policy is warranted.

Can these difficulties be overcome and a rational economic strategy for natural resource use be developed? Many commentators argue not (Norgaard 1989, Gowdy 1997, Spash & Hanley 1995). Norgaard (1989) argues that there is no single correct method of understanding the complex interactions between economic development and the environment. A rational approach to combining environmental and economic information is forestalled by the lack of single consistent theories of either economics or ecology, by the value-aggregation dilemma, i.e. societal values differing from the simple arithmetic sum of a society’s individuals, and by the bounded-knowledge synthesis dilemma, i.e. the synthesis of bounded knowledge into knowledge of the whole is necessary for the derivation of a consistent model of the interaction of economic development and the environment but we have no meta-model to provide the framework for such a synthesis.

The difficulty in resolving the issues discussed above is reflected in the range of value theories that have been proposed. Some commentators argue that the issues are unresolvable and that valuing biodiversity should not be attempted, others that biodiversity is infinite in value because it is essential to the sustainability of life on earth (Gowdy 1997). Alternatives to neoclassical economic models have also been suggested. One that has received considerable attention is based on energy and mass balances (Patterson 1996). Perhaps, if there is any level of consensus at all, it is for methodological pluralism. A range of value frameworks and units may need to be used because the complex values and issues being assessed can not be reduced to a universal one-dimensional unit (Gowdy 1997, Norgaard 1989, Patterson 1996).

The utilitarian view of biodiversity that we use as our behavioural assertion in this paper follows from Krutilla's original observations, and sees the continued loss of species and ecosystems as the result of the failure of policy- or market-based decisions to consider the value of ecosystem services. From this perspective, it may be possible to reduce the rate of loss of biodiversity if we can address that market failure by making the value of those services explicit and building them into cost benefit analyses. An intrinsic value as implied by the Conservation Act cannot be incorporated in such a utility maximising framework.

In this paper we use the following definition of biodiversity as an economic good:

> The marginal value of biodiversity is the return from an incremental change in the services provided by the diversity of genetic material, species and ecosystem stocks in New Zealand.

This definition draws on Hueting's definition of an environmental function (Hueting *et al.* 1997) and implies a marginal valuation of diversity <u>not</u> a total value of biotic resources. The services provided by biodiversity are discussed later.

Biodiversity is an exhaustible good within human time frames. While there is continuing growth in biodiversity through evolutionary processes, estimated rates are slow in human terms. Clearly we are only in a position to affect the rate of biodiversity loss rather than increase the total stock of genetic material despite our ability to rearrange specific parts between species. The supply curve for biodiversity is therefore incomplete in that it describes the costs of measures to maintain biodiversity at any level up to that which exists at present. The shadow price for biodiversity, then, reflects the elasticity of substitution of biodiversity as an input to production, in its broadest sense, and the rate of change in the shadow price would be equivalent to the social utility discount rate. Although not pursued further in this paper, it would be interesting to attempt to infer the latter from estimates of change in biodiversity.

4.1 Quantifying biodiversity

Biodiversity can be described at each of three fundamental levels of biological organization – genetic diversity, species diversity, ecosystem diversity. Biodiversity is maintained by processes that operate at regional, community, species, and genetic levels, and interact with one another. There appears to be no clear consensus on how to measure biodiversity. Genetic diversity is measured in terms of phenotypic traits, allelic frequencies or DNA sequences. While natural diversity between individuals is commonly quite wide, an endangered species saved from extinction will probably have lost much of its internal genetic diversity. Species diversity is a function of the distribution and abundance of species. More recent measures also incorporate the relatedness of the species in a fauna or the measures of the degree of genealogical difference and the spread of species from across the subgroups of the cladogram (Pearce & Moran 1994).

At the community or ecosystem level unambiguous boundaries delineating units of biodiversity do not exist. Many different units of biodiversity are involved including the pattern of habitats in the community, the relative abundance of species, population age structures, patterns of communities on the landscape, trophic structure and patch dynamics (Pearce & Moran 1994). Furthermore, these characteristics and the relationships between them change continually. As a result, there is a range of different approaches to measuring ecosystem diversity, including biogeographical provinces based on distribution of species, and ecoregions based on soil and climate attributes.

The lack of consensus on how to measure biodiversity has important implications for the economics of biodiversity conservation. At its most basic level any measure of cost-effectiveness used to guide investments in conservation must have some index of biodiversity change. In New Zealand there are moves to develop improved methods of quantifying biodiversity. Stephens and Lawless (1998) have proposed a method for measuring conservation status and outcomes at the ecosystem level using a Natural Heritage Status Index (NHSI) that contrasts the present extent and condition of a given ecosystem with its pre-human state. The system is built on a classification of ecosystems being developed by Landcare Research which uses species distribution, climate, land form, and disturbance factor data. The NHSI is a measure of site size and condition and while it provides an indication of the *potential* of a site to maintain indigenous biodiversity, it does not provide information about the actual diversity of that site. Stephens and Lawless (1998) also acknowledge a weakness in that it does not take into account the loss of iconic species or habitats, and so overestimates the status of sites.

The degree to which recent valuations of ecosystems services have explicitly captured the value of biodiversity is questionable. Diversity valuation requires some idea of the willingness to pay for the *range* of genetics, species, and communities within an ecosystem rather than the few specific biological resources an ecosystem may happen to support.

4.2 Services from biodiversity

The most commonly used value framework for environmental services is that of Pearce and Moran (1994) in which total economic value is broken down into use and non-use values. Use values include both direct use of environmental resources for production, recreation, etc., and indirect use through services provided by conservation of soil and water values etc. A third category of use values – option values, are akin to insurance where the potential for future use is retained by maintaining the resource in its current state. Non-use values include those relating to the knowledge that the resource continues to exist (existence values) or that future generations have access to the resource (bequest values). In this framework, the economic value of a resource reflects the flow of services it provides. Costanza *et al.* (1997) argue that the total economic value thus defined is still inadequate because the level of flow or extraction of services may be unsustainable. The value therefore fails to include the value of the ecosystem integrity which underpins the provision of such services.

This issue is particularly pertinent to the valuation of biodiversity. While species extinction is an important indicator of biodiversity loss, recent thinking is that it is not the crux of the problem. Rather, conservation of biological diversity is of vital importance because some level of biodiversity is essential to the functioning of ecosystems on which not only human consumption and production but also existence depend (Barbier *et al.* 1994). Biodiversity is akin to a capital stock value (Söderbaum 1992) . In this sense, although natural capital is not subject to depreciation, it is subject to erosion. A measure of biodiversity is central to the economic value of ecosystem services because it reflects the ongoing ability of the ecosystem to supply those services.

a) Ecosystem resilience

Ecosystems are a complex set of interdependencies between the system's components, and are continually in a dynamic process of development and change. The resilience of an ecosystem is a measure of the extent to which it can be subjected to disturbances without the system's parameters being changed. A system's resilience is not constant, however. Ecologists suggest that resilience tends to be greater the higher the degree of complexity, diversity and interlockedness of the ecosystem. Economic behaviour tends to reduce ecosystem complexity, diversity and interlockedness. As resilience is reduced, so the level of disturbance to which the ecosystem can be subjected without parametric change is reduced. These processes are not well understood. The precise impacts of a loss in biodiversity are difficult to predict, but the eventual costs of continued loss in terms of ecosystem collapse and decline in fundamental functions such as nutrient cycling, biological productivity, hydrological regulation and sediment control, are unavoidable.

For any individual, maintaining ecosystem resilience is an indirect use value of biodiversity. In the absence of clear evidence of local loss in ecosystem services it does not seem to be well understood or highly valued. Perversely, it may be that the natural resilience of ecosystems to significant modification may also be part of the reason that society adopts a "business as usual" strategy to resource use rather than a more precautionary approach which would seem rational under conditions of uncertainty and potential catastrophic loss.

b) Cultural and existence values

In the absence of strong information about the functional and system roles of species within ecosystems, existence or passive values have come to be recognised as the most significant source of value in non-market valuation of biodiversity. These values include elements of moral conviction, cultural and historical values, altruistic motives and beliefs about the rights of future generations (bequest values). Existence values accrue both to users and those not actually "using" a natural resource but with an interest in. The validity of quantifying existence values continues to be debated because of the range of values that underpin them. The debate hinges around the question of whether such values actually constitute an economic preference with its associated assumptions and can be commoditised so as to be included in economic analysis (Gowdy 1997, Sagoff 1988, Vatn & Bromley 1994).

c) Potential productive value

Despite a small number of highly publicised contracts for access to property rights in indigenous biodiversity for genetic prospecting in the medical field (e.g., Merck and the Costa Rican Instituto Nacional de Biodiversidad), the quasi-option value of biodiversity does not appear to be high. Although there remains potential for new discoveries, expensive plant-based screening research has been overshadowed by the use of molecular biology and biotechnology applications to microorganisms. Plant searches are now targeted to families of plants with known benefits in traditional medicine. While an insurance argument for conserving plant biodiversity remains, the value of that in conserving biodiversity is uncertain and unproven.

5. Valuing biodiversity – a cost benefit framework

An ecosystem valuation forum established by the US EPA (Bingham *et al.* 1995) canvassed the issues associated with systems of value and methods of valuation. The forum put particular emphasis on approaching ecosystem valuation from the perspective of decision makers and on identifying decisive information. The decision-making context will determine whether cost effectiveness or cost benefit analysis are appropriate approaches. The forum identified the importance of understanding the roles that individuals play, i.e., personal, advisory, public citizen, the time frames in which concerns shift and environmental damage occurs, and the lack of understanding of ecosystem attributes as being critical factors that are still to be addressed in ecosystem valuation.

Cost Benefit Analysis (CBA) is a non-market project evaluation procedure that estimates the aggregate net effect on an individual’s utility in terms of marginal changes in observable consumption or output. It is founded on the Kaldor-Hicks potential compensation test that recommends projects be approved where there is potential to make at least one person better off and none worse off, i.e., a potential resource distribution after the project could result in a Pareto improvement (Spash & Hanley 1995). In the neo-classical framework, such potential compensation would be based on an increased utility associated with individual preferences.

The use of CBA is subject to a number of restricting assumptions including equal marginal utility of income across all individuals, a utilitarian intertemporal social welfare function, and a fair discount rate. These conditions potentially hold for marginal changes in localised benefits over short time frames. The benefits of biodiversity conservation, however, tend to affect many individuals over many periods of time, stretching the assumptions of equal marginal utility of income and of the appropriateness let alone the ability to define a fair discount rate. If the level of biodiversity is, at some point, critical to the continuing functioning of life-supporting ecosystems then the impacts are considerably greater than marginal for those affected.

5.1 Costs of biodiversity conservation

The costs of supplying different levels of biodiversity is only slightly less problematic than valuing the potential benefits. Costing the implementation of particular management activities is relatively straightforward as is determining opportunity costs of such measures, although little appears in the literature. What is more difficult is determining and quantifying the precise relationship between management action and the impact on biodiversity. Issues of definition, scale and magnitude, as discussed previously, mean that it is difficult to quantify marginal changes in biodiversity supplied. This issue is being pursued by others in New Zealand (Cullen *et al.* this conference, Stephens & Lawless 1998).

5.2 Valuing the benefits of biodiversity

a) Cost-based measures of value

Cost-based approaches include assessment of opportunity costs, defensive expenditures, and replacement cost. They are useful for valuing indirect services through cost of replacement but are dependent on a reasonable knowledge of exactly which services are under threat. They omit non-use values which are the major components of biodiversity value.

b) Willingness-to-pay (WTP) and willingness-to-accept compensation(WTAC) measures of value

Revealed preference methods (hedonic pricing, travel cost) are data intensive and are limited to the valuation of current use values, e.g., environmental quality in urban settings, recreational value of remote sites. Valuing non-market bequest, existence and cultural values is primarily dependent on stated preference techniques (contingent valuation, choice modelling). Patterson and Cole (1997) conclude that these passive or non-use values are extremely poorly researched for New Zealand.

The use of stated preference techniques for the valuation of non-use values of environmental goods and services is a subject of considerable debate (Portney 1994, Hanemann 1994, Diamond & Haussmann 1994).

5.3 Stated Preference Techniques

The economic validity of stated preferences is dependent on the preferences elicited from respondents being subject to the fundamental requirements of economic rationality. These include:

a) Completeness and comparability, i.e. that there be no holes of ignorance or points at which individuals are unable to express a preference
b) Transitivity and consistency, i.e. that there are no threshold effects or inconsistencies between close bundles of goods, and that there are no framing effects associated with the valuation survey instrument
c) Continuity of preferences, i.e. that individuals are prepared to substitute income or some other good for more or less of the good of interest
d) Non-satiation, i.e. that more of a good is always preferred to less of that good.

The valuation of biodiversity has the potential to challenge all these assumptions.

a) Uncertainty, complexity of ecological functional relationships, and the risk of irreversibility mean that it is quite probable that there are points at which individuals would be unwilling or unable to express a preference for a given value of biodiversity. In any given ecosystem there is commonly uncertainty about biodiversity extent (undiscovered species), scarcity, usefulness (e.g., future genetic potential), function (roles of component species), resilience to external impacts, and critical limits (e.g., thresholds beyond which collapse occurs). Ecosystems exhibit complex response mechanisms including threshold and cumulative effects, and asymmetric responses to impacts on keystone species and processes. Uncertainty differs from risk in that the probability of any specific outcome is unknown and so cannot be planned for. As a result an individual attempting to value biodiversity has considerable difficulty in defining what is being valued and commoditising that in monetary terms.

b) Similarly the potential for irreversibility and extinction make it highly likely that there will be marked threshold effects in individuals preferences for biodiversity in the vicinity of irreversible modification or extinction. There is considerable evidence already available for the existence of framing effects in stated preference valuations including difficulties with comparisons across scales, income effects, information effects, and payment effects(e.g., McDonald & McKenny 1996).

c) Ethical and moral judgements: The assumption of continuity of preferences has also received considerable attention in a variety of frameworks e.g. citizen versus consumer debates (Blamey *et al.* 1996), and lexicographic preferences or moral repugnance (e.g., Spash & Hanley 1995). As a composite good, biodiversity has a range of attributes that are arguably incommensurate (Vatn & Bromley 1994). The debate primarily hinges on whether moral and personal preference structures are incongruent. If so, individuals who approach decision making about public goods do so in such a way that their preferences reflect values either inconsistent with WTP/WTAC or inconsistent with the assumption of substitutability e.g., they accord species an intrinsic right of existence irrespective other considerations. This characteristic is likely to be reflected in protest bids. Spash and Hanley found significant numbers of protest responses reflecting lexicographic preferences in one study of biodiversity preferences. (Spash & Hanley 1995).

d) It is unclear whether diversity preferences will be consistent with the non-satiation assumption – are communities or species that are naturally more diverse inherently of greater value?

Other criticisms of the use of stated preference methods, which focus on estimating WTP rather than WTAC to establish believable contingent markets, include:

a) that non-market valuation techniques confuse social and private choices and that the social welfare function relating to natural resources is more than the aggregate of individual utility functions

b) that time preference discounting is incompatible with the social value of sustaining natural resources because it implies the acceptability of the complete collapse of all living systems at some future point in time

c) that the use of willingness-to-pay over willingness-to-accept measures has no coherent theoretical justification and results in an undervaluation of the environment (Bromley 1995).

The detailed review of contingent valuation methodology by the US Department of Commerce through the national Oceanic and Atmospheric Administration gave qualified approval to the use of contingent valuation as a method to evaluate non-use values and provided guidelines that are now universally applied. These include the use of face-to-face interviews, the use of willingness-to-pay (WTP) for a future hypothetical incident rather than willingness to accept (WTAC) compensation for a past loss, the use of the referendum rather than open-ended format, adequate scenario description and information about effects, the inclusion of reminders of the impact of WTP offers on expenditure on alternative items, and of substitutes for the commodity in question, and the inclusion of follow-up questions that ensure the respondents understood the choice they were being asked to make and elicit some of the reasons for making that choice.

An alternative stated preference approach that has recently been tested in environmental evaluation is that of stated preference or choice experiments (Boxall *et al.* 1996).The approach has been developed and is widely used in marketing analysis, geography and transportation economics. Choice modelling is similar to the contingent valuation method (CVM) in that its behavioural basis is grounded in random utility theory and it provides an absolute measure of welfare (Morrison *et al.* 1996). It differs from CVM in that environmental attributes are varied in an experimental design that requires respondents to make repeated choices between bundles of attributes. The use of an array of attributes and a range of choice situations makes it less dependent on the accuracy or completeness of the description of a specific change in good or service than CVM. The repeated choice experimental aspect of the approach also has advantages in that it specifically elicits tradeoffs being made between marginal changes in different attributes of a good (Boxall *et al.* 1996). Choice modelling also has an advantage over CVM because substitutes in consumption can be included in the valuation survey framework reducing the likelihood of pure embedding. Embedding is an effect where willingness to pay bids are inflated by a “warm glow” desire to contribute to a good cause.

5.4 Conclusion

Valuing diversity is more complex than valuing one specific “service” or “good” provided by a natural resource. It is highly likely that individuals will not assign the same value to different ecosystems even if levels of diversity are similar. Cultural values and values associated with iconic species or ecosystems are likely to influence perceptions of value.

Biodiversity also has an implicit value structure that is independent of immediate anthropic interest. The option, bequest and existence values associated with the concept imply a strong association with individuals assessments of the rights of future generations. Similarly, the lack of information on the future use value of diversity means any assessment of value implies certain attitudes to decision making under uncertainty.

For all these reasons, research into estimates of value for biodiversity will make an important contribution to our understanding of factors affecting natural resource management.

6. Proposed research

6.1 Rationale and objective

Our main objective in the proposed research is to test individuals' expressed preferences for lowland biodiversity in New Zealand against some of the prerequisite assumptions of stated preference analysis, and to determine whether the scale of biodiversity being considered affects the rationality or validity of those preferences. We characterise lowland biodiversity as a non-renewable environmental resource providing valued services in which marginal changes are characterized by uncertain outcomes and potential irreversibility.

We interpret the history of biodiversity loss as being consistent with an economic understanding of biodiversity as a non-renewable good with asymmetric information about attributes (valued on a utilitarian non-market basis, zero value while no evidence of scarcity, increasing value with growing scarcity of charismatic species but less so for habitats without commercial value, likely extinction where there is a high elasticity of substitution between species regardless of functionality). There is some evidence for non-economic interpretations of societal behaviour vis-à-vis biodiversity conservation in developed countries, (National parks, Conservation legislation) but these are consistent with economic expectations of efficiency in institutional arrangements for public goods.

We have selected lowland biodiversity because marginal changes are likely to have to be achieved on private land using incentive mechanisms given the institutional framework in New Zealand. The research will have direct relevance to District Councils as they develop responses to RMA requirements in their District Plans.

Two questionnaires are required to examine scale effects: one addressing biodiversity at the ecosystem scale, and the second addressing biodiversity at the species and sub-species scale. The framing of the questionnaires, the level of information and attributes to be included in the survey, and an exploration of the values and perceptions underpinning individuals' preferences for biodiversity conservation will require pre-survey exploration using Focus Groups (Morrison *et al.* 1997).

We accept that values and preferences are dynamic and that any valuation is pertinent to a point in time and related to the present state of the environmental good being valued and to societal norms at the time. We also accept that individual valuations will be determined by a multiplicity of factors including, among others, social norms, personal income, spiritual and moral beliefs, and existing property rights, but assert that a reasonable approximation of social benefit at a regional scale can be derived from analysis of responses from a sufficiently large random sample of individuals in that region.

6.2 Hypotheses

The logical implications of our rationale are that:

1. Functional and systems-based contributions to ecosystem resilience are undervalued, primarily as a result of lack of knowledge, and will result in continued biodiversity loss (cf. with the safe minimum standard level implied by moral/citizen approaches).

2. Preferences for biodiversity at the species or smaller scale where charismatic and moral factors predominate are likely to exhibit a high degree of part-whole bias i.e., the warm glow effect. For non-charismatic species a high marginal rate of substitution might be expected.
3. Estimation of environmental "prices" for biodiversity at the ecosystem level may provide a more consistent indication of preferences for biodiversity based on a wider range of attributes than at the species level where non-uniform preferences may be expressed based on charismatic factors unrelated to biodiversity issues.
4. The true social value of biodiversity is unlikely to be reflected in individuals' preferences until science or circumstance provide better information about functional relationships and effects on valued environmental services.

Therefore, we hypothesise:

a) Individual preferences for biodiversity conservation at the species and sub-species level are more likely to:
   i) be based on a particular set of existence values, specifically anthropocentric attributes such as cultural or historic significance, charismatic value and a sense of moral satisfaction in contributing to environmental health
   ii) be sensitive only to non-marginal changes in biodiversity i.e., for changes where there is a risk of extinction rather than for incremental reductions in extent or richness
   iii) have lower marginal rates of substitution between alternative biodiversity conservation options
   iv) demonstrate scope-related embedding when aggregated to the ecosystem scale.
b) At the community, ecosystem and landscape scales, preferences for biodiversity are more likely to
   i) include ecological attributes such as functional contributions to ecosystem and landscape resilience
   ii) be articulated for marginal changes in both extent and richness
   iii) have higher marginal rates of substitution between alternative biodiversity conservation options.

We also note that many of the methodological constraints relating to non-market valuation of biodiversity at the species or genetic scale, are less critical at the larger scale. Similarly the degree of uncertainty about functional relationships at the species level is less critical when considering an individual's willingness to pay for an aggregate measure of biodiversity preservation. We expect that our results will contribute to the argument that biodiversity conservation should be pursued at an ecosystem management scale in order to limit taxonomic bias.

6.3 Issues to be addressed by Focus Grouping and Survey Questionnaire

The goals of the focus groups will be to elicit underpinning values, understandings, and scale of conceptualisation of biodiversity and biodiversity conservation, and to develop a survey questionnaire that is framed to solicit the intended decision-making process.
Specifically, the outcome from the groups will include:

- an array of appropriate values that underpin individuals assessments of the value of biodiversity conservation. Values that may be appropriate include uniqueness, species intrinsic rights, quasi option value to future generations, functional role in maintaining ecosystem services, historic or cultural significance

- an assessment of the appropriate level of information to provide in the survey questionnaire
- appropriate substitutes to incorporate in the framing of the survey design
- appropriate scales of change to reflect marginal and non-marginal changes
- appropriate policy context, payment vehicle, and starting bids
- an array of appropriate reasons for zero or protest responses.

The contingent ranking framework involves:

- two scales of biodiversity selection of scale
- comparisons between conservation options involving threatened, non-threatened, charismatic, and non-charismatic diversity
- comparisons of marginal change where extinction is avoided, marginal changes with no impact on extinctions, non-marginal changes in diversity and extent
- opportunity to analyse the incidence of scope embedding and the marginal rate of substitution between diversity attributes
- sufficient demographic data and spatial distribution of sample individuals to test income, location, and environmental attitude effects.

6.4 Expected outcomes

Following from our primary hypotheses, we would expect the proposed research to provide some:

- indications of appropriate bases for regional scale biodiversity policy (values, scale)
- evidence of the effect of scale and framing on the validity of contingent value estimates for biodiversity
- indication of the level of individuals’ understanding of biodiversity issues and the values that underpin their preferences concerning biodiversity
- indication of the impact of locality and distance on preferences for biodiversity conservation.

6.5 Future work

Issues that potentially arise from or follow from the proposed research include:

- Implications for sustainable management under the RMA if there is significant differences between estimated WTP and estimated costs of preserving areas of significant indigenous vegetation and species habitat
- Implications of individuals’ WTP for biodiversity conservation on active ecosystem restoration proposals
- Potential for the use of choice modelling to determine WTP and preferences for alternative policy options in implementing biodiversity conservation
- The potential for a constructed preferences approach to evaluate preferences for goods subject to significant uncertainty (Vatn & Bromley 1994).
- The reliability of WTP estimates determined through choice modelling under actual payment scenarios.

Table 1 Terrestrial Biodiversity in New Zealand

Taxonomic Group			Estimated number of indigenous species	Number described	Endemic species (%)	Threatened species
Bacteria			?	200-300	?	?
Protozoa			7500	2600	5 %	?
Algae			4000	3700	?	?
Fungi			22000	5800	?	200
Plants	Vascular plants		2300	2022	81 %	200
	Mosses / Liverworts		1100	1060	20-40 %	85
Invertebrates	Arthropods	Insects	20000	10000	90 %	175
		Arachnids	4600	2600	90 %	36437
		Crustacea	2000	1517	?	5
		Other	720	250	?	?
	Molluscs		4800	2500	?	15
	Worms		17500	1500	30-40 %	?
	Other		2600	2100	?	?
Vertebrates	Fish	sea dwelling	1100	870	5 %	2
		rock pool	100	94	62 %	9
		fresh water	35	28	90 %	10
	Amphibians (frogs)		4	4	100 %	4
	Reptiles		61	61	100 %	25
	Birds	land / fresh water	88	88	57 %	37
		seabirds	61	61	30 %	18
	Mammals	land	2	2	100 %	2
		marine	41	41	5 %	4

Source: Ministry for the Environment 1997

Table 2 Land Use and Land Cover in New Zealand

Land Use and Land Cover		Land Area	
		Million ha.	Percent
Domesticated land	Pasture	13.52	50
	Crops	0.32	1.2
	Horticulture	0.09	0.3
	Other farm land (retired, fallow)	0.49	1.8
	Exotic forests	1.4	5.2
	State indigenous production forest	0.16	0.6
	Privately owned indigenous forest	1.32	4.9
Conservation land	Indigenous forest	4.8	17.7
	Tussock and subalpine vegetation	0.7	2.6
	Other (mountain tops, coastline, islands)	2.6	9.6
Built-upon land	Urban areas, rural roads, railways	0.89	3.2
Other land	Lakes, river beds, other	0.76	2.8
Total New Zealand land area		27.05	100

Source: Ministry for the Environment 1997

Table 3a A contingent ranking schema to elicit valid WTP bids for biodiversity (species level)

Implications	Option A Current situation	Option B Hold the status quo	Option C Marginal change	Option D Non-marginal change
Levy on your property rates	$0	$20, $60, $100, $140	$20, $60, $100, $140	$20, $60, $100, $140
Management of existing DOC reserves	No change	No change	No change	No change
Numbers of sites with sustainable populations of threatened charismatic species X	-ve	No change	+ve	+ve
Numbers of sites with sustainable populations of threatened non-charismatic species Y	-ve	No change	+ve	+ve
Numbers of sites with sustainable populations of unthreatened charismatic species Z	-ve	No change	+ve	+ve
Change in stock grazing days as a result of additional area of private land retired and under perpetual covenant	+ve	No change	-ve	-ve

Table 3b A contingent ranking schema to elicit valid WTP bids for biodiversity (ecosystem level)

Implications	Option A Current situation	Option B Hold the status quo	Option C Marginal change	Option D Non-marginal change
Levy on your property rates	$0	$20, $60, $100, $140	$20, $60, $100, $140	$20, $60, $100, $140
Management of existing DOC reserves	No change	No change	No change	No change
Change in area of protected, sustainably managed, culturally important diminished ecosystem A including sustainable populations of threatened charismatic species X	-ve	No change	+ve	+ve
Change in area of protected, sustainably managed but less culturally significant diminished ecosystem B including sustainable populations of threatened non-charismatic species Y	-ve	No change	+ve	+ve
Change in area of protected, sustainably managed, culturally significant but well represented ecosystem C including sustainable populations of unthreatened charismatic species Z	-ve	No change	+ve	+ve
Change in stock grazing days as a result of additional area of private land retired and under perpetual covenant	+ve	No change	-ve	-ve

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