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Valuation of Impacts of Incursions on Biodiversity: A Review of the Literature

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Valuation of Impacts of Incursions on Biodiversity: A Review of the Literature[1]

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Summary

This report provides an overview of the economic literature on invasive species with focus aimed at economic valuation. The report opens with an overview of economics concepts related to invasive species. Costs of invasive species are then discussed, followed by a description of the pathways and impacts of incursions relevant to valuation, economic models underpinning valuation of incursion events, valuation methods and methodological options, as well as prospects for benefits transfer. Conclusions and recommendations are then presented.

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Background

Pest, or alien invasion species (AIS), are an extremely important issue relevant to biodiversity today (Kelly, 2001)[2]. People transport invasive species all the time and the introduction of invasive species typically occurs in one of three ways. Some species are introduced purposefully, such as food crops like corn and wheat, which are alien species to most countries. Unintended escape of purposefully introduced species is common. For example, garden plants can spread to distant locations by seed dispersal or by spread of tubers. Another method of introduction is natural dispersal. Plants, animals and diseases spreading to other areas is a natural occurrence, an example is coconuts falling into the ocean and floating to other lands. However, the rate of spread of non-natives can be increased significantly by people importing non-native species. Finally, there are accidental introductions. An example of this is mice that board a ship in one port and depart at another port. The ship owners may have no idea when and how the mice came aboard or when and how they left (Ruesink *et al*., 1995).

Many non-native plants and animals can spread around the planet without being overly destructive. One example is the dandelion; it invades lawns and is found around the world. However, it does not cause any major problems. Other non-natives can completely eradicate other species. Kudzu, a creeping vine that was introduced to the United States to reduce erosion on stream banks is one of these. Planting kudzu along stream banks aided in the erosion problems, but once it established, it quickly spread to other areas. In some parts of the south-eastern United States, it has become one of the most visible plants. It completely covers entire forested areas, including tall trees, shading them entirely from light and destroying all smothered plants. Kudzu growth is so quick that in North Carolina one can literally hear it growing. In New Zealand, the possum, brought in from Australia for the fur industry, eats the leaves of many New Zealand tree species. Possums eat so much that they have made some native plants locally extinct and threaten to make them extinct nationwide if they are not controlled (reference needed here).

[2] Kelly (2001) ranks habitat loss and habitat fragmentation as the most important issue and AIS as the second most important issue.

One thing that most non-native species have in common is that they were either directly or indirectly introduced for economic reasons: corn to feed the world, morning glory to beautify gardens, koi carp for people's fish bowls and ponds, and rats and mice (indirectly) while transporting goods from one nation to another. It is for this reason that many believe that biological invasions need to be addressed with economic solutions (Amatya and McCoy, 2002; Chapman, 2002; Gundel, 2003; McNeely, 1995, 2001; Perrings *et al*., 2002). Born *et al*. (2005) reviewed 23 studies to determine whether economic studies are suitable to help decide what to do about biological invasions and concluded that the most important aspect of biological invasions is prevention. Turpie (2004) suggests that economic valuation of biodiversity benefits of alien control is useful, but noted the desirability of new and improved valuation methods. While it may be good practice to address invasive species issues with economic solutions, economists need to work closely with ecologists to understand potential pest impacts, to identify preferred ecological outcomes and to design control strategies (Evans, 2003).

It has been common for economic benefit-cost analyses of invasive species to include only market transactions where money changes hands; i.e., selling timber, planting trees, and paying contractors to poison animals. However, looking solely at market transactions does not account for the total economic value of resources, which requires consideration of both market values and non-market values. Non-market values include existence value (knowing a good exists), option value (knowing there is the option of using the good in the future) and bequest value (knowing it will be around for future generations). Changes in some ecosystem services affected by invasive species directly influence human welfare, such as the supply of fish available for commercial harvest. Other ecosystem services have indirect effects. Examples include habitat for bees that pollinate crops, nutrient cycling, and oxygen creation from carbon dioxide through the process of photosynthesis. The challenge for economic analysis is to identify how invasive species affect ecosystem services and then to identify how those changes affect human welfare.

Cost of Invasive Species

The cost of invasive species to agriculture has been studied intensively for some time (see for example, Taylor and Burt, 1984). When an invasive species impacts a commercial venture it is relatively straightforward to estimate cost because the value of lost production is market priced. Concern over invasive species has now expanded to encompass both market valued impacts (such as the value of lost production) and non-market valued impacts (such as a loss of biodiversity). Pimental *et al.* (2000) estimate the annual environmental and economic cost of non-indigenous species to the United States at US$137 billion annually with 50,000 non-native species having invaded the country. Reinhardt *et al.* (2003) estimate the annual direct economic damage and control costs for 20 invasive species in Germany at €167 m. In Canada, cost of control and eradication of 16 invasive species is conservatively estimated at between $13.3 and $34.5 b per year (MacIsaac, 2004). Zavaleta (2000) estimated the economic impacts of *tamarisk sp.* on ecosystem service because of municipal water loss, agricultural water loss, hydropower generation, and flood control losses. This resulted in costs of US$127~291 million in lost ecosystem services on 284 to 447 ha of land. Most estimates of cost do not include the value of lost biodiversity (Normile, 2004).

While the above cost estimates might appear impressive and sufficient to warrant action, economic analysis directs attention to both costs and benefits and, in particular, the changes in net benefits associated with policy options. Policy costs can be high. For example, Henri *et al*. (2004) evaluated costs of island restoration for globally threatened coastal birds in the Seychelles. Costs ranged from US$98 per hectare to $13,056 per hectare, depending upon the habitat in which restoration occurs. Transaction costs can be significant and may be the deciding factor in whether a pest management policy is implemented and whether it is successful or not. Nunez (2002) stated that the most important thing to accomplish to achieve invasive species reductions and preventions is to obtain public support, a notion supported by Saunders (1999). However, cost-benefit analysis can be useful, even with incomplete information. For example, Sinner (2003) studied *Didenmnum Vexillum*, an invasive species in Shakespeare Bay, Picton. His study indicated that containing spread while conducting eradication treatment trials and attempting eradication the following year was superior to other strategies evaluated, with total expected costs of approximately $173,000 yielding benefits of approximately $712,000 over 5 years. Sharov (2004) found that the benefits of eradication of gypsy moth, *Lymantria dispar,* in North America were greater than the benefits of slowing the rate of spread.

The overarching aim of this research is to create a method that can be applied by Biosecurity New Zealand to rapidly and accurately evaluate and rank projects aimed at protecting indigenous biodiversity from incursions of exotic pests and diseases. To achieve this objective an economic framework is developed that maps the exotic invader's pathway within the ecosystem, identifies exposure and risk, traces out the time dimension and links management responses to outcomes, costs and benefits.

Pathways and Impacts

A pathway is the route an invader takes. A vector is the means by which invaders move around the globe. Naturally occurring movements of invaders is not common. The first stage along the pathway is usually associated with human activity. Trade and travel, are common cause of incursions. Invasive species can arrive as freight, hitch hike on or inside imported foods, plants, livestock or pets, and in human travelers and their luggage. Incursions may be purely incidental, such as pests in wooden packing crates, and animals inadvertently trapped inside containers. Ballast water is a major means by which aquatic organisms enter. New trade patterns and new innovations in transport create opportunities for incursion. In addition to commerce and tourism, some invaders can take unusual pathways; for example ornamental wildlife, pets, aquaculture, and recreational boating. Far more is known about routes into a country than pathways within the country (Office of Technology Assessment, 1993). Once in, the invaders spread with and without human assistance. Time lags arise along the pathway (e.g. ballast water), in detection and in identification of invasive species, and in making decisions and taking preventative action.

A number of studies liken invasive species to a form of biological pollution. However the impact of invasive species more difficult to predict than the impact of pollutants because invasive species multiply, disperse in ways that are hard to predict, their interactions with other biota and ecosystems are difficult to see, and they can mutate. Distinguishing between good and bad invaders may not be straightforward. The impact of harmful invaders on ecosystems can range from wholesale changes and extinction to more subtle changes and increased biological homogeneity.

Early efforts to predict the vulnerability of ecosystems to invasions centered on the ecosystem's degree of resistance, which in turn was viewed as a function of diversity, isolation or level of human disturbance. However, nowadays ecologists consider [which ones?] both high and low diversity communities as being potentially invasible. One of the problems in identifying the determinants of invasion success is that the alternative invasion mechanisms are often confounded and no one mechanism can adequately determine the degree of invasibility.

Stokes *et al.* (2003) categorize invasive species according to whether the introduction has had a negative, positive, or no significant impact upon native biota. Negative impacts are further categorized according to the mechanism by which native biota are affected; for example, through competition, predation, alteration of habitat, introduction of parasites, and dilution of native gene pools. The vectors and pathways by which invaders are transported are numerous and result from a wide array of human activities that operate over a range of scales. For example, the primary source might be hull fouling or ballast water; secondary expansion can follow via a range of vectors including human activity (e.g. recreation) and natural expansion (e.g. water currents). The pathways for expansion are often multiple and success at various stages is stochastic in the sense that uncontrollable climate variables (e.g. temperature) can determine both the rate and spatial extent of invasion.

Without describing impact, Horan *et al.* (2002) model the firm as the carrier (vector); for example, incursion of aquatic species through discharge of ballast water. In contrast, Settle and Shogren (2002) accept the existence of an invasive species (lake trout) and trace its pathway and impacts using a system of differential equations. They show how lake trout not only affect the population of cutthroat trout (the endemic species) but also other populations that depend on trout, such as bears and pelicans; and of course, the population of visitors that enjoy the flow of services associated with Yellowstone National Park. Thus, there are both first-order and second-order values to account for. Moreover, if the feasible set of management options is to be modeled, then information on the opportunity cost of spending scarce resources on controlling invasive species is necessary. A comprehensive valuation exercise will encompass a range of both market and non-market valued goods and services.

According to Larson (2003) aquatic environments are more susceptible to invasion than terrestrial environments because they are more homogeneous – e.g. smaller variations in temperature – and water is an efficient vector for invasive organisms. This observation, coupled with the difficulties of monitoring aquatic habitats, means that when an aquatic invader is detected it might be very expensive to eradicate or control. Incursion of alien aquatic plants provides an illustration of the range of economic values at stake. Economic damage could include clogged irrigation water intakes (market valued impact), and reduced recreational values (non-market valued impact); and, environmental damage through loss of biodiversity (non-market valued impact).

Once an invader arrives, the dynamic response of the host ecosystem can be complex. When first introduced an aquatic invader increases biodiversity but eventually competition with native plants results in a decrease in diversity. The flow-on effects are potentially enormous when the invasive organism interacts with other species in the ecosystem. For example, native species might hybridize with the invader. Wiedenmann *et al.* (2001) cite the introduction of the macroalga *Caulerpa taxifolia* into the Mediterranean as an example of hybridization. In other cases, the invader can alter ecosystem processes such as hydrology and water quality. Vitousek *et al.* (1997) report that the Eurasian watermillfoil *Myriophyllum spicatum* can reduce water quality, with flow-on affects that impact fish habitat. Born *et al.* (2004) conclude that it is "fairly impossible" to predict whether a species will become invasive or not. Only once an invasion has occurred do the impacts become clear.

Only species that succeed in all transitions – introduction, establishment and dispersal - may become pests. Williamson and Fitter (1996) proposed the "tens rule" for plant species, illustrated in Figure 1 below. While the tenths rule is informative, it does not identify which species will successfully invade. Screening systems use information such as life history, biogeography, habitat characteristics, and weed history to classify species as potentially invasive. Daehler and Carino (2000) tested screening systems for non-indigenous flora of the Hawaiian Islands (United States). They found that for Hawaii an Australian template was the most promising method as it was 93% accurate in its predications, whereas a North American screening template was accurate to 82% and the South African version predicted only 60%.

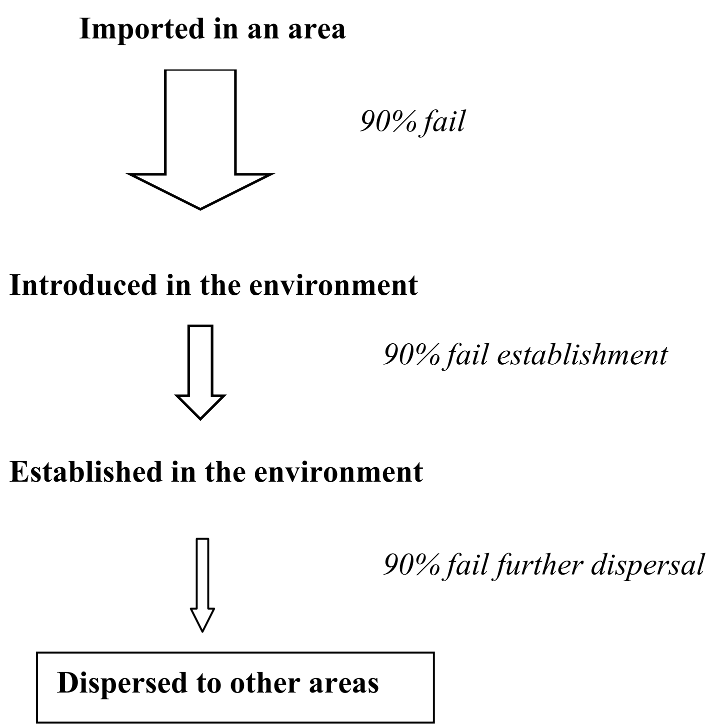

Figure 1: The tens rule (Williamson and Fitter, 1996)

Larson (2003) offers the following characteristics that contribute to a plant's invasiveness.

a) *Taxon affinity:* predicting future invaders is based on a list of species that are already invasive elsewhere. Although this might draw attention to problematic species its predictive value is low.
b) *Biological characters:* common invasive traits; such as continuous seed production, high seed output and lack of special conditions for germination, could provide a basis for predicting the likelihood of a species successfully invading an ecosystem. The geographic range of a species in its area of origin can be important because species with a large range are: (i) more likely to come into contact with vectors; and (ii) more likely to have highly tolerance spectrums.
c) *General list of species character:* high relative growth rate, small seed mass, low generation time, large geographical range, frequent contact with humans, niche position of invader similar to indigenous species (Larson, 2003).

Born *et al.* (2004) suggest that the following characteristics are associated with high habitat invasibility.

a) Biotic factors: such as vacant niches, absence of predators, low competitive resistance, and high biological diversity.
b) Abiotic factors: such as suitable climate and nutrient levels, water disturbance and quality, immigration of humans.

In summary, knowledge about the means by which invaders arrive exceeds knowledge about the invader's pathway, and apparently even less is known about how ecosystems respond to invasions. At a conceptual level, while the pathway might appear linear there are likely to be unexpected branches and interactions along the way. For example, invasive species A might have an obvious first round impact (clogging water intakes to irrigation schemes); the second round might be a reduction in habitat quality (lower oxygen levels); followed by a loss of biodiversity; and so on. This suggests that the valuation of any impacts on biodiversity should be broad rather than narrowly focusing on the invader itself, or its primary impacts.

Implications for valuation:

1. Impact of an invader on biodiversity will most likely be wide as opposed to narrow. The valuation framework should consider the invader within the context of a broad ecosystem.
2. Attention should be paid to the temporal pattern of ecosystem response/adjustment.
3. While first round effects are important, the valuation exercise should at least consider second round affects.
4. Both *ex ante* and *ex post* values are likely to be relevant.
5. The approach to valuation will, most likely, have to handle uncertainty.

Economic models

Because the overall aim of the project is to develop an economic framework we provide a brief overview of a range of models that have been developed to derive policy responses to species invasion. Rather than focusing on the analytics per se we use these models to draw out some implications for the valuation exercise. The following studies illustrate how modeling has been approached and the methodology used to incorporate the economic value of an incursion.

Theoretical models

Decision making under uncertainty:

Horan *et al.* (2002) argue that pre-incursion control of "biological pollution" does not easily fit within a risk management framework because: (a) although the likelihood of incursions increases with trade and travel the probability of establishment, spread, and damage is very low; (b) scientists argue that a probability density function can't be constructed from one-time events; and, (c) once established an introduction is most likely irreversible.

Horan *et al.* (2002) develop a model of invasion that is dependent upon the actions of individual firms. The invasive biomass of any firm is not directly under the firm's control due to the influence of environmental factors. However, the particular level of "emissions" is conditional on the firm's biosecurity choices. Successful establishment occurs according to some probability distribution; which is independent of introductions by other firms. The probability of a successful invasion increases with the number of firms and decreases with the level of biosecurity.

Static optimization: the objective is to minimize the expected social cost of invasion. Social cost is the sum of control costs and expected damages. This, of course, is the standard model applied to pollution and the first order condition is such that the marginal cost of biosecurity measures balances against the marginal expected benefits (reduction in damages).

Ignorance: the ability to derive risk management strategies using the above model is severely limited by ignorance and uncertainty. To get around this limitation, Horan *et al.* (2002) propose the use of a "potential surprise function" which might be based on expert opinion as demonstrated by Eiswerth and van Kooten (2002). Unlike probability density functions, potential surprise functions do not have to sum to one over events. The potential surprise function is a measure of disbelief and is of interest because it captures the public good nature of biosecurity actions by the individual firm. When the surprise function equals zero, the event is considered possible; when it is equal to one, the event is considered very unlikely. When an individual firm undertakes biosecurity measures the surprise function does not increase (i.e. the event becomes more unlikely) given no preventative actions by other firms. One insight of this approach is that the decision maker will focus on the least unbelievable loss or gain from their action; akin to the notion of the min-max approach in the sense of minimizing the potential surprise. The firm minimizes cost by balancing the marginal cost of a biosecurity measure against the impact of the measure on the loss (damages) plus the measure's impact on uncertainty.

From a policy point of view, this paper suggests that policy initiatives based on uniform technological mandates are a way of limiting uncertainty uniformly across firms.

Stochastic Optimization:

Variations in weather patterns and extreme weather events such as hurricanes are considered important factors in the arrival and dispersion of exotic species. Olson and Roy (2002) analyze the economics of controlling a biological invasion whose natural growth and spread is subject to environmental disturbances. The invasive species is assumed to grow according to its natural growth function which depends on a random process with outputs that enhance the conditions for incursion to increase. Although no specific functional form is used to model growth, the traditional "S" shaped growth function is possible within the structure of the model.

Objective function: minimize the present value of control costs, which depend on the removal of biomass; and damage function, which depends on the biomass that remains at the end of each period.

State equation: one first order difference equation is used to describe the relationship between the size of the invasion that remains after control and the natural growth function of the invasive species, which is a stochastic variable.

Control variable: removal of the invasive specie biomass.

Model results: If the invasion is a controlled biological invasion then an optimal policy is one for which the marginal costs of control are balanced against the expected discounted sum of marginal damages that are incurred if the species is not controlled. The authors define eradication as an outcome where the size of the invasion converges to zero in the long run. No particular time period is suggested. If the marginal costs of eliminating an arbitrarily small invasion are less than the damages, compounded indefinitely at a rate equal to the discounted expected growth of the invasion, then complete eradication with probability equal to one is optimal. Note that the size of the invasion must be sufficiently small. If the damages associated with controlling the invasion are too high relative to the costs of control then it is optimal to reduce the invasion to some finite level. The impact of adverse stochastic environmental conditions results in the optimal size of the remaining invasion being reduced below the deterministic result. Eradication is not optimal if the damages from an arbitrarily small invasion are less than the marginal costs of removing the entire invasion. In summary, this paper concludes that, for large invasions, the interaction of costs and damages with the discount rate and the invasion growth rate determines whether or not eradication is optimal.

Deterministic dynamic optimization

Early work focused on building dynamic optimization models to derive optimal policies for invasive species management in cropland (see for example, Taylor and Burt, 1984). In these cases estimates of damage functions are derived from the impact on output that is market valued. In contrast, more recent applications have attempted to include estimates of the value of lost biodiversity. For example, Burnett *et al.* (2005) provide a useful case study that illustrates the application of a dynamic programming model to determine optimal policies to control an invasive weed and Brown Tree Snake in Hawaii. The invasive species are represented by a standard logistic model and the damage functions include estimates of the value of lost biodiversity. The objective is to identify least cost control policies.

<u>Native-exotic incursions Yellowstone Lake:</u>
Settle and Shogren (2002) develop a constrained dynamic optimization model in which the objective function is to maximize total benefits subject to ecosystem and household budget constraints. The case study is set in Yellowstone National Park. The ecosystem component of the model is based on a predator-prey relationship between an exotic predator (lake trout) and the local native trout (cutthroat trout). A range of policy alternatives are assessed under different discounting scenarios. Key components of the model are summarized below.

Objective function: maximize visitor utility; visitors derive utility from: targeting the native species, catching the exotic species, the flow of public good benefits from other species (e.g. pelicans) that feed on the native species; visitors are assumed myopic; agency expenditures on improving the flow of public goods enters the objective function as a control variable.

State equations: differential equations describe: the interactions between the species, including other species, such as pelicans and bears; agency expenditures on intervention, such as lake trout trapping; and the flow of public goods from the park.

Control variables: two activities available to park management viz. allocation of their budget between killing lake trout and improving roads in the park.

Model results: The model shows that the integration of the human system with the ecosystem results in a higher cutthroat population. This result arises from visitors substituting away from targeting cutthroat to other park attractions. Interestingly, the current policy of killing lake trout turned out to be non-optimal because visitors cared more about protecting roads than protecting cutthroat. If existence values, as opposed to use values, were entered into the model, and the discount rate was lowered then the policy of lake trout trapping becomes economically attractive.

Stochastic dynamic optimization

Eiswerth and van Kooten (2002) recognize the role of uncertainty in modeling the economics of invasive plant species. They develop a stochastic dynamic optimization model to identify economically optimal management choices from a set of potential options, including eradication. The model is applied to the spread of an exotic weed called starthistle in California. One distinguishing feature of their model is the use of fuzzy logic to capture the often qualitative judgments of scientists. For example, a scientist might classify an incursion as “minimal’ in one location but “high” in another.

Objective function: each producer is assumed to maximize the present value of future net revenue; agricultural yield is expressed as a decreasing function of the extent of infestation; the control variable is the producer's choice of technology to control the weed.

State equations: the equation of motion for the invasive species stock is a function of the previous period stock level and the control policy plus a random variable. This is, in essence, a Markovian transition condition that maps current levels of the invasive species and the affect of controls into the next period. Transition probability matrices are defined for each of the five policy options considered. The transition probabilities were based on a qualitative survey of "expert judgments" obtained from weed scientists, farm advisors and other specialists. The experts also gave their impressions as to the impact of starthistle on agricultural output.

Control variables: included do nothing; one-time chemical application; a range of alternative control measures such as mowing, burning or grazing; site vegetation.

Model results: not surprisingly, as the productivity of the land increases the optimal solution gravitates toward the more expensive options. The authors find that eradication is not optimal, being inferior to strategies that controlled the spread of starthistle. Perhaps one of the more interesting aspects of the approach was the use of experts to elicit linguistic descriptors of growth and potential damages.

Other studies to include stochasticity in bio-economic evaluations of pest invasions include Leung *et al*. (2002) and Finnoff *et al*. (2005) who studied zebra mussels (*Dreissena polymorpha*) in the Mid Western Lakes of the United States.

Implications for valuation:

1. Practical application of an economic model requires an assessment of damages as a function of changes in the ecosystem and policy responses.
2. Time is an important dimension, the state equations used to model ecosystem responses link up changes in the value of biodiversity.
3. Empirically, it might be important to explicitly link the growth rate of the invasive specie with the damage function. Linear extrapolation is likely to misrepresent the economic impact.
4. In general terms, environmental conditions – winds, temperature, humidity – can impact the damage function.
5. The ecology of the system extends beyond the invasive specie itself and linkages should be made to other impacted species.
6. Interventions should be plausibly linked to pathways and impacts on target species and associated components of the ecosystem.
7. Uncertainty over damages.
8. Although expert judgments might be able to provide potentially useful information it should be considered alongside data from other sources.

Biodiversity incursion valuation methods

Any study that attempts to value impacts of, and responses to, biodiversity incursions must address some fundamental questions before it is possible to determine which methods are appropriate.

Firstly, there is the matter of what should be valued. Valuation is never absolute; it entails measurement of changes in value that occur between one reference state of the world and another. Consequently, there is no "value of biodiversity" as such, but it is possible (at least in theory) to measure the change in the value of biodiversity. However, in an ever-changing world it is not always obvious what the counterfactual should be – it may not be appropriate to use the status quo in every case.

Understanding of how ecosystems (including human components of those systems) function is imperfect, even for acknowledged experts. Understanding by the general public is likely to be poor. However, valuation is a process for incorporating public values into the decision framework. This has implications for two key matters; the valuation target and whose values to measure.

The valuation target is the item whose values are sought, which can be either policies or outcomes. Valuation of policies entails measurement of support for a particular activity. An example is aerial spraying to eliminate Painted Apple Moth. The values that people place on the policy will be determined by, *inter alia*, their impressions about its likely success in eradicating the moth, perceptions about health impacts of the sprays, expectations about externalities (such as aircraft noise), and perceptions about the nature, extent, and desirability of impacts of the Painted Apple Moth. In other words, people's values for a policy are some weighted combination of the probabilities and values they ascribe to particular outcomes of that policy. Outcome valuation measures the values associated with specific changes, such as noise, health and ecosystem functions. In order to derive a policy valuation, the analyst must then link the policy to changes in salient outcomes.

Policy valuation confounds perceptions with values. If members of the public do not have correct information on the likely outcomes of the policy their values of the policy will not reflect the outcomes of policy implementation. Continuing the Painted Apple Moth example, if people believe that the spray used is highly toxic when it is not, they will evaluate the policy less positively than if they believed the spray to be benign. This is independent of how the biodiversity, recreational, commercial or other impacts of the Painted Apple Moth are perceived.

Misunderstanding amongst the public raises the question of <u>whose</u> values should be measured. One line of argument is that valuation should be undertaken by experts who have strong understanding of likely impacts. However, this approach is a valid way of representing community values only if experts have the same values as the rest of the community. Experts, by their nature tend to be better educated, higher earning members of society than the norm and also self-select into particular specializations that are of interest to them. Ecologists, for example, become ecologists because they are highly interested in the natural world and its functioning, which may lead them to value biodiversity preservation more highly than other community members.

Another solution to this problem is to provide valuation participants with superior information. In this way, community values enter the valuation process, but valuers' expectations about outcomes are more realistic. However, it may not be possible to change understanding without affecting values. Valuation participants subject to such educative processes may change their values along the way, possibly simply because of the value cues provided by the process. Educating the valuers also means that the values measured do not represent the values of the community at large so that policies that are acceptable to the educated group may be unacceptable to the wider community, presenting potential political acceptability issues for proposed policies that have passed the valuation test.

Another fundamental to be considered in any valuation exercise is <u>when</u> impacts will occur, how they are valued at the time, and how those values are converted to a common numeraire. Biodiversity incursions may start as localized outbreaks that could take years to spread and their impacts may take even longer to manifest themselves – other incursions may have almost immediate effects. Similarly, responses to biodiversity incursions may be capable of elimination of the total threat immediately, or may only slow it down (or in the worst case have no affect at all). Values associated with biodiversity incursions may change through time. For example, leisure activities may change in value through time as incomes and lifestyles change. Attitudes and values about the natural environment may also change through time.

There are two main groups of tools for valuing non-market monetary impacts of biodiversity invasions; revealed preference and stated preference methods.

Revealed preference methods

Revealed preference methods rely on observations or recall of actual behaviors to infer non-market values. While revealed preference methods appears less subjective than stated preference methods, value inference using revealed preference methods relies on application of models that utilize analyst expectations about the ways that people make decisions, so they are far from value-free. Revealed preference methods can value only things for which there is an associated market. Examples include recreational activities, for which it is necessary to invest in time and travel services to enable participation (Travel Cost Methods). People's choices in housing markets can be used to assess the values they place on some environmental attributes, such as noise pollution and water quality (Hedonic Price Methods). Application of revealed preference methods can be constrained by access to data or insufficient variation in the data for model estimation. Events that have not occurred cannot be valued using these approaches.

Stated preference methods

Stated preference methods rely on responses to hypothetical scenarios. These scenarios are a form of controlled experiment in which the valuation analyst controls variables and measures participant responses. Normally, those responses entail revelation of a preferred choice from a set of alternatives presented to the individual, or some kind of ranking or rating of choices. Commonly used stated preference methods include contingent valuation in many different guises, contingent behaviour, choice experiments (choice modeling), contingent ranking, contingent rating, and conjoint methods. The great bulk of stated preference studies have employed contingent valuation in various forms, although this approach has been increasingly supplanted by choice experiment methods in recent years because of increased flexibility and perceptions that choice experiments overcome many of the undesirable aspects of contingent valuation (Bennett and Blamey, 2001; Hanley *et al*., 1998, 2001).

Because of the high degree of analyst control, stated preference approaches do not suffer the limitations of revealed preference methods. They can measure values not associated with existing markets (e.g. existence values), they can value outcomes that have not previously been experienced, and appropriate experimental design can be used to collect data suitable for demand identification purposes. Stated preference methods are not universally accepted and have been subject to considerable criticism (Hausman, 1993; Haddad and Howarth, 2006). Smith (2006) provides a comprehensive overview of these concerns but concludes that stated preference approaches will continue to be important because missing markets preclude revealed preference, although he urges analysts to seek more opportunities to calibrate preferences wherever market information is available. This theme is supported by Whitehead and Blomquist (2006), who conclude that "[f]or many government projects and policies the CVM is a crucial and necessary component of benefit-cost analysis".

Benefits Transfer

Benefits transfer is a process that transfers environmental valuation estimates obtained in one situation to a particular case study. Benefit transfer is the practice of adapting available economic value estimates of a quality or quantity change for some environmental resource to evaluate a proposed change in some other "similar" resource. In these situations, the policy analyst takes the results or data from the context of one or several existing studies (defined in terms of their time frame, location, environmental resource, environmental quality change, and/or their affected population) and transfers them to a context that is specifically relevant for a policy of interest.

For example, if there is an estimate of the loss in total economic value associated with Land Disturbance Activities in region A, then the process of benefits transfer would adapt and apply the estimates to case study B. Benefit transfer has two main potential advantages: speed and cost. However, difficulties are likely to arise where there is a dearth of relevant high quality studies to draw on; where estimates are needed for new kinds of policies and projects; or where there are important differences between the context of past studies and the context of the analysis.

Ultimately, the intended use of the benefit estimate determines whether benefit transfer is appropriate and provides adequate reliability. When precision matters in the intended policy application, the appropriateness of benefit transfer is questionable – direct benefit transfer involving seemingly similar sites can produce notable errors in benefit estimates (Kirchhoff *et al*., 1997).

The analysis of past applications of environmental valuation techniques shows that these do not bode well for benefit transfer studies. Valuation exercises tend not to be designed with future benefit transfers in mind, but rather to explore new methodologies, survey design, data modeling, or to test specific hypotheses. Progress could be made by constructing a database of all environmental benefit estimates including details of the modeling procedures used, and all relevant assumptions; and requiring researchers to bear transferability in mind when undertaking valuation studies. A broad code of practice for the conduct of benefit studies should be drawn up to ensure that the outputs from valuation studies would be usable in future benefit transfers (Willis and Garrod, 1995).

A Valuation Typology

Pearce (2001) identifies several aspects of value for biological resources. These include:

i. Direct use values (tourism, recreation, harvest, information value, pharmaceuticals)
ii. Indirect use values (ecosystem resilience, ecosystem services), and
iii. Non-use values

Table X (Pearce, 2001, modified from Bann (1998)) illustrates the range of values that can be associated with a mangrove resource.

Table X: Values associated with mangroves

Direct use	Indirect use	Option	Non-use
• Timber, fuelwood, charcoal • Fisheries • Forest products: food medicine, wildlife, etc. • Agricultural resources • Water supply • Water transport • Genetic resources • Tourism and recreation • Human habitat • Information	• Shoreline, riverbank stabilization • Groundwater recharge/discharge • Flood and flow control • Waste storage and recycling • Biodiversity maintenance • Provision of migration habitat • Nursery/breeding grounds for fish • Nutrient retention • Coral reef maintenance and protection • Prevention of saline water intrusion	• Future direct and indirect values	• Cultural, aesthetic • Spiritual, religious • Global existence value

Pearce concludes (2001, p. 39) "it is not clear that the many willingness to pay studies of biological resources such as wetlands, forests, endangered species, etc. are also studies of biological diversity. Studies tend to focus on individual ecosystem services, a given ecosystem, or particular species. While people may be valuing these resources because they 'represent' diversity, we cannot be sure … The reality is that, despite the massive growth of economic valuation literature in recent years, we still have little idea of the value of diversity *per se*, even if we know a lot about the local use values of biological resources."

The issues identified by Pearce are even more pressing when it is considered that biological invasions can have effects in different biomes and impact upon many different environments, species and activities.

Many ecosystem services are valued in the market place. For example, the value of protecting agricultural land, homes, and industry from flooding are readily measurable using market signals. However, other ecosystem services are not amenable to market measures. Examples include ecosystem functioning, water purification, species habitat, scenic amenity, and so forth.

Outcomes arising from biological invasions include displacement, predation, and competition with other species. These factors may be important matters where species are at risk, such as invasion of island bird sanctuaries by predatory mustelids. Another form of existence value is associated with preservation of historic artifacts and buildings. An example is the increased rate of degradation of historic buildings because of inhabitation by rats and pigeons.

Analysis of economic impacts can take several different directions. Possibilities include:

- Analysis by valuation method, and
- Analysis by type of value.

This study will proceed by reviewing studies grouped by valuation methods, both in the international literature and in New Zealand. Subsequently, conclusions will be drawn about different types of value.

Examples from the literature

Literature is available that measures components of value that can change because of biodiversity incursions. Principal amongst these are recreation use values, ecosystem function values, and existence values for flora and fauna. This literature illustrates that people do place significant economic values on environmental changes that may be attributable to biological invasions. A great deal of the literature does not focus specifically on the valuation of the invasive species but on determining the importance of specific impacts of biological invasions. Both types of study are discussed. For clarity of exposition, details of the studies discussed below are reported in the appendices.

Contingent valuation studies

Appendix 2 list a large number of contingent valuation studies. These studies have been conducted in different countries, including the United States, Australia, Canada, China, Greece, Holland, Sri Lanka, South Africa and Great Britain, yet they consistently indicate that members of those different communities are willing to pay significant amounts of money to protect or enhance natural environments.

The studies reported in Appendix 2 generally do not attempt to value impacts from invasive species. Some studies value policies to prevent spread of introduced species (Bell and Bonn, 2004; Jetter and Paine, 2004; Tumaneng-Diete *et al*., 2005). Others focus on protection (Bandara and Tisdell, 2004; Cherry *et al*., 2006; Christie *et al*., 2004,; Hoehn and Loomis, 1993; Jakobsson and Dragun, 2001; Kramer and Mercer, 1997; Langford *et al*., 1998; Pate and Loomis, 1997; Stanley, 2005), and others address the value of restoration (Chambers and Whitehead, 2003; Hoehn and Loomis, 1993; Kontoleon and Swanson, 2003; Kotchen and Reiling, 2000; MacMillan *et al*., 2001;Reaves *et al*., 1999). Whereas many studies address outcomes of policies (e.g. Kontoleon and Swanson, 2003; Kramer and Mercer, 1997), others value the policies themselves, even though the outcomes may not be known with certainty (Giraud *et al*., 2002).

The studies address different sources of value. Some focus on ecosystem services (e.g. Loomis *et al*., 2000), others on existence values for specific locations or ecosystem types (Bennett, 1984; Turpie, 2003; MacMillan *et al*., 2001, Cherry *et al*., 2006), others on individual species (Jakobsson & Dragun, 2001; Hoehn and Loomis, 1993; Bandara and Tisdell, 2004; Chambers and Whitehead; Kotchen and Reiling, 2000), some on habitat (Christie *et al*., 2004), and others on recreation (Nunes and van den Berg, nd).

Some studies have valued the same item for different populations. Christie *et al*. (2004) found differences in WTP for the same environmental enhancements in different parts of England. While residents of the United States were WTP $100 annually for expanding a recovery program for steller sea lions in Alaska, residents of Alaska were WTP only $40. However, residents of sea lion habitat areas had negative mean WTP (-$255). Not everyone is in favor of environmental enhancements, particularly if they diminish use values – in this case sea lions were seen as a threat to commercial fishing profitability. A similar effect was illustrated by Chambers and Whitehead (2003) who found that non-locals were WTP a one time amount of $21 per household for increasing the Minnesota wolf population to 1600 animals, whereas locals were WTP only $5.

In their study of WTP for protection of Mediterranean monk seals Langford *et al*. (1998) were able to separate out different components of value. Option value was estimated to be more than five times the magnitude of use value and existence value was 14 times the magnitude of use value. Using use value as an estimate of total value of protecting the seals would have captured less than 5% of total value.

Methods used for obtaining desired conservation outcomes can be important. In their study of the benefits of controlling eucalyptus snout beetle in Ventura County, California, Jetter and Paine (2004) measured annual WTP for seven years of $23 per person when Carbaryl insecticide was proposed as the control agent, compared with $131 when using Btt insecticide. This result highlights the other objectives that may feature in people's valuation of control policies and the difficulties of transferring benefit estimates when different management programmes are used to obtain the same outcome.

New Zealand contingent valuation studies

Mortimer *et al*. (1996) conducted a contingent valuation study by telephone survey of Auckland households to determine preservation value for offshore islands, most specifically, Little Barrier Island. They found 54% of respondents believed that preservation of endangered species was the most important reason for conserving offshore islands. Based on the population of Auckland alone, they measured WTP for maintaining conservation activities on Little Barrier Island to be $8 million per household per year, or $30 per household for a once only payment.

Greer and Sheppard (1990) studied whether New Zealand research into biological control of *Clematis vitalba* is justified. They found 91% of respondents were WTP something for research into biological control. Their estimate of $50 per household for a one time payment indicates national WTP of approximately $111 million.

Kerr and Cullen (1995) sampled people in the Nelson area of New Zealand to evaluate public preferences for possum-control budget in Paparoa National Park. They found that the most important aspect of possum control expressed by the public were the protection of vulnerable rare species. Their estimate of annual mean WTP for possum control was $300 per adult.

Lock (1992) looked at the value of possum control in the Manawatu-Wanganui Region. There was a high degree of awareness of the possum problem in the area and over 80% of people were WTP something to control the possums. Factors that affected WTP included belief that possums were a problem, residence in an urban or rural area, household income, and occupation (farmer vs. non-farmer).

Fahy and Kerr (1991) studied a royal albatross colony at Taiaroa Heads in Otago. Economics students at Otago and Auckland Universities were WTP $22 annually to fund research into albatross chick fatalities.

Beanland (1992) measured mean WTP of $9 per household per year for implementation of an indigenous forest policy in the Manawatu-Wanganui Region's Aorangi Awarua Forest. Data collected using a mail survey showed that 52% of respondents felt that preserving the forest was very important and 24% thought it was moderately important, while 66% thought it was a priority to protect wildfire and its habitat.

Studies by Moore (1998) and Williamson (1997) indicate that New Zealanders are WTP to protect the quality of the coastal environment, and White *et al*. (2001) illustrates concern about aquifer quality. The study by Guria and Miller (1991) measured the value of a statistical life. There has been additional study to update their figure, but results are not public. However, the value of a human life is significant for evaluation of impacts of introduced species that can affect human health and morbidity, either directly or as vectors for disease.

Several contingent valuation studies provide indicators of benefits from a range of recreational activities (Kane, 1991; Kerr, 1996b; Meyer, 1994; McBeth, 1997; Walker, 1992; Wheeler & Damania, 2001).

Attribute-based methods

Choice experiments and random utility models (Attribute–Based Methods: ABMs) are increasingly being used to identify attributes of recreational experiences and environmental amenity that affect value.

Recreation studies will prove useful for identifying the likely impacts on recreational activities of a biodiversity invasion. For example, the value of fishing is likely to be dependent upon, *inter alia*, the abundance and supply of sport fish. Both of these attributes may change because of ecosystem effects of an invasion. Other important aspects of the fishing experience might include scenery, isolation, congestion, and so on – factors not affected by a biological invasion. The change in recreational values can be measured by ABMs once the biological invasion-induced changes in recreation-relevant attributes have been identified.

Hatton-MacDonald and Morrison (2005) derived values of different habitat types for South Australia, ranging from $0.72 for scrublands through $1.02 for grassy woodlands and $1.40 for wetlands. Studies of this type might be important where pest invasions transform one type of habitat into another, say where possums or deer transform forest to scrub or grassland.

Jeff Bennett, John Rolfe, Stuart Whitten, Vic Adamowicz, Peter Boxall,

Garrod & Willis (1997) applied contingent ranking to measure desirability of converting 3,000 hectares of intensively managed remote UK conifer forest to alternative management regimes. While mean WTP for each of the alternatives, involving different degrees of improvement in biodiversity, was positive, the values obtained were small. Notably, the lowest valued alternative was complete restoration to native woodland. Native woodland was preferred to the existing intensive commercial management, but was less valuable than more modest changes.

Hanley *et al*. (2003) investigated attitudes to goose hunting with a choice experiment. They found that households were WTP £9 per year for 10 years to stop shooting all geese at Islay in Scotland. A higher value (£12) was obtained for a policy that would stop people shooting only endangered geese. This result is suggestive of positive values associated with shooting non-endangered geese, possibly reflecting the value of goose hunting for recreation, or even pest-status for non-endangered geese.

Bienabe and Hearne (2004) surveyed Costa Ricans and foreign tourists about support for nature conservation and scenic beauty through payments for environmental services. Costa Ricans were willing to pay only \$0.33 per month for protecting biodiversity and less than that (\$0.25 per month) for scenic beauty. Tourists, on the other hand, would pay a one time fee of \$6.77 for protecting biodiversity and a \$3.36 one time fee for scenic beauty.

New Zealand choice experiment studies

Values of changes in attributes of Auckland streams were the focus of a choice experiment undertaken by Kerr & Sharp (2003). Amongst other attributes, the study found that North Shore households were WTP \$10 per year to prevent the loss of a single native fish species from a high quality stream, whereas they were WTP \$3* for an additional native fish species in a degraded stream. Loss of fish habitat on a high-quality stream was valued at \$1.38 km^{-1*} and additional fish habitat on a degraded stream was valued at \$13 km^{-1}. Corresponding figures for South Auckland were: quality stream fish species \$5*; degraded stream fish species, \$5*; quality stream fish habitat, -\$3 km^{-1*}; degraded stream fish habitat, \$6 km^{-1}. The value estimates marked with an asterisk are not significantly different from zero, indicating some of the potential problems arising from reliance on point estimates. It is notable that both communities had significant positive values for restoration of fish habitat on degraded streams.

Travel cost studies

There are many thousands of travel cost studies of the value of recreational activities. Specific studies are not described here, but there is a clear indication of the significant benefits obtained from recreation in natural environments.

New Zealand travel cost studies

Appendix 3 reports results from several travel cost studies of values of New Zealand recreation activities. The case studies have adopted several variants of the travel cost method and applied them to a wide range of activities. Estimates range from less than \$2 per recreational visit (Riley & Scrimgeour, 1991; Walker, 1992) to as high as \$400 per recreational visit (Woodfield & Cowie, 1977), indicating that loss of recreational amenity has the potential to generate very large costs when aggregated over the large number of people recreating in natural environments..

Hedonic studies

Hedonic studies of the types of impacts caused by invasive species are rare. One study (Holmes *et al*., 2005) values one impact of an invasive species, estimating the value of a 10% increase in hemlock woolly algelid-induced defoliation of hemlock trees in Sparta, New Jersey at US\$7300 per house at risk.

US beach studies???

There are no New Zealand studies of relevance to invasive species.

Benefit transfer studies

Loomis and White (1996) applied meta-analysis to responses to twenty different contingent valuation studies of rare, threatened and endangered species. They report values ranging from US$6 per annum (1993 dollars) for striped shiners to $95 for spotted owls. Lump sums ranged from US$13 for Arctic grayling to US$254 for bald eagles. Their regression models explained about 68% of variation in WTP, leading Loomis and White (page 204) to conclude that meta-analysis can "provide a rough first estimate to determine whether the benefits are likely to be much larger or much smaller than the costs." One advantage of meta-analysis is identification of human and species-related factors that affect values. An important finding from this study is that people value marine mammals and birds significantly higher than other species. Site visitors are willing to pay more than non-visitors.

While Loomis and White (1996) are enthusiastic about the use of benefits transfer, at least as a rough filter, other authors are somewhat more skeptical. Navrud (2001, pp.66-68) warns, "One should be very careful when transferring estimates, particularly for complex goods, goods with a large non-use component, or both, for instance for ecosystems and biodiversity. … Unit values for non-use values of e.g. ecosystems may be even more difficult to transfer than recreational (use) values for at least two reasons. First, the unit of transfer is more difficult to define. While the obvious choice of unit for use values are [sic] consumer surplus (CS) per activity day, there is greater variability in reporting non-use values from CV surveys …Second, the WTP is reported for one or more specified discrete changes in environmental quality, and not on a marginal basis." Navrud cites evidence from several studies in support of his skepticism, including Loomis (1992) who found he could not transfer sport fishing benefits between different parts of the USA, Bergland *et al*. (1995) and Brouwer and Spaninks (1999) found statistical evidence of lack of transferability despite small percentage differences in values at different sites, and Downing and Ozuno (1996) who rejected transferability of recreational fishing benefits between bays within Texas. The OECD (2002, pp.125-126) concluded "the approach [meta-analysis] revealed some high risks in transferring estimates. … The study therefore suggests considerable caution in adopting benefits transfer techniques at this stage."

Brouwer *et al*. (1999) used more than 100 estimates of WTP from thirty studies to develop a meta-model of wetland values. Study location had a significant effect on values, with North Americans willing to pay more than Europeans. Four sources of wetland value were identified, with biodiversity being less valuable than flood control, but more valuable than either water generation or water quality.

Shrestha and Loomis (2001) have tested the validity of transferring recreational values across international boundaries.
McLeod

<u>*New Zealand benefit transfer studies*</u>

In their study of Auckland streams Kerr & Sharp (2006) evaluated transfer of benefits between North Shore and South Auckland.

Ball *et al*. (1997) transferred recreation values from the United States, adjusting for exchange rate and inflation, to estimate the value of recreation in Auckland regional parks. The value they derived, \$11 per user day, is very close to recreation-day values (\$10) estimated for Wellington regional parks using contingent valuation (Kerr, 1996a). This does not provide a test of benefit transfer because there is no reason to believe that regional parks in different parts of New Zealand should have the same value.

Kaval *et al*. (2003) transferred values to derive an estimate of the benefits from a proposed park at Te Kouma, on the Coromandel Peninsula of \$28 per person-day.

Fishing is an activity that has been the subject of several different non-market valuation exercises in New Zealand. Estimates are available from travel cost and contingent valuation studies. Kerr (2004) used information from angling valuation studies on the Tongariro, Greenstone, Rakaia and Rangitata rivers to derive a mean value of \$39 (2003) per angler-day. The same study estimated a mean value for non-angling recreation activities of $21 per activity-day.

Values from New Zealand outdoor recreation studies are presented in Table 3. User-day values have been derived by dividing trip values by an estimate of mean trip length. The point value estimates in Table 3 are diverse, with a mean of about $21 per recreation day for non-fishing activities. The range is $1 to $63 per recreation-day for non-fishing activities.

Table 3: New Zealand Outdoor Recreation Benefits

Activity	Location	Principal Author [reference]	Valuation method	Approximate benefit per user day (2003$NZ)
		Fishing		
Freshwater fishing	Rakaia	Leathers [26]	TCM	$33
Freshwater fishing	Greenstone/Caples	Kerr [21]	CVM	$49
Freshwater fishing	Tongariro	McBeth [27]	TCM	$35
Freshwater fishing	Rangitata	Kerr [22,23]	TCM	$38
Average for fishing (sd=standard deviation)				**$39 (sd=$8)**
		Other Activities		
Lake recreation	Lake Tutira, Hawkes Bay	Harris [14]	TCM	$27
Roadend camping	Otaki Forks, Wellington	Kerr [24]	CVM	$7
National Park	Mt Cook National Park	Kerr [25]	TCM	$63
Canoeing	Wanganui River	Sandrey [35]	CVM	$26
Mountaineering	Mt Cook National Park	Kerr [19]	TCM	$38
Deer hunting	Oxford Forest, Canterbury	Nugent [30]	TCM	$16
Deer hunting	Greenstone & Caples Valleys, Otago	Kerr [21]	CVM	$31
Deer hunting	Kaimanawa & Kaweka Forests	Sandrey [36]	TCM	$22
Tramping	Hollyford Valley, Fiordland	Kane [16]	CVM	$24
Tramping	Greenstone & Caples Valleys, Otago	Kerr [21]	CVM	$15
Tramping	Kaimanawa & Kaweka Forests	Sandrey [36]	TCM	$27
Park recreation	Kaitoke Regional Park	Walker [41]	TCM	$13
Park recreation	Wellington regional parks	Kerr [20]	CVM	$12
Park recreation	Auckland regional parks	Ball [1]	BT	$13
Forest recreation	Bottle Lake, Christchurch	Walker [42]	TCM	$2
Forest recreation	Kauaeranga Valley, Coromandel	Riley [31]	CVM	$1
Average for other outdoor activities (sd=standard deviation)				**$21 (sd=$15)**
Other values mean/fishing values mean				**0.54**

Note: All money values have been adjusted using the consumers' price index.
TCM: Travel cost method CVM: Contingent valuation method BT: Benefit transfer

Kaval *et al*. (2003) have used a different methodology to assess the value of New Zealand outdoor recreation activities using results from existing New Zealand non-market valuation studies. Their results are not dissimilar from those of Table 3, with mean values of fishing and of recreation activities in general both being $28 per person per day.

Conclusions

Whether deliberate or accidental, there will always be introductions of non-native animals and plants to New Zealand. How these non-native species are managed will determine what New Zealand will look like in the future.

Several themes have emerged from this review.

1. Economic analysis can be useful for prioritizing resource allocations for management of invasive species.
2. Non-market impacts can be significant and should be included in benefit-cost analyses through non-market valuation.
3. Suitable economic valuation methods for deriving values to be used in benefit-cost analyses include: contingent valuation, choice experiments, travel cost analysis, and in some instances benefit transfer.
4. New Zealand studies are similar to studies conducted elsewhere in that they show the community is collectively willing to pay significant amounts of money to restore or protect natural environments.
5. Non-use values can be large compared to use values.
6. Studies of individual species or sites may not provide a great deal of information necessary to evaluate the impacts of an invasive species.
7. Economists need to work together with ecologists, especially those ecologists focusing on spatial modeling and invasive species spread predictions. This will help us to gain understanding of specific invasives and how they threaten an area. What are the chances of a particular species coming into the country, and if it does enter, where and how will it spread? This information can be modeled with geographic information systems, such as the Australian Screen System.

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Appendix 1: Website References

The Australian Screen System:

http://www.aqis.gov.au/docs/plpolicy/wrmanu.htm

Environment Canada

http://www.cws-scf.ec.gc.ca/publications/inv/cont_e.cfm

Invasives section

The Global Invasive Species Programme

http://www.gisp.org/

Here there are the IAS and ISSG (global invasive species) databases for searching taxa, as well as references to other publications on IAS.

Valuation of Ecological Benefits: Improving the Science Behind Policy Decisions

http://yosemite.epa.gov/ee/epa/eerm.nsf/vwSER/96291273F5DF6C2085256F9B00733175?OpenDocument

National Centre for Environmental Economics, US Environmental Protection Agency.

The Ecovalue Project

http://ecovalue.uvm.edu/evp/default.asp

Biodiversity Economics

http://biodiversityeconomics.org/library/index.html

Environmental Valuation & Cost-Benefit News

http://envirovaluation.org/

Appendix 2: International Valuation Studies

Author(s)	Method	Year	Period	Mean Value (point estimates)	Item valued
Bandara & Tisdell (2004)	CV		5 years annual	1322 rupees/household	Conservation of the Asian elephant in Sri Lanka
Bell & Bonn (2004)	CV		Annual	US$14/individual lake user	Weed control stamp for aquatic weed control in Lake Istokpoga, Florida
Bennett (1984)	CV		One time	AU$27	Maintain existence benefits at Nadgee Nature Reserve, Australia
Blamey *et al*. (2000)	CE	1997	Annual	One endangered species: AU$11.39 % reduction in population of non-threatened species: AU$1.69 % loss in area of unique ecosystems: AU$3.68	Brisbane residents' WTP for Desert Uplands attributes
Blamey *et al*. (2000)	CE		Annual	One endangered species: AU$17 % reduction in population of non-threatened species: AU$2.51	Brisbane residents' WTP for Desert Uplands attributes
Chambers & Whitehead (2003)	CV		One time	Locals: US$5/household Non-locals: US$21/household	Increase Minnesota wolf population to 1600 animals
Cherry *et al* (2006)	CV		Annual	US$11/park visitor	Protect Yellowstone Lake ecosystem
Christie *et al* (2004)	CV		Annual	£55/individual (Cambridgeshire) £47/individual (Northumberland)	Habitat creation
Christie *et al* (2004)	CE		Annual	£93/individual (Cambridgeshire) £98/individual (Northumberland)	Protection of rare and common familiar species
Christie *et al* (2004)	CE		Annual	£36/individual (Cambridgeshire) £91/individual (Northumberland)	Protection of rare familiar species
Foster & Mourato (2000)	CR	1996	na	£0.0525 per loaf of bread	Number of farmland bird species in a state of serious long-term decline as a result of pesticide use in cereal cultivation
Garrod & Willis (1997)	CR	1995	Annual	£0.32: Basic biodiversity conservation £0.54: Desired biodiversity conservation £0.19: Native woodland	Conversion of 3,000 ha of intensively managed remote UK conifer forest to alternative management regimes.
Giraud *et al* (2002)	CV	2000	Annual	Local: -US$255 Alaska: US$40	Expanded federal Steller Sea Lion recovery program

Author(s)	Method	Year	Period	Mean Value (point estimates)	Item valued
				USA: US$100	
Hanley at al (1998)	CE			Woods: £50/household Heather moors: £23/household Wetlands: £21/household	Land in Breadalbane environmentally sensitive area
Hanley at al (2003)	CE	2000	10 years annual	£9/household	Stop shooting geese at Islay, Scotland
Hanley at al (2003)	CE	2000	10 years annual	£12/resident household	Stop shooting endangered geese at Islay, Scotland
Hatton-MacDonald & Morrison (2005).	CE		5 years annual	AU$0.72: scrublands AU$1.02: grassy woodlands AU$1.40: wetlands.	Marginal value of 1000 hectares of habitat type.
Hoehn & Loomis (1993)	CV	1989		US$103/	Wetlands maintenance in San Joaquin Valley, CA
Hoehn & Loomis (1993)	CV	1989		US$137/	Wetlands improvement in San Joaquin Valley, CA
Hoehn & Loomis (1993)	CV	1989		US$99/	Contaminant maintenance in San Joaquin Valley, CA
Hoehn & Loomis (1993)	CV	1989		US$142/	Contaminant reduction in San Joaquin Valley, CA
Hoehn & Loomis (1993)	CV	1989		US$63/	Salmon improvement in San Joaquin Valley, CA
Holmes, Murphy & Bell (2005)	HP	2002	One time	US$7261/house at risk	10% increase in area of moderately defoliated hemlock due to hemlock woolly algelid: Sparta, New Jersey
Jakobsson & Dragun (2001)	CV		Annual	AU$0~68/individual (some responded for household)	Protect Leadbetter's Possum in Victoria, Australia
Jakobsson & Dragun (2001)	CV		Annual	AU$117~267/individual (some responded for household)	Conservation of all endangered species of flora and fauna in Victoria
Jetter & Paine (2004)	CV	post 1994	7 years annual	US$23/individual	Control of eucalyptus snout beetle in Ventura County (CA) using Carbaryl insecticide
Jetter & Paine (2004)	CV	post 1994	7 years annual	US$131/individual	Control of eucalyptus snout beetle in Ventura County (CA) using Btt insecticide
Jetter & Paine (2004)	CV	post 1994	One time	US$485/individual	Control of eucalyptus snout beetle in Ventura County (CA) using egg parasitoid
Kontoleon &	CV	1998		US$3.90	Increase Wolong Reserve (China) Giant Panda population to

Author(s)	Method	Year	Period	Mean Value (point estimates)	Item valued
Swanson (2003)					500 animals in cages
Kontoleon & Swanson (2003)	CV	1998		US$8.43	Increase Wolong Reserve (China) Giant Panda population to 500 animals in pens
Kontoleon & Swanson (2003)	CV	1998		US$14.86	Increase Wolong Reserve (China) Giant Panda population to 500 animals in natural habitat
Kotchen & Reiling (2000)	CV	1997	One time	US$26/	Recovery plan for Peregrine falcon
Kotchen & Reiling (2000)	CV	1997	One time	US$27/	Recovery plan for Shortnose sturgeon
Kramer & Mercer (1997)	CV	1992	One time	US$21~31/household	Protect additional 5% of tropical forests globally
Langford at al (1998)	CV	1995		Use Value: 162 drachmas/person Option Value: 838 drachmas/person Existence Value: 2321 drachmas/person	Protect the Mediterranean Monk Seal in the Aegean Sea
Loomis, Kent *et al* (2000)	CV	1998	Annual	US$252/household	Restoration of 5 ecosystem services on the South Platte River
Loomis & White (1996)	BT	1993	Annual	US$44~95: Northern spotted owl US$31~88: Pacific salmon/steelhead US$46: Grizzly bears US$35: Whooping cranes US$10~15: Red-cockaded woodpecker US$29: Sea otter	Rare and threatened/endangered species
Loomis & White (1996)	BT	1993	One time	US$178~254: Bald eagles US$173: Humpback whale US$120: Monk seal US$16~118: Gray wolf US$13~17: Arctic grayling/Cutthroat trout	Rare and threatened/endangered species
MacMillan *et al* (2001)	CV		Annual	£35~37/household	Restoration of Affric woodland
MacMillan *et al* (2001)	CV		Annual	£25~101/household	Restoration of Affric woodland and reintroduction of beaver

Author(s)	Method	Year	Period	Mean Value (point estimates)	Item valued
MacMillan *et al* (2001)	CV		Annual	£-2~31/household	Restoration of Affric woodland and reintroduction of wolf
MacMillan *et al* (2001)	CV		Annual	£24~53/household	Restoration of Strathspey woodland
MacMillan *et al* (2001)	CV		Annual	£19~100/household	Restoration of Strathspey woodland and reintroduction of beaver
MacMillan *et al* (2001)	CV		Annual	£-13~61/household	Restoration of Strathspey woodland and reintroduction of wolf
McLeod (2004)	BT		Annual	Fox: AU$190million ($26/fox) Cat: AU$144 million ($8/cat) Carp: AU$11.8 million	Environmental costs imposed by invasive animals in Australia (mostly based on Pimental *et al*(2000): 1 lost native bird = AU$1)
Mallawaarachchi *et al*. (2001)	CE	1998	Annual	1000 ha Teatree woodlands: AU$2.56 per household. 100 ha Herbert wetlands: AU$40 per household.	Protecting land from sugar cane farming
Nunes & van den Berg (undated)	TC	2001	Annual	€55/individual	Recreational benefits lost because of beach closure due to algal blooms, Zandvoort, Holland
Nunes & van den Berg (undated)	CV	2001	Annual	€76/individual	Beach closure due to algal blooms, Zandvoort, Holland
Pate & Loomis (1997)	CV	pre 1991	Annual	US$68~215/household	Wetland improvement in San Joaquin Valley, California
Pate & Loomis (1997)	CV	pre 1991	Annual	US$52~233/household	Protect & expand wetlands and reduce wildlife contamination in San Joaquin Valley, California
Reaves *et al* (1999)	CV		Annual	US$8~13/individual	Red cockaded woodpecker habitat restoration
Rolfe *et al*. (2000)	CE				Rainforest preservation
Stanley (2005)	CV	2001	Annual	US$21~28/household	Protection of the Riverside Fairy Shrimp Orange County, California
Stanley (2005)	CV	2001	Annual	US$38~59/household	Protection of all 32 endangered and threatened species in Orange County, California
Tumaneng-Diete *et al* (2005)	CV		unknown	AU$56~74/household	Stop spread of Lantana in Queensland areas of high conservation significance
Tumaneng-Diete *et al* (2005)	CV		unknown	AU$53~73/household	Reduce infested area of Lantana in Queensland (scale of reduction not specified)

Author(s)	Method	Year	Period	Mean Value (point estimates)	Item valued
Tumaneng-Diete *et al* (2005)	CV		unknown	AU$52~71/household	Stop spread of Singapore Daisy in Queensland areas of high conservation significance
Tumaneng-Diete *et al* (2005)	CV		unknown	AU$50~70/household	Reduce infested area of Singapore Daisy in Queensland (scale of reduction not specified)
Turpie (2003)	CV		Annual	US$6.78/household	Existence value of vegetation predicted to be lost in South Africa by 2050
White *et al* (1997)	CV	1996	One time	£12/individual: Otter £7/individual: Water Vole £10/individual: Otter & Water Vole	Maintain species populations and, where possible, restore them to all areas inhabited 25 years ago by 2010

Appendix 3: New Zealand Recreation Valuation Studies

Author(s)	Method	Year	Period	Mean Value (point estimates)	Item valued
Ball *et al* (1997)	BT	1995	Day	$11 /person	Auckland regional parks recreation
Clough & Meister (1991)	TC	1985	Year	Summer: $66 /person Winter: $124 /person	Whakapapa skifield recreation
Harris (1981)	TC	1980	Day	$8 /visitor	Lake Tutira recreation
Kane (1991)	CV	1990	Year	$96-$137 /visitor	Hollyford Valley hiking
Kaval *et al* (2003)	BT	2003	Day	$30 /person	Outdoor recreation
Kerr (1989)	TC	1984	Visit	$160-$200 /climber	Mt Cook National Park mountain climbing
Kerr (1996a)	CV	1994	Visit	$10 /person	Wellington regional parks recreation
Kerr (1996b)	CV	1985/ 86	Visit	Hiking: $28 /person Fishing: $39 /person Hunting: $59 /person	Greenstone & Caples Valleys recreation
Kerr (2004)	BT	2003	Day	Freshwater fishing: $39 /person Other activities: $21 /person	Outdoor recreation
Kerr *et al* (1986)	TC	1984	Visit	$44 /person	Mt Cook National Park recreation
Kerr *et al* (2004)	TC	1983	Trip	$6 /angler	Rakaia River salmon angling
Kerr & Greer (200*)	TC	2000	Trip	$40-103 /angler	Rangitata River angling
Meyer (1994)	CV	1994	Year	$37-80 /household	New recreational lake, Ashburton
McBeth (1997)	TC CV	1997	Visit	$56 /person $67 /person	Tongariro River angling
Nugent & Henderson (1990)	TC	1986/ 88	Day	$14 /hunter	Oxford Forest deer hunting
Riley & Scrimgeour (1991)	TC	1989	Visit	$1.62 /person	Coromandel Forest recreation
Sandrey & Simmons (1984)	TC	1982	Visit	$27 /person	Kaimanawa Forest Park recreation
Walker (1990)	TC	1985	Visit	$6.40 /person	Kaitoke Forest Park recreation
Walker (1992)	TC CV	1989	Visit	$1.62 /person $29 /person	Bottle Lake Forest (Christchurch) recreation
Wheeler & Damania (2001)	CV	1999	One time	Snapper: $6 /fish Kingfish: $20 /fish Blue Cod: $24 /fish Kahawai: $3 /fish Rock Lobster: $48 /fish	Marine recreational fishing
Woodfield & Cowie (1977)	TC	?	Visit	$70-$400 /walker	Walking the Milford Track

Appendix 4: Non-recreation New Zealand Valuation Studies

Author(s)	Method	Year	Period	Mean Value (point estimates)	Item valued
Beanland (1992)	CV	1991	Year	$9 /household	Preservation of Aorangi Awarua forest
Fahy & Kerr (1991)	CV	1990	Year	$22 /person	Albatross chick fatality research
Greer & Sheppard (1990)	CV	1989	One time	$40-$53 /adult	Clematis vitalba control research
Guria & Miller (1991)	CV	1989	One time	$1.9 million	National value of a statistical life
Kerr & Cullen (1995)	CV	1992	Year	$299-$435 /Nelson adult	Paparoa National Park possum control
Kerr & Sharp (2003)	CE	2002	Year	Native Fish: $10 /species /household Fish habitat: $1.38 /km /household	Attributes of a single high quality North Shore stream
Kerr & Sharp (2003)	CE	2002	Year	Native Fish: $5 /species /household Fish habitat: -$3 /km /household	Attributes of a single high quality South Auckland stream
Kirkland (1988)	CV	1987	Year	$6-$13 /household	Preservation of Whangamarino wetland
Lock (1992)	CV	1991	Year	$21-$70 /household	Manawatu possum control
Moore (1998)	CV	1998	Fort-night	Tapeka: $16 /household Russell: $17 /household Horeke: $10 /household	Community sewage scheme upgrades
Mortimer *et al* (1996)	CV	1993	Year	$21-$37 /household	Maintenance of conservation activities on Little Barrier Island
White *et al* (2001)	CV	1999	Year	>$183 /household	Protect Waimea Plains aquifers
Williamson (1997)	CV	?	Year	$10 /household	Orakei Basin water quality