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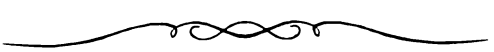
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VALUATION OF ENDANGERED SPECIES: TOWARDS A NEW APPROACH

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ABSTRACT

The loss of biodiversity, as measured by extinctions of species, is presumed to be occurring at a rate that poses a serious danger to the human and natural world. Species conservation is now, a public policy issue. However, a major obstacle to achieving efficient conservation policy, is the difficulty in attributing value to species.

This research develops a methodology, that could be used to allocate resources among recovery programmes for endangered species. An economic approach is used because the resource allocation dilemma is an economic problem. The situation is analogous to an investment decision, where costs are borne now in order to generate uncertain future benefits.

The research identifies where market failure may be present. If property rights to a species can not be defined, market failure may occur because of the public good nature of species. This is likely for species that are ecologically important. If property rights can potentially be defined, but are not actually defined, then market failure may occur. However, in this case, species are not public goods.

Inefficiencies also occur as a result of government failure. It is difficult for public agencies to gather accurate value-information among the candidates for recovery work. Voting and lobbying by environmentalists are an unreliable guide to the relative values of endangered biota. The agency also may face incentives to lower the efficiency of recovery work.

The subjective basis for the neoclassical axiology is explained. The ability of subjective value theory to account for market behaviour, chiefly that voluntary exchange is a win-win game, accounts for its acceptance over objective value theories.

The use of environmental ethics is rejected on the grounds that such value theories are not operational. This point is conceded in some of the environmental ethics literature. The scope of the environment, comprising things both animate and inanimate, both holistic and individualistic and both, sentient and non-sentient presents a major obstacle. In addition, the problem of subjective reduction is difficult to overcome. Finally, there is the teleological problem. It is not possible to know *a priori* which species are more vital for the well-being of ecosystems in the future.

Several approaches to valuing endangered species are considered. The first is the Contingent-Valuation-Method. Such an approach has little use for deriving relative values for biota. The technique depends on being able to describe the good in question accurately to subjects. However, the technique does not identify the problems of time and uncertainty. Thus, species can not be described accurately. It is a partial equilibrium approach. It is also subject to bias and embedding problems.

The second approach is the Safe-Minimum-Standard approach. The technique argues that preservation should always occur so long as the costs are not too high. It does not identify what constitutes too-high costs. There is an explicit bias towards preservation despite the low odds that any particular species will yield future economic benefits.

The third approach is to derive objective criteria to rank species. In New Zealand, this approach is manifested in the Species-Priority-Ranking-System. This approach sums seventeen unweighted ordinal numbers to produce a cardinal sum. This is mathematic nonsense. The approach is vulnerable to subjective reduction, given the choice of criteria has a strong effect on rankings. An implicit bias towards rare species, rather than species of high biological value is noted.

The research instead adopted a hypothetical market approach based on game theory. This was determined to more accurately capture the nature of the problem facing a species manager. In the game, subjects must allocate a fixed budget among twenty-four candidates for recovery work. Each species is described in terms of ecological value, genetic value, taxonomic group and cost. Time to extinction is correlated to cost, the more expensive a candidate is, the less time it is expected to persist. Species are not named but are coded to eliminate mega-fauna effects.

The subject plays against nature which generates a strategy randomly. The strategy may be favourable to recovery work or unfavourable. Players are scored on being able to produce a net improvement in the status of the candidates. Several efficient solutions exist, which permits preferences for some species (eg. birds) to be pursued. Money payoffs were not used in the trial market, only the real market game.

The bidding data was analysed using the TOBIT, LOGIT and PROBIT statistical techniques. Players shifted their strategy towards an optimising one during their real games. In the trial games, over-investment in very costly, risky candidates was observed.

Relative to the Contingent-Valuation-Method, the game theory approach produced data of comparable statistical significance. Its weakness lies in its cost and self-selection bias of subjects.

The research identified costs as being the dominant variable affecting bidding behaviour. The decision among the sample, to indulge in preferences for taxonomic groups such as reptiles, was only observed when the marginal dollar was being allocated. Thus, as the budget was exhausted, only then did players allocated their marginal dollar on the basis

of biological variables. This reflected the desire to save as many species as possible. The deterrent effect of costs was corroborated with a study of real bidding behaviour at the Otorohanga Kiwi House.

Public choice problems in New Zealand were also discussed. The bureaucratic goal of budget maximisation produces a bias towards high profile species, often of low biological value. Finally principal-agent problems and a low ability to monitor field staff, also have the potential to limit the effectiveness of conservation expenditures. This is illustrated with examples from the kakapo recovery programme.

CHAPTER ONE

INTRODUCTION

1.1 Background to Research

The preservation of endangered taxa is now asserted, by many people, to be an imperative activity. Considerable public and private resources have been allocated to preserve threatened species (Solow *et al.*, 1993). New Zealand has been no exception to this activity. The legal status of New Zealand's indigenous taxa has been enhanced, a Department dedicated to species conservation established[1] and an increasing flow of private resources directed towards saving threatened species (Davis *et al.* 1992).

This effort presents several policy problems. One of these relates to the total value of preservation activity. For instance, is the current level of expenditure too little or too much? That is, is enough being devoted to species' conservation, or is species conservation simply taxing other activities that would otherwise be undertaken? Obviously, the resources that are used to preserve species have alternate uses. If the value of these resources exceeds the value of the pool of endangered species, then conservation expenditure becomes value-subtracting. The outcome of the effort is worth less than the combined value of the inputs.

Currently, there is a serious lack of data available for the analysis of this issue. It is not known what the value of any particular species is, let alone the entire set of endangered taxa. To justify or criticise the current level of effort requires a much firmer notion of the value of this activity.

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[1] The 'Conservation Act' was passed in 1987. This enhanced the value of indigenous species with the provision that they had 'intrinsic value' (*Sec 6*). It also created the Department of Conservation (*Sec 5*) which is charged with preserving species and ecosystems native to New Zealand (*Sec 6*).

The absence of explicit values contributes to a further problem of perhaps greater importance. This issue is the *relative value* of each endangered species. The policy problem is for instance, justifying why more effort is devoted to species x than to species y Does this reflect differing values attributed to these species? If not, the allocation mechanism would fail to achieve the most valuable outcome. In order to allocate funds between species that are candidates for recovery work it is necessary to have credible relative values.

Species' conservation, like any other action involves costs. One source of this cost is scarce resources (whether these be conceived as funds, scientific expertise or breeding facilities). As a consequence of this scarcity some species must get less if other species are to have more. That is, allocating relatively more funds to a particular species implies lost opportunities to assist other species. The loss of these opportunities can carry a high cost (for the affected species and human society in general) as extinctions are generally regarded as irreversible (Ehrlich and Ehrlich, 1992). In consequence lower valued outcomes are achieved at the permanent cost of relatively more valuable outcomes (Williams, 1986).

Time also impacts on species preservation. For instance, what will be the effect of postponing a recovery programme? Given that population decline is a near universal condition among endangered taxa, postponement would typically place the species concerned in greater risk of extinction (Williams, 1986). Rates of decline are also different among different species. This implies that postponing a recovery programme for a species in rapid decline could have more serious consequences than for a species experiencing a slower decline. However, preferring to save high risk, rapidly declining species may quickly exhaust the conservation agencies' budget (assuming

risk is related to cost)[2] and hence hastening the decline of many more lower risk taxa. The demand on the general preservation effort would increase greatly in the future (Williams, 1986).

Given that there are several hundred species competing for attention in New Zealand (Molloy and Davis, 1992) recovery programs are under pressure to achieve quick outcomes. Yet, the absence of detailed knowledge about a taxa imposes an additional cost. That is, precious time is spent acquiring knowledge that will make a programme technically feasible. It is an unfortunate fact that the species will still be declining while these studies are going on (Williams, 1986). The time delay produced by a lack of knowledge is going to put a species at greater risk of extinction.

These policy problems revolve the central issue of *value*. How important is any one species relative to another? At what rate should the future be discounted in the design of recovery programs? Further, to what extent do the values of the providers of 'species preservation' (largely the Department of Conservation) coincide with others who also value conservation outcomes? Is an entomologist or ornithologist going to agree that, in New Zealand, the Department of Conservation has given an appropriate weight to their concerns (Hutcheson, 1991)? In light of limited public resources how important is species' preservation relative to the other goals of the government? All of these issues are value problems. Their solution (if possible) will make for better, more consistent outcomes than if they are not solved.

For instance, the current effort being expended on behalf of a species could be considered relative to the measure of that species value or importance.

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[2] This assumption is explained later (Chapter 6). A statistical analysis of the adopt-a-bird programme at the Otorohanga Kiwi House showed a high correlation between costs and risk (Moyle, 1995). There were exceptions: for example the NZ Brown Teal is endangered (high risk) but one of the cheapest species to breed.

The views of those affected or involved may be vindicated or challenged as a result.

The Department of Conservation's recent attempt to rank species (Molloy and Davis, 1992) is in itself a valuation exercise. Species that have a higher ordinal rank have more value than lower ranked species. Attempts such as these need to make their value judgements explicit. Otherwise, achieving a consistent conservation effort will be much more difficult.

Crucially, "valuation exercises" must be robust if they are to stand up to critical examination. In view of the aims of the "priority-setting" exercise, the criteria should be seen to promote these aims. If the criteria does not meet these aims then the attempt is not robust. Likewise, if the particular criteria used do not meet the scientific claims made for them, the valuation attempt is not robust.

The fact that recovery programmes take time to yield results, requires that policy makers consider this effect. Costs will probably be concentrated in the short term with benefits (that is, persistence) occurring in the longer term. The policy maker has to judge the relative importance of these costs and benefits. To do so requires a valuation methodology which takes account of time. Ideally this should be done explicitly rather than implicitly. It is only by having these values explicit that consistent policy be developed. When these values are explicit, they can be subject to critical appraisal. Only then is it clear that lower valued species are not preferred to higher valued species, simply because a highly variable discount rate is being used.

The decision to acquire knowledge about a species implies that a low valuation of current knowledge has been made. This decision is not costless; postponing a recovery programme to improve the knowledge of a taxa may not be the best decision. Launching a programme on the basis of imperfect

knowledge could conceivably be more successful than waiting for adequate information.[3] The value of the additional research effort has to be contrasted with the subsequent increase in risk that the species will become extinct.

Treating the above value judgements in an explicit manner has further benefits. These are that it is likely that the costs of decision making could be lowered. Funds and expertise will be less likely to be diverted or captured by groups with differing views on the appropriate value a species should take. An agreeable set of values or valuation technique will reduce the time cost of preparing and launching a species' recovery programme. Having an appropriate set of values to use will make estimating (or measuring) the returns from species preservation effort possible.

Species preservation is seen as a broad, multi-disciplinary problem. It is an area where biologists, philosophers and economists have an input. For the biologist, the principal concerns are often technical. For both the philosopher and economist the principal concern is value. For the philosopher, the question to answer is 'why is a species valuable?'. For the economist, the question is more likely to be 'how valuable is that species?'. The focus on 'why is a species is valuable' has however, lead on to much writing about species rights and intrinsic value. Several commentators (Norton, 1986; Rolston, 1988; Gunn, 1990; Johnson, 1991) and perhaps a significant part of the public regard species' conservation as a moral issue. There is an ethical dimension to this topic as some people are willing to accept responsibilities and obligations to threatened species. At this level people are more resistant to expedient choice as the only grounds for decision making. The motivation to conserve is not guided entirely by self-interest.

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[3] The California condor provides an example of this. The information available, prior to the captive breeding program, suggested captive breeding would do more harm than good (Cohn, 1993). Rather than spend time refuting this information, the FWS initiated the breeding programme.

It is readily observed that there is a growing interest in the plight of New Zealand's rare biota. Endangered species are newsworthy items. The recent releases of the North Island saddleback on the mainland, the struggles of the kakapo to successfully breed, the fate of the black stilt and the dispersal of the takahe have all been reported (Anon., 1991a, 1991b, 1992a, 1992b, 1992c; James, 1992). Likewise, public support has been forthcoming for the hoiho and Hooker's sea lion. Many people enjoy observing and learning about New Zealand's endangered biota. For instance, over 100,000 people visited a kiwi exhibition at the Auckland Museum over the 1991-92 summer (Barr, 1992).

The participation of private individuals and organisations in species' recovery programs demonstrates the high value placed on endangered taxa by these agents. Individuals and trusts breed the North Island brown kiwi, the brown teal, the blue duck and the weka (Anon., 1991b, 1992b; Harding, 1991). Corporations are also getting involved in conservation. The kakapo, hoiho, kiwi and Hector's dolphin are all benefiting from the interest of corporate New Zealand (Louisson, 1990; Sarney, 1991; Anon., 1992c; Barr, 1992; Davis *et al.*, 1992; Rasch, 1992).

The extent of this private involvement is consistent with very high perceived values for New Zealand's endangered taxa. It could also indicate that a significant sector of society believes that not enough public resources are committed to reverse the decline in status of New Zealand's indigenous taxa. The growing interest in conservation is captured by the changes to membership numbers and funds of the Royal Forest and Bird Society. The Royal Forest and Bird Society is the largest, mainstream conservation group in New Zealand. The changes it has experienced is evident in the table below:

Table 1.1: Royal Forest & Bird Statistics

YEAR	MEMBERSHIP	SUBSCRIPTIONS	DONATIONS
1980	30545	$95935	$8392
1981	32588	$131267	$10642
1982	34975	$182131	$30132
1983	36285	$210754	$16020
1984	40500	$234530	$14322
1985	43800	$345968	$12378
1986	47534	$368182	$39029
1987	51625	$389678	$23903
1988	52704	$556042	$32657
1989	52476	$623619	$31035
1990	58168	$684681	$48418
1991	62047	$825307	$65611
1992	c56000	$773344	$89754
1993	c57000	$688079	$66279

Source: Annual Reports 1980-1993 Royal Forest and Bird Society. Dollar values are in nominal terms.

Of particular interest in this table is the sustained increase in membership from 1980 to 1990. However, since 1990, there have been falls in membership and subscription income. It would be dangerous to assume that

the demand for conservation will continue to rise at historical (eighties) levels.

Overseas evidence also supports a growing trend over the eighties for more public concern to be expressed towards environmental issues. For instance Smith (1990) states that public opinion polls show there is much more support for "continuing environmental improvements regardless of the cost".

In summary the task of species' conservation require solutions to the problems of uncertainty and unknown values. It proceeds with the confidence that the public at least demonstrates an interest and involvement in species' conservation. The policy problem is to determine the level and efficacy of the current efforts and identify ways by which they could be improved.

1.2 The Case for an Economic Approach

This thesis adopts an economic (neoclassical) approach to the problem of valuing endangered taxa. However, the research does not ignore related aspects to this problem. For instance, attention will be given to philosophical issues and biological parameters.

The problems of risk and uncertainty, resource allocation across time and inadequate information that face a species manager are economic problems. Attempting to achieve optimal outcomes, that is the best possible result given resource constraints, is an economic aim. Given the familiarity of the economist with these problems considerable assistance can be offered to policy makers, species managers and other agents.

Many conservationists display an antipathy towards economic analysis (Ell, 1991, 1992) and would suspect that a species would be the least suitable

subject for this approach. How is it possible to put a value on New Zealand's biota? In fact this is done at an implicit level by all decision makers. There exists a scarcity problem with respect to species given there is a large number of vulnerable species, but only a limited quantity of resources and technical expertise. The result is that choices have to be made (Vane-Wright *et al.*, 1991).

In order to make choices, that is, justify a particular course of action, values are used. The decision to fund one species over another gives the preferred species a higher ordinal value. An effort to save high risk species may indicate that the future existence of other species is being discounted. Therefore, any ranking system or any method of resource allocation, involves choice and the attribution of value to a particular species.

The problem is that while these values are implicit the judgements made by decision makers may not take into account the values of other agents affected by these decisions. Further, the risk is very high that the decision maker will apply their criteria inconsistently. The aim of this thesis is to make these kinds of value decisions much more explicit and transparent.

Economic science does not restrict the set of valuable objects to market goods only. The absence of a market, does not indicate something has no value. After all, many valuable goods and services are provided in society that do not rely on markets. These include the police and legal systems and the provision of certain public utilities and elementary education. The existence of a market is neither a sufficient or necessary condition for value to exist. A species may have value to both individuals and groups of individuals. For instance, the white heron is a very important species for Maori.

Individuals manifest their values of threatened species in many ways. They reveal it by visiting aviaries or sanctuaries; they purchase books, posters

and videos on wildlife; they donate time and money to environmental groups and recovery programs. Some individuals assist by breeding and observing endangered species. These choices, leading to these costly courses of action demonstrate that many endangered species do have value.

While individuals can capture some of the benefits from conservation some of these benefits exist, in part, in the public domain. These include the benefits from biodiversity, ecosystem stability and the cultural values of some species.

Biodiversity (Ehrlich and Ehrlich, 1992; Perrings *et al.*, 1992) in a broad sense is the genetic material contained within a natural system's biota. This genetic material gives the biological system (for example, a population or ecosystem) resilience to change. It may contain genes that are (or will be useful) to humans. These include medical and agricultural benefits (Randall, 1992; Perrings *et al.*, 1992). Biodiversity in its extreme form, is seen as vital to maintain the health of the earth's biosphere. Depletion of biodiversity may produce collapses at threshold levels with catastrophic effects (Ehrlich and Ehrlich, 1992). In this extreme case, the benefits of avoiding such a collapse can be said to exist largely in the public domain.

Increased biodiversity, in the sense of many species, improves the productivity of ecosystems (Ehrlich and Ehrlich, 1992; Perrings *et al.*, 1992). A large set of species implies that a large number of ecological niches are being occupied. This is consistent with the Gaussian Theorem which asserts it is impossible for two species to occupy the same niche. As a result biomass production is increased as organisms fill the ecosystem and take advantage of all the opportunities. In consequence the efficiency of the nutrient, mineral and water cycles is improved.

An ecosystem can withstand more stress where biodiversity is high. It will return oxygen to the air, consume green house gases, stabilise slopes on marginal land and act as a recycler and store of nutrients. The ecosystem subsidises human waste disposal, agricultural and pharmaceutical production. Increased biodiversity provides opportunities for evolution and human selection and enhances ecosystem stability (Soule, 1991; Ehrlich and Ehrlich, 1992; Randall, 1992).

As indicated above, ecosystem stability is important to human society and life on this planet. Some species may adopt key roles in an ecosystem. The loss of such a key stone species has widespread effects on other components of the system. This will reduce the ability of that ecosystem to function or in extreme cases, cause a collapse of the system. For example, pollinators are important in forest ecosystems. The loss of a pollinator species would greatly slow the regrowth rate of the forest.

Ecosystems therefore provide some benefits that lie in the public domain. In some cases these will be localised in effect. For instance, a group of farmers downstream of a native forest will benefit from less soil erosion than counterparts not so blessed. Even in the case where one farmer actually owns the forest it would be difficult to get compensation from the other farmers. The point to focus on however, is that these benefits in the public domain are provided by an *ecosystem* and not individual *species*. Arguments to preserve species in terms of the public benefit are on tenuous grounds. The appropriate preservation unit is the ecosystem.

In a cultural sense, some species may have a public benefit through the knowledge of their existence. These are likely to be restricted to species that are rather well known to the public. In the New Zealand case, the best example would be the kiwi. The kiwi is a national symbol of New Zealand. Here, the knowledge that the kiwi survives would provide many people

satisfaction and a significant welfare loss could be experienced if it became extinct. It is this knowledge that remains in the public domain. Nearly all individuals benefit from the preservation of the kiwi even if not all contribute to it.

The discussion above has indicated ways in which individuals and society manifest their values towards species. The distribution of the benefits from preserving species is an important consideration (Randall, 1992; Perrings *et al.*, 1992). Where benefits are in the public domain it is not clear that individuals will have the incentives to bear the cost of provision. Should the individual be prevented from obtaining a reward for bearing this cost then there will be an impediment to the private provision of preservation. In this sense the distribution of costs and benefits in the public and private domain are important. This distinction between the public and private domain provides the basis for an important economic concept. This is the difference between private and public goods (Varian, 1984). This separation is useful and necessary. The method by which value is attributed to a public good is different to a private.

Economics does not deny that species and species preservation are valuable activities. Indeed, the actions of conservationists vindicate this assumption. Economics brings to this topic rigorous ways of examining, defining and measuring the value obtained from species preservation. Further, the analysis of costs are an important part of the economic method. The resources and expertise, as well as the populations of endangered species are scarce. This implies choices have to be made and that these choices will have costs. To determine the best course of action careful comparisons of the value of each option have to be made. An economic approach is a valid and extremely useful way to look at this problem.

1.3 A Guide to the Thesis

This research begins with a single question. Given that a conservation agency has limited funds and expertise, how should they allocate these resources among the many candidates for recovery work? Conversely, what characteristics of a species make it preferable to withhold support?

This is an economic problem given the scarcity of resources for recovery work. There is no way, given the limitations of resources and scientific expertise, that all endangered species can be "saved". Ideally, an approach that selects the most valuable species for recovery work and rejects the least valuable is preferable.

Part of the difficulty in finding an answer to this problem, is the lack of adequate knowledge about endangered species. For instance, only one arachnid has been included in the Department of Conservation's lists, largely because the group is less well studied than other groups such as the birds (Molloy and Davis, 1992). Lack of information generates a high degree of uncertainty about the outcomes of recovery work. These problems are discussed in more detail below.

Much of this research is concerned with what constitutes value. This study adopts a neoclassical approach. The neoclassical methodology considers value to be attributed subjectively. The philosophical roots of this subjectivity are contrasted with alternative objective value theories (Chapters 2 and 3). For instance, both Marx and some Environmental Ethics methodologies argue that value is in fact objective. These approaches are considered and criticised. One of the major difficulties with an objective approach is the scope of what is being valued. A species is made up of individual organisms as well as being holistic. A species may include sentient forms, as with many bird species or non-sentient, such as plants

and fungi. Finding criteria to account for all the set of species is a task that has yet to be solved (Chapter 3).

Several attempts have been made to value species. The relevant, objective approach in New Zealand is the Species Priority Ranking System (Chapter 4). This system is fatally flawed and can not be used to answer our research question. The dominant subjective approach is the Contingent Valuation Method (Chapter 4). However, it is unsuitable to attribute relative values to species. This stems from a number of inherent biases and the difficulty of describing the "good" for subjects. The other alternative is the Safe Minimum Standard (Chapter 4) which has been less popular. Again, there are problems with using this approach. It is not suitable to derive relative values and the welfare basis of its "rule" is rather contentious.

The study therefore, explores a different hypothetical market approach to determining relative values. Like the CVM approach, subjects are placed in an hypothetical market and bidding behaviour is observed. However, by drawing on research done in game theory and experimental economics (Chapter 5), some of the difficulties with CVM are overcome. For instance, the simulation approach provides feedback to players from different bidding strategies.

This simulation approach is developed using Lotka-Volterra ecological modelling. The approach is tested on a sample of students. Their bidding behaviour is analysed using qualitative dependent variable, and limited dependent variable regression techniques. The results are compared to results from CVM studies (Chapter 6). The simulation approach has some shortcomings, notably cost (Chapter 7).

However in contrast to the CVM technique, the simulation generates relative values for species with certain characteristics (Chapters 6 and 7). A

comparison of the coefficients and elasticities, attached to the variables that describe a species, generates the weights that can be applied to those characteristics. A study of actual bidding behaviour at the Otorohanga Kiwi House is used to corroborate the results (Chapter 6).

In short, game theoretic methods compare favourably with other valuation approaches and provides a means by which the research question can be answered. Characteristics of species can be given appropriate, subjective weights. Finally, public sector obstacles to efficient conservation policy are discussed.

CHAPTER TWO
PARAMETERS TO THE RESEARCH

This chapter discusses two important parameters of the research. Firstly, the nature and role of social institutions, and secondly, the nature and impacts of uncertainty.

2.1 Social Institutions

Species' recovery programs involve human institutions. Of all the species on the planet only humans appear to be concerned about extinctions. In society, there are a number of institutions by which resources are allocated. Achieving a greater level of species preservation requires the commitment of resources by society. Further, this is an activity where human values are relevant. There are a number of ways these values are revealed. Political and market institutions can be used.

The Role of the Public Sector:

One approach to the allocation of resources to rare and endangered species is to use the political process. Individuals express their values through a combination of voting and lobbying behaviour. For instance, the Royal Forest & Bird Society of New Zealand has lobbied the government to increase the funding of the Department of Conservation (Moyle, 1994).

Another example of this lobbying effort involved the Hooker's sea lion. This animal is inadvertently drowned in trawler nets around the Auckland Islands. The suggested solutions from the Royal Forest & Bird Society included a 100km radius sanctuary around the Auckland Islands and a change from trawl nets to jiggers to catch squid (Anon., 1992d, 1992e). These proposed solutions would provide significant protection to the species

at the cost of squid-fisher incomes. Notably, this was not the solution selected by MAF and the government.

The political process is not an institution that provides accurate information on values. The salient feature of the lobbying process is that the lobbyist does not bear the major costs of the programme. In the case of the Hooker's sea lion, it is the squid fishers who bear the costs of the programme. In the case of increased funding towards the Department of Conservation, the programme costs are shared amongst all tax payers. However the benefits of these programs are captured largely by the lobbyist and their interest groups.

The outcome of this is that lobbyists have little incentive to prioritise their campaigns. The ability to shift the costs on to other parties means that the lobbyist can avoid the "scarcity problem". In each case, the resource-cost of the programme does not fall on the lobbyist. All programs are thus likely to pass their benefit-cost calculus. There is little need for the lobbyist to rank preferences.

Lobbying behaviour provides very little information to a decision-maker on the values these people attribute to certain species in absolute or relative terms. For instance, how valuable is the Hooker's sea lion relative to the kiwi? If both are attracting lobbying effort, the only relevant information the decision maker has is that they can be presumed to have some positive value. This information would not be adequate to allocate the marginal dollar of conservation funds between the two.

Reliance on the political process is likely to result in *government failure*. Weak incentives and poor information imply that observed behaviour will give very inaccurate data on values. The lobbyist or voter has weak incentives to consider the costs of a programme. The decision maker is

ignorant of the size and distribution of benefits the public receives from their efforts. The result, is insufficient information.

In this situation the decision maker either has to impute some set of values to the public or rely heavily on their own. Historically in New Zealand, high risk birds have been the preferred choice (Williams, 1986). By 1992 almost half of the recovery programs adopted by the Department of Conservation were for birds (Department of Conservation, 1992). On the other hand, birds made up around 25 percent of the total numbers of endangered species (Molloy and Davis, 1992).

Another dimension to the government failure problem is that political solutions tend to be "rented" rather than "brought". A political solution can always be reversed by a change of policy or Government. There is no guarantee that today's political solution will survive indefinitely, or indeed, will be rigorously enforced. The lobbyist therefore has to expend resources to obtain the policy they desire, monitor its enforcement and try to perpetuate it.

The Role of Markets:

Competitive market institutions provides information lacking with public sector institutions. Further, a competitive market contributes towards the efficient allocation of resources. The price mechanism informs the provider and other agents about the value people put on a good. The more a subject is willing to pay, the more value that can be attributed to that good. Thus prices, as a manifestation of willingness-to-pay, convey value data on a set of goods.

The provider of a good must also set their prices to reflect the resource cost of producing that good. This ensures that the most valuable use of resources

are made. If the prices charged could not cover these resource costs, the production being undertaken would be *lowering* the value of the inputs. Producer price setting acts to make consumers aware of the resource cost of obtaining the things they value. This encourages priority setting behaviour with the consumer.

A competitive market directs resources to their most valued use. If any market participant values a resource more than its current user, they can increase their bids to obtain it (Heyne, 1993). This results in an important welfare outcome of a market. Given that all voluntary trades are encouraged the welfare of market participants is maximised. That is, a point will be reached where nobody can be made better off without someone being made worse off. In short, after all voluntary trades have been made, no imposed reallocation can be a Pareto improvement (Varian, 1984). In the absence of voluntary exchange a Pareto outcome can not be guaranteed. That is, someone who has a good to which they attribute a relatively low value, can not exchange it for a relatively highly valued good.

Markets therefore are an institution by which values can be discovered and revealed (Walker, 1993). Were kiwis and sea lions market goods, then a provider would have little difficulty in allocating their marginal dollar. The bids offered by consumers, would communicate their preferences to the producers.

Market Failure:

Underwriting any market process is a system of well defined property rights (Alchian, 1989, 1993). That is, the good in question must be capable of being owned, the owner of the good has the rights to appropriate benefits of ownership and these rights must be transferable, enforceable and enforced. Without private property, there can be no market (Alchian, 1989).

One conspicuous feature of endangered species is that there is not a market for their preservation. The absence of a market is in turn, related to the absence of well defined property rights. There are two reasons why the property rights associated with a species may be poorly defined. The first is if the good concerned is a public good. Public goods have two features which prevent them from being "capable" of being owned and any rights are "unenforceable" (Cowen, 1993). These features are that of non-depletability and non-excludability. The consumption of a public good does not reduce the quantity of it available to other consumers and it is impossible to prevent others from benefiting from its provision.

Where exclusion of consumers is impossible, any property rights specified are unenforceable and the owner does not have, in reality, any appropriability rights. If they act to increase the quantity of the public good, then the only things they appropriate are the costs of doing so. The benefits lie entirely within the public domain. In this case, the supplier is not rewarded for their willingness to bear production costs (Perry, 1968).

The second reason why private property rights are not well defined is that they have not been specified. That is, despite being capable of private ownership, legally the good concerned remains in the public domain. Here, the appropriability of the good is diminished. As in the case of a public good, the supplier does not have the incentive to bear production costs. The difference between the two cases is that here, specifying property rights will give suppliers such an incentive. In the case of a pure public good, no such specification will result in producers being able to appropriate benefits.

The question is therefore, is "species preservation" an activity that is a public good or one in which the benefits could be potentially captured by private agents? Traditionally three categories of benefits are supposed to flow from species conservation. These are the "ethical and aesthetic", the

"direct benefit" and the "indirect benefit" categories. Each are discussed in turn.

The first is the "ethical and aesthetic" (Ehrlich and Wilson, 1991). Species are aesthetic (illegal trade in species, after all demonstrates this) and it is asserted, by some, that humans collectively have a moral responsibility to protect species (Ehrlich and Wilson, 1991; Ehrlich and Ehrlich, 1992).

Aesthetic gains exist largely as private goods. For instance, attractive orchids or tropical fish are capable of being traded as private goods. To facilitate this, there may need to be increased specification of rights to lower the incidence of poaching or "exploitation of common property" resources. However, it is not obvious that "aesthetic appeal" exists as a public good.

Further, the set of aesthetic things is very large. It encompasses human made art and architecture, new plant varieties and many other assets. It is unlikely that there will be a shortage of aesthetic things if the numbers of species on this planet decreased. Finally, most endangered species are tropical plants and invertebrates. There is little unanimity that these species are as aesthetically pleasing as bird or mammalian species (Hutcheson, 1991). For instance, no beetle species is listed in CITES despite an active trade in these organisms[1].

The issue of whether there is an ethical element to species conservation is best left to a later chapter. Suffice to say at this point, traditionally, ethical theory is reluctant to allow a group of people to impose their view on ethics

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[1] "There are **no** beetle species on CITES despite an apparently massive trade in them. At the recent Conference a proposal by the Netherlands to list *Colophon* spp. on Appendix I was scuttled when South Africa withdrew its support..."
Ronald I. Orenstein
International Wildlife Coalition
29 November 1994

on the rest of society. Moral actions are attributed to individuals freely choosing to behave in a certain manner. To have this choice taken away does not make the resulting, coerced actions ethical.

The second relevant issue is the extent that moral concern for the environment dominates other moral issues. An analysis of charitable contributions by U.S. households showed that only 1.5 percent of the moneys donated went to environmental causes (Soule, 1991). It is difficult, therefore, to attribute a very high ethical regard to the environment in the face of this behaviour.

However, saving species may generate moral "externalities". That is, some people are able to obtain free "moral" benefits by depending on other people contributing to recovery programs. In this case, the *knowledge* that a species is being saved would qualify as a public good. A private provider could not appropriate the welfare gain to the free rider.

However, it should be noted that the majority of endangered species are not known to the public. World wide, most endangered species are presumed to occur in tropical rain forests (Mann, 1991). Few of these have been described or are otherwise known. In New Zealand, few people are able to name more than a handful of endangered species. Those that are identified are largely drawn from the set of birds. Conversely, the majority of New Zealand's endangered species are plants and invertebrates (Molloy and Davis, 1992). The knowledge argument has limited justification as a public good.

The second class of benefits produced from species conservation is the direct benefits provided. These are the foods, medicines and industrial products that originate from species (Ehrlich and Wilson, 1991). In other words,

species are treated as complex assemblages of molecules that might have economic benefits in the future.

Many of these benefits are capable of private ownership. Take the case of a new pharmaceutical product developed by a company. It can protect its investment by patenting the process by which the new product is obtained. They could also modify the molecule and patent that. They could keep the specific source of the new product as a secret. All of these processes provide a means by which competitors can be excluded from duplicating their success.

Further, there are substitute means by which new and interesting molecules can be generated. Bioengineering and biotechnology provide an alternate means to come up with new products (Huber, 1992). Saving endangered species is unlikely to be a cost effective way of discovering new products (Moyle, 1994). It is possible that the value of species for deriving new products falls over time as human knowledge, with respect to bioengineering improves.

Very little effort is expended on screening species for "pharmaceutically useful" molecules. In part this is because the odds of success are very low (Beattie, 1992). By 1992 Xenova had screened 23,000 tropical plants and fungi for new products without success (Moyle, 1994). The odds of any particular species containing useful molecules is rated at between 5,000 and 10,000 to 1 against (Moyle, 1994). At these odds, threatened species are likely to be too costly, given their expected payback.

The third class of benefits are the array of services provided by ecosystems. Here species are regarded as the "working parts" of the ecosystem (Ehrlich and Wilson, 1991). Ecosystems have the capacity to influence the atmosphere as well as the production of soil and its maintenance. The

consumption of insect pests and the like is also a service to human economic systems.

These benefits are much more likely to be public goods than the first two classes of benefits. Stabilising climatic change is likely to be a service that is strictly non-excludable. No single human agent is able to be excluded from this benefit. Soil fertility and pest destruction are less likely to be public goods as there are technological substitutes for this.

This provides an argument for limiting the extinctions of species that have high ecological importance. For instance, if we accept the climate change hypothesis and the threatened ecosystem has important atmospheric effects, a public good argument has some merit.

However, for many of New Zealand's endangered species this is not a very supportive argument. Many rare species exist as relic populations with little impact or effect on their surrounding ecosystem. While many New Zealand plants are adapted for pollination by birds, most of this is undertaken by a group of birds that are not threatened (Clout and Hay, 1988). There is no evidence that any particular bird species and flowering plant are mutually dependent (Clout and Hay, 1988). Many such species have been transferred to off-shore islands from the mainland. In short, they have not demonstrated themselves to be indispensable elements of mainland ecosystems.

Species that have significant impacts on their ecosystems are generally termed keystone species. Unfortunately, there is little evidence that keystone species exist (Mills *et al.*, 1993). After-all, it makes little evolutionary sense for other species to link their persistence with an unrelated species. Further, while it is conceded that there are advantages to high levels of species diversity, there is little data on what constitutes the

ideal level of species richness (Baskin, 1994). For instance, it has been argued that many species are in fact superfluous to the proper functioning of ecosystems (Baskin, 1994).

The lack of private sector interest in species preservation is unlikely to be a product of their public good status. Rather, an argument could be made that the process of giving species state protection and banning trade has contributed to the loss of species.

There does exist an illegal market for New Zealand flora and fauna (Anon., 1990a, 1990b). The existence of trade bans reduces, in the legal market, the appropriability of investments in breeding. If it is not legal to recoup production costs through revenue from sales, the incentive to breed such specimens is eliminated.

Property rights produce powerful incentives to preserve species. An important example is elephants in Southern Africa. In African countries such as Zimbabwe and Botswana, local communities have property rights in elephants (Simmons and Kreuter, 1990). They are able to appropriate the benefits of having elephants about. If a hunter wishes to shoot an elephant in Zimbabwe they must buy a permit (around $US 25,000) from the local inhabitants (Larson, 1993). This fee goes directly back to the villagers. As a result, elephants are not seen as destructive pests but as an economic resource. In such communities, there is a willingness to bear the costs of having extant elephant populations. Poaching in Zimbabwe fell by 90 percent when ownership of elephants was transferred back to local communities (Larson, 1993).

In contrast, where the state exercises full control over the elephant populations, elephant numbers are in decline. In Kenya, elephants are managed with armed guards and electric fences (Larson, 1993). For local

communities in Kenya elephants have only a pest status. Villagers receive none of the benefits of the presence of elephants. Kenyan farmers compete with elephants for agricultural land (Moran, 1992). The view that elephants are a pest rather than a valuable resource leads inevitably to one outcome. Large numbers of elephants are poached every year in Kenya (Simmons and Kreuter, 1990). As a result, since 1981 Kenya has lost 75 percent of its elephant population (Larson, 1993). In Zimbabwe and Botswana elephant numbers have been on the rise (Larson, 1993). Indeed, Zimbabwe is in the position where it is forced to cull 5-7,000 elephants annually.

The impact of property rights, as this example illustrates, is an important aspect of species management (Moran, 1992). If private agents can appropriate the benefits attributed to a species, then they have an interest in the long term survival of that species (Moran, 1992). The absence of a market for species limits the private sector willingness to commit resources to recovery programs.

Further, the absence of a legal market does not *eliminate* the demand for native specimens. New Zealand plants and animals can be obtained through the black market in the USA. A tuatara may fetch $US 100,000 in the black market (Anon. 1990c). However, poachers and this illegal demand lower the ability of a native species to persist. For one thing, smuggling results in a very high attrition loss of specimens. Many birds and reptiles do not survive the smuggling process (Wildlife Conservation International, 1992). Thus, a smuggler has to remove more specimens from the native population than are delivered to customers.

A second aspect of the illegal trade is that such specimens are usually removed, from the breeding population. The collector has no incentive to contribute to a breeding programme. If they disclose they have such a

specimen then they are likely to face legal sanctions. Any specimen collected is thus effectively lost from the gene pool.

A trade ban may in fact contribute towards the loss of rare species. The economic gains to be had from legal trade are eliminated. The trade ban effectively increases the demand for illegally supplied specimens and eliminates the gains from private breeding. It is relevant to note that the property rights associated with New Zealand's endangered plants and animals, lie almost entirely in the *public domain*.

However, the association of private property rights with species has succeeded in New Zealand and elsewhere at attracting more private sector support. A number of corporations have been "willing to pay" for the green image that comes with the sponsorship of an endangered species. Comalco has sponsored the kakapo and the BNZ has sponsored the kiwi recovery programs. These have attracted almost $NZ 2million in support (Davis *et al.*, 1992).

In Costa-Rica, Merck & Co have been willing to pay for the screening of native organisms for pharmaceutically useful molecules. Part of the price they are willing to pay is royalty payments to maintain wild life reserves in Costa Rica (Moyle, 1994). The use of private property rights to enhance the survival prospects of rare species has the potential to be exploited further. Thus, both the public sector and the private sector have a role to play in the preservation of species (Bennett, 1994). The alternative assumption that this issue is just market failure would lead to severe sub-optimal policies.

2.2 The Problem of Uncertain Outcomes

Species conservation policy rarely deals with a world where outcomes are known. The risk associated with a particular policy can not easily be measured or observed. For this reason, species' managers have to make decisions in an environment of uncertainty (Maguire, 1986). Risk is a factor that can be accounted for. It tends to involve regular stochastic events of known effects. Uncertainty is much harder to incorporate. It involves infrequent events where the effects are much more varied.

There are many sources of uncertainty facing the species manager (Maguire, 1986). The first is the impact of exogenous events on the species being managed (Williams, 1986; Maguire, 1986). One dog, by itself, was able to cause considerable losses to the kiwi population at Waitangi (Taborsky, 1988). The kakapo would be under considerable threat if imported parrot diseases infect the already low population. Kiore have predated on kakapo chicks and eggs (Anon., 1992d, 1992e). The very cold winter of 1992 disrupted the breeding of the black stilt. Given the small populations coincident with endangered species, these exogenous shocks can significantly reduce the probability of the species avoiding extinction.

There are also endogenous factors which are hard to measure. The success (ability to persist into the future) of any species is dependent on the genetic fitness of the offspring. An inability to produce genetically-fit offspring implies the effective extinction of the species. This state is termed **quasi-extinction** (Montgomery and Brown, 1992). It is, given current technology, difficult and expensive to measure the genetic fitness of a species.

In addition, endangered taxa are typically associated with small and single populations. Approximately half of the New Zealand species considered to be in urgent need of recovery programs have one population only (Molloy

and Davis, 1992). This condition is normally associated with a narrowing of the genetic base. Several populations of a species, through subtle differences in habitat, will have a wider genetic base than single species. Small populations are associated with an increase in homogenous genes, lethal alleles and inbreeding depression. The resilience of the species to shocks is expected to be low (Maguire, 1986).

This reduction in genetic fitness implies that it is difficult to determine whether a species is capable of producing viable offspring. A species may be able to breed successfully, but if the new generation is unable to do so then quasi-extinction has occurred. The ability of the current generation to produce offspring, is not therefore a reliable indicator that the species is 'genetically fit'. For this reason, the view that species such as the kakapo and black robin are 'lost causes' is not without its supporters (Williams, 1986). Genetics does provide a rough rule of thumb that suggests at least 50 breeding individuals are needed in a managed population for it to be genetically viable (Soule, 1991). Many of the critically endangered species in New Zealand have populations lower than this.

Conversely, a small population and narrow genetic base may still successfully breed and flourish. The ability of small numbers of individuals to colonise new habitat is also well known. The Galapagos finches speciated from a narrow base of colonisers. In New Zealand, many pests also had their origin with small populations. These include the German wasp (Miller, 1971) and the wild pig (Captain Cooker). However, the requirement here is that the population be able to expand rapidly to break out of any genetic bottlenecks. Further, the success of some colonisers does not give us any indication of the proportion of would be colonisers that fail. Failures after all, go largely unrecorded.

These genetic factors produce a great deal of uncertainty over the outcome. It would appear for many endangered species that a recovery programme is playing rather long odds. However, the success of colonisers implies that success may be an achievable goal.

Uncertainty may also be a product of a lack of knowledge. For instance, it may not be known which factors are contributing to a species decline. For instance, it may be a function of habitat destruction, predation, human action or climate change. Alternately, there may be nothing that can be done if the decline is an outcome of natural selection and the evolutionary process. After all, it is estimated that over 90 percent of the species that have in the past existed on this planet, have since become extinct (Simon and Kahn, 1984; Norton, 1986; Mann, 1991; Soule, 1991; Perrings *et al.* 1992). In New Zealand the kakapo was already in decline prior to European settlement (Williams, 1986). Clearly, a programme that does not address the causes of decline may be expensive and ultimately futile.

Knowledge about a species may be very incomplete. For instance, a population of the little spotted kiwi was thought to occur in Westland. This belief proved to be false but as a consequence, delayed any recovery programs that could be launched. The weka has also suffered an unexpected and precipitous decline in numbers (Williams, 1986; Anon., 1991b). Again, the prior belief that the species was not facing a decline delayed the launch of a recovery programme (Williams, 1986).

At the other end are species that were thought to be extinct. Extinct species do not, for obvious reasons, qualify for recovery programs. At least two species of birds, the takahe and taiko, presumed extinct earlier this century have had populations rediscovered. These delays clearly also impact on a species. The lack of knowledge of some classes of biota, for instance the invertebrates, contrasts strongly with what is known about the birds or

amphibians (Davis *et al.*, 1992). This lack of knowledge implies that achieving a balanced conservation strategy will be uncertain and that costs of recovery for poorly known groups will increase over time.

This lack of knowledge implies an additional cost represented by the time forgone as data is collected and lag times are taken into account. During this period it can only be expected that the species concerned will be continuing their decline. This increases the chances of extinction for the species and increases the material resources needed to preserve the same.

The species manager faces two related problems: these are how to optimise effort, and how to take into account uncertainty. In order to optimise, the economist uses the method of marginal analysis. For instance, the underlying solution to the lack of knowledge would be to accumulate data up to the point where the marginal benefit of the research is equal to the marginal cost.

Decision making under uncertainty is also a familiar problem to the economist. The aim is to achieve the best outcome possible in **expectational** terms. Outcomes are expressed in terms of the actual outcome adjusted by an estimate of the probability of success or failure. For instance, where two outcomes are equivalent then it would pay to prefer the least risky option. Efforts to save a species (with near certainty of success) gives a better outcome than efforts to save a species with little hope of rescue, other things being equal.

Concentration on expected outcomes finds its expression in the expected utility and game theory approaches first developed by von Neumann and Morgenstern (1944). Game theory is now used in a wide variety of settings. It is used to examine political and economic situations (Fudenberg and Tirole, 1991). It is also used in evolutionary biology (Maynard Smith, 1982).

It accounts for behavioural aspects in populations and species and indicates the advantages of retaining a wide gene pool. These outcomes, in biology, politics and economics are occurring in a background of uncertainty. It is not known how political opponents will act, nor how business competitors will respond. It is not known how the biological future state of the world will be and hence the need to retain useful genes. The analysis of these issues demonstrates that Game Theory has an extraordinary breadth of uses. This research is predicated on the view that Game Theory will play an important role (Chapters 5, 6) in both estimating value and guiding decision making.

CHAPTER THREE
PHILOSOPHICAL ISSUES

3.1 What is Value?

The aim of this project is to discover an approach that reveals *values*. The "objects" that are being valued are a variety of endangered and threatened taxa. Thus, at the completion of the research a number of value claims will be made. These could be that species X is worth Y or that property A which some species share has value B. These claims are similar to many value claims people make about other objects.

People are familiar with a wide range of value claims. For instance, 'Jones prefers spicy food' or 'Bach was a better composer than Vivaldi' are value claims. Other claims have less concrete objects. These include 'it is wrong to steal' or 'better dead than red'. Value judgements therefore, permeate almost every aspect of our lives. Shopping, voting and a wide range of other social interactions involve value judgements.

To understand what values are, it is not sufficient to just be familiar with the language of values. What is meant, for instance, by the claim that 'Bach was a better composer than Vivaldi'? The claim has no ethical content (although many value claims do have an ethical content). It is not an empirical claim either. That is, it is not a factual assertion. For instance, it would be impossible to design a value-free test that could determine whether or not Bach was Vivaldi's superior. For example, it is not a value-free test to use the total output of concertos. That would be predicated on the 'value' that a high output was better than a low output.

The value claim concerning Bach and Vivaldi does not assume that everyone will have the same value belief. It does not rule out the possibility that other

people will not regard Bach as Vivaldi's superior. The claim may be intended to be universal in the sense that all people should share this belief. The intention of universality however is not a sufficient condition for the value to be universally held.

A value claim, like the one above, does have positive attributes. At one level, it means that a subject thinks that Bach is better than Vivaldi. This in turn, underwrites certain expectations of behaviour. We would expect the subject to buy Bach's music in preference to Vivaldi. We would expect the subject to listen to Bach's music in preference to Vivaldi. The subject's monetary and time budgets should be allocated to reflect this belief. Otherwise we would regard the subject as being deceitful, pretentious or hypocritical.

Value claims are also useful for the subject. This is because they justify a course of action. For instance, the subject buys a Bach concerto in preference to a Vivaldi one. This is justified because, in the subject's view, Bach was better at composition than Vivaldi. In another example, the subject who believes it is better to be 'dead than red' is likely to vote for a party that is anti-communist. This vote can be justified in terms of this belief.

In summary, someone who claims to hold a certain value should act in a manner consistent with it. This provides the basis for attributing value to a subject. It is possible to observe a subject's actions to determine what their values are. On the other hand, it is rarely possible to know someone's inner mind well enough to discover their values. Generally actions, by which we mean budget allocations, provide the least contentious way to attribute values (Rescher, 1969; Gorman, 1971). In this case the subject has to compare and rank their values of a set of objects to make their decision. Decision making under a budget constraint imposes a cost on the subject. They have to give up something in order to have their most valued object.

This condition (opportunity cost) is rarely met by oral discourse or voting. For this reason, budget allocations are treated as superior to other manifestations of values (Rescher, 1969). Evidence has also been supportive of budget allocations as the most reliable (but **not** exclusive) indicator of value (Smith, 1990).

The outline on 'value' above illustrates how value claims make sense. It is clear that both a subject and an object needs to exist if a value claim is to have meaning (Frondizi, 1971). Fantastical objects are therefore not valuable. An imaginary desert island would attract no part of a subject's budget. There is however, no requirement that the object be concrete. Subjects will have many abstract values that guide their behaviour in the real world. The example of the Dodo or the Passenger Pigeon could therefore said to be valuable to a subject. The extinctions of these species makes them valuable to some subjects. This could be because these cases illustrate a state of the world where conservation principles do not apply. The dodo and passenger pigeon justify why a subject might undertake actions to benefit species.

For value claims to have meaning it is also necessary to have a subject (Frondizi, 1971; Norton, 1986). The subject is the thing that subscribes to a particular value (Rescher, 1969). This will underwrite the requirement for the subject to display some consciousness or self awareness. This is not considered axiomatic by some conservation axiologies and these are discussed below. The assumption of self awareness or consciousness will restrict the subject to being human. It is worth noting that some conservation ethics insist that animals can be legitimate subjects too.

Historically much debate in philosophy and in economics has had to do with the location of value. The options were seen as value being located on the

object (hence objective theories of value) or value being located on the subject (hence subjective theories of value). Essentially, both proceed from the idea that either the object or the subject is indispensable in making value claims. As with all simplifications, this trivialises the arguments made in favour of all views. A reasonably balanced presentation of these views can be found in Frondizi (1971).

The following is an account of the development of axiological theories in economics. It is an account of economic history and philosophy. It provides important background to the development of the current neoclassical axiology.

Historical and current economic thought has subscribed to both subjective and objective value theories. Sometimes a value theory has incorporated both subjective and objective elements. The appropriate starting point is, as it is so often in economics, Adam Smith. Adam Smith was a proponent of the idea that value was outwardly subjective, but had an underlying, objective basis. Smith (1961) asserts this objective quality:

> " *The real value of all the different component parts of price, it must be observed, is measured by the quantity of labour which they can, each of them, purchase or command. Labour measures the value not only of that part of price which resolves itself into labour, but of that which resolves itself into rent, and of that which resolves itself into profit*"

Smith regarded value as being an objective property embodied in the object. The quantity of labour, measured in terms of the time it took to produce a good, was the value of the good. This involved taking the cost of producing the good and reducing it to a labour time equivalent. If a good was exchanged at this cost of production, it was said to be exchanged at the **natural price.**

Smith also recognised that objects might not be exchanged at this natural price. Thus the second element of Smith's axiology was the concept of the market price. Conditions might exist where consumers were willing to pay more or less than the natural price. This was seen as a temporary phenomena. The entry and exit of suppliers would adjust the costs of production of the object. Hence the natural price and market price were seen to gravitate towards each other.

The subjective element to Smith's value theory was this market price. Demand was able to vary from the natural price, but the explanation for this was not forthcoming. Given Smith could not identify what produced this variation it had to be a subjective reason. That is, for reasons best known to the subject, they are willing to pay more or less than the natural price.

The outward manifestation of market price therefore, provided the subjective basis to Smith's value theory. The underlying costs of production, reduced to labour time, provided the objective reality. This value theory provided the basis for much of the debate in the nineteenth century. Smith, in fact raised many of the value questions that would face later economists. His answers were not as satisfactory.

The concepts and issues that Smith identified were: what was the relationship between market price and value and was value objective or subjective? The scope of inquiry was limited markets. This narrow range of inquiry limited the ability of early economists to talk of value outside a market context. However, markets should be easier to account for. After all, prices are a manifestation of people's valuation of a commodity. It is also a valuation that lends itself well to quantitative analysis. A study of a human market can therefore answer two axiological questions. The first is, does the object have value?. The second is, how valuable is the object?

The idea that value could be objective had many proponents in the nineteenth century. Perhaps the most famous of these was Karl Marx[1]. Marx adopted a Cartesian (scientific) method of inquiry. To identify the objective basis to value one had to examine all valuable (market) objects. If there was a single property of these objects that was universal, then this property would provide the objective basis of value.

Clearly Marx was looking for an objective basis for value. This eliminated from his discussion a universal, but subjective aspect of market objects. That is, that the property common to all market objects is that they are desirable to subjects (Conway, 1987). Marx did not consider this explanation for value in his seminal work, Das Capital (1933). Logic, *under the limits Marx set to his inquiry*, lead to labour time being the basis of value[2].

These objective theories have not been free from serious criticism. The most pragmatic problem is that it has been impossible to measure objects in terms of embodied labour (Prychitko, 1993). Workers after all, often have differing skill levels and different productivity. Smith's solution was to propose a skill-investment by more productive workers. For instance, a skilled worker invests time in acquiring productive skills. When this extra time is taken into account then there is little or no difference in the time it takes skilled or unskilled labour to produce an object. This fails to explain why any one would try to acquire skills if this was the case and does not account for differences in productivity among *similar* skill levels (Conway, 1987).

Marx managed to create a solution that, in the end was not a solution. Marx (1933) proffers as axiomatic an abstract, **homogenous** labour unit to which

[1] Ricardo was also prominent as an "objectivist" and influenced Marx (Henderson, 1993).
[2] "...they [commodities] have only one common property left, that of being products of labour". Marx (1933)

all labour can be reduced[3]. This approach is somewhat Newtonian in that it mirrors Newton's axioms of absolute space and time. To measure value it is necessary to reduce 'embodied labour' into this unit of account. This has never been achieved and it remains an article of faith that it is even possible (Deane, 1978) .

These objective theories could not explain the relationship between market price and value. Marx (1933) felt there was no relationship between the two. This was hardly a detailed or satisfactory explanation. Perhaps more damaging to the objective case was it could not explain voluntary transactions. Exchange is a fundamental property of market economies. Yet, if two subjects, each possess an object of differing "objective" value, then no voluntary exchange could take place. One subject, for instance, would have an object of greater value. There would be no incentive for them to exchange it for an object of lesser value.

The Marxist has two solutions that are *ad hoc*. The first is to emphasise the coercive nature of many exchanges. Thus, capitalists exploit workers and imperialist states exploit colonies. The second is to emphasise that specialisation leads to greater material output and that exchange should lead to specialisation. The problem is that this solution suffers from a tension. Specialisation also produces alienation of the worker from their output, which is not desirable. Second, these benefits flow after exchange takes place. The problem is how to initiate exchange under such a value system; not that there are benefits to be had at some remote future point.

By ignoring the desirability of an object in the eyes of a subject, objective theories could not explain the 'perceived' reality. They could not measure value despite its numerical (price) outward manifestation. The concept of

[3] "The labour, however, that forms the substance of value, is homogenous labour, expenditure of one uniform labour power." Marx (1933).

value was also constrained to human produced goods. This limited the applicability of this value system. Environmental goods, social goals and broader values (honesty, virtue etc.) could not be explained in this value system.

Crucially, objective theories failed the empirical test; that is, why voluntary exchange would take place if values were objective. For any objective value system places the same value **for all subjects** on an object. The subject appraises this property when recognising value. Therefore, no subject which possessed a higher valued (sic) object would voluntarily exchange it for a lower valued one. Exchange can not make both parties better off under such an axiology. Finally, the view that price was not related to value could not win adherents in mainstream economic thought.

The value questions economists in the nineteenth century addressed could not be answered with the objective approach. The alternative was the subjective approach which culminated with, and was part of, the 'Marginal Revolution' (Henderson, 1993). The explanation for exchange followed logically from this point. Any two subjects would be willing to put a different value on the same object. If both possess an object that is more highly valued by their opposite, then exchange would make them better off. Value is in the mind of the subject, and as such it is contingent on time and circumstance. For example, icecream has the same 'objective' value and properties in summer and winter but is much more valued in summer.

The marginalists also had an explanation for why goods were valued. This was a result of their utility. This was used in a variety of senses. For instance, Jevons regarded utility in a very hedonistic way (Harsanyi, 1988; Henderson, 1993). It is more common, in utilitarian thought, to regard the utility of an object as fulfilling needs and desires of the subject. To resolve the 'Diamond-Water' paradox they explained price in terms of its marginal

utility. The 'Diamond-Water' paradox referred to the fact that diamonds command a higher price than water. Yet, it was intuitively obvious that water had more total utility than diamonds. After all, in the absence of water people die but the absence of diamonds is not noted for its fatal consequences.

The marginalists argued, within a partial equilibrium (or single market) analysis that the price that was paid was equal to the marginal utility. Given declining marginal utility with increased consumption, price should also decline. In a competitive market it is only possible to charge the equilibrium price. This price is equal to the marginal utility of the last unit of consumption. For instance, water was consumed in great quantities so that the last unit had very little marginal utility. Diamonds were consumed in very low quantities hence the marginal utility was much higher.

The marginalists thus had satisfactory solutions to the axiological problems identified above. They could relate price to value, which the objective theories could not. Further, they gave a rationale for that most basic human activity; voluntary exchange. As a measure of this success these axiological ideas have been incorporated into mainstream and Austrian economics.

3.2 Aggregating Value

Given the personal nature of value judgements and utility it is difficult to compare the values of subjects. Value is contingent on time and circumstance. The value a subject professes to a species now may be different from that in ten years time or three years ago. Given that value is subjective it can not be known but it can be **observed**. Values underwrite the behaviour of subjects (Rescher, 1969).

Observing values for market goods is a matter of recording prices. A price imposes a cost on the subject. It implies a sacrifice in order for the subject to have their most highly valued object. The subject must give up another good (because of their budget limit) in order to have the object they want. The market system is able to assign dollar values to all market goods.

Consider the value claim that an object X is worth $100. What this means is a subject would have to be willing to give up a sample of objects, that also cost $100 in order to possess the object X (Perry, 1968). The value claim is therefore relative to this other sample of goods and not the dollars themselves. The dollars are a measure of value and are not intrinsically valuable (Perry, 1968).

Staying with this example, if two people are willing to buy this object it does not mean that it has the same value to them. One person could be willing to pay more for it, for instance. Further, we do not know that it will give the same level of satisfaction to both subjects. What we do know is that both people are willing to give up a total of $200 (at least) for the X. Hence the pair is willing to have $200 less of other goods so as to possess X.

Given values are subjective it is meaningless to adjust them into objective values. Thus, adjusting each payment by the person's income and use of the object is an **impossible** task. The claim that the value of the object, for two people, is $200 tells us what sacrifice, in total, our population (in this case two people) will bear to have the good. There is no pretension that these value claims have an objective quality.

Further, most objects (goods) people are familiar with, can be reduced to dollar-values. A stereo for instance, might have a dollar value of $2000. Dollars in this sense are a metric of value. Given that market goods can be reduced to this metric, a subject is also able to determine the value of non-

market goods. That is, a voluntary contribution to a non-market good implies that some quantity of market goods must be given up. A gift of $100 to an endangered species means that the subject is willing to have $100 less of market goods to assist the species.

It does not imply that a market exchange is possible with the species. For instance, if a species has a total value of $Z this means our population is willing to sacrifice at least $Z to assist the species. It does not mean that a business or agency could purchase the species for $Z. Dollars are restricted in this sense to a measure of value, and they measure what sacrifices people are willing to make. They describe, in reduced form, the underlying set of goods and services a subject would be willing to forgo in order to have the object they desire.

3.3 Environmental Ethics

Introduction

The neoclassical approach to valuation and ethics consists of the following premises: one, that value is subjective, and two, it is reducible to human welfare. Therefore the only thing that is intrinsically valuable (valuable for its own sake rather than valuable as a means to satisfy some ends) is human welfare. Ethics is involved when changes to human welfare are considered. For this reason, the ethical approaches usually associated with neoclassical economics are utilitarianism or contractarianism. These approaches are concerned with *human welfare* as an *outcome* of choice. However, the subjectivity of values implies that a subject is the best judge of their welfare. In consequence, neoclassical thought "values" individual freedom.[4]

[4] "It seems the destiny of individual freedom at the present time to be defended mainly by economists..." (Leoni, 1991)

Environmental ethics comes in many forms. However, it rejects all or some of the premises of the neoclassical approach. For instance, some writers in the field of environmental ethics argue that value can be objective. For instance, Rolston (1988) asserts that in the area of the natural world (ecology) objective values abound. Other writers reject the premise that only human welfare is intrinsically valuable. For instance, Singer (1976) argues that pain or suffering is intrinsically bad and should be minimised whether the affected subject is a human or animal. This rule however does not provide a decision guide to saving species. Species as a collective entity can not suffer or feel pain. Hence, Singer (Johnson, 1991; Gunn, 1990) is usually thought of as contributing to the field of animal rights rather than environmental ethics.

A common denominator of the environmental ethics approach is the view that the environment can be *intrinsically* valuable. Recent legislation in New Zealand (for instance the Conservation Act 1987 and the Resource Management Act 1991) asserts that ecosystems (and their components) have intrinsic value. At this legislative level, the subjectivity of value is not rejected. Instead, this legislation argues that the environment can still be valuable even if it does not contribute to human welfare. That is, even if there is no instrumental value the environment still has intrinsic value. It is valuable for its own sake or of right.[5]

The rejection of neoclassical premises on ethics and value usually leads to quite different policy conclusions. As a result, an environmental ethics approach can be thought of as an alternative means to value species or an approach to supplement the neoclassical axiology.

[5] "Intrinsic Values", in relation to ecosystems, means those aspects of ecosystems and their own constituent parts which have value in their own right. Resource Management Act 1991 *Sec 2.*

Therefore a non-economic (environmentalist) stance could be used to discriminate between species that are candidates for recovery programmes. These non-economic stances range to the extreme deep-ecologist stance which accepts no trade-off between the environment and human activity (Kuipers, 1993; Bidinotto, 1993). The deep-ecologist view is that human welfare has little or no ethical content. Humans are likened to a disease infecting the planet. At a more moderate level is the belief of some that the environment carries additional value in a way not perceived by an anthropocentric stance (Johnson, 1991). This demands more constraints on human activity and more conservation.

Intrinsic Value of the Environment

The common feature of these views is that the environment can carry value. This value is not related to human welfare but may exist independently from human valuers. This value further carries some degree of moral weight (that is we ought to take this value into account when making decisions). Further, traditional expedient ways of looking at the environment are immoral (as they tend to be anthropocentric and exploitative). The environment should thus be valued intrinsically (Johnson, 1991)[6]

If this ethical case is taken as given it has consequences for the level of conservation and the types of policies likely to be approved. In economic analysis, the optimal level of conservation would be determined where the marginal value of the last dollar spent on conservation was equal to the marginal cost of achieving that level (**Fig. 3.1**). This is E1 on the diagram. The value considered would only be that relating to human welfare and would ignore the claim for environmental intrinsic value.[7]

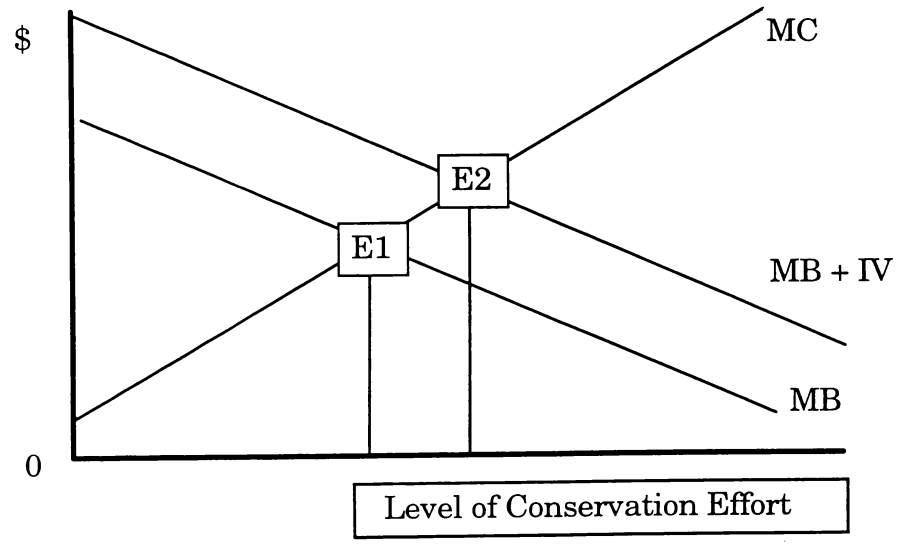
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[6] "...,we also believe that there is intrinsic moral significance in wilderness, ecosystems, species, and so forth, in addition to their significance for humans..." (Johnson, 1991 p4).

[7] The marginal benefit attributed to conservation may include (or be totally made up of) some values not included in market transactions.

If the intrinsic value of the environment is taken as significant and thus included, this will produce a rightward shift of the marginal value function. In consequence, the intersection of the marginal value schedule and the marginal cost schedule will be at a higher level of conservation (**Fig. 3.1**). That is, E2 on the diagram. This is clearly at a level of conservation effort greater than the case above. Thus at the very least, the claim for intrinsic value if accepted, leads to a higher level of conservation.

Figure 3.1. Optimal Conservation Effort:
Impact of Intrinsic Value



At a more extreme level, that of a Deep-Ecologist, the optimal level of conservation would be without regard to the costs of provision. That is, the preferred level of conservation would be best described as lexicographic. All conservation above that level is strictly preferred to any level of conservation below that level. In this view, the value of human welfare is heavily

discounted. Quite simply, human welfare is not accorded much moral status by advocates of this environmentalist position[8].

In addition to the effect on the level of conservation, the types of policies approved of would also be different. For instance, the economic approach tends to be a consequentialist system. That is, policy is directed at achieving a particular goal with the highest possible level of efficiency. However, this tends to judge policy tools on one dimension only. That is, these tools are compared on the basis of efficiency. The tool that is most cost-effective in achieving our policy goal will be the one preferred. A subject who feels that species conservation has a moral dimension would be prepared to be more deontological. That is, also place values on the feasible policies as well as their outcomes.

A consequence of this is the rejection of policies that may be very effective but are judged immoral (Ng, 1988). When Comalco started its sponsorship deal with the kakapo- offering an extraordinary level of support- some conservationists stated they preferred that the kakapo become extinct rather than accept Comalco's involvement (Westaway, 1990; Hutching, 1991). Commercial trade in some New Zealand species is opposed on the grounds that it is immoral. This is despite the concession that it may be an effective recovery tool (Anon., 1990a).

Clearly the financial assistance of Comalco should help the kakapo stave off extinction. The commercial trade of native species could also help some species (Bennett, 1994). The input of private sector funds and interest has the potential to increase endangered species' populations. Despite the

[8] "...We are not interested in the utility of a particular species, or a free-flowing river, or ecosystem to mankind. They have intrinsic value, more value- to me- than another human body, or a billion of them. Human happiness, and certainly human fecundity, are not as important as a wild and healthy planet...Until such time as Homo sapiens should decide to rejoin nature, some of us can only hope for the right virus to come along". Quotes by David Graber in Reisman (1993).

probable conservation gains from company sponsorship this involvement as seen as immoral. Likewise, there is great resistance to the policy of abandoning some species to enhance the survival chances of others (Williams, 1986).

When subjects treat the issue of species' preservation as one with a strong moral element a number of problems are introduced. There is the identification of environmental decline with commercial activities, hence an antipathy towards placing dollar values on the environment. This is consistent with bidding behaviour of subjects in Contingent Valuation surveys. Protest bids figure here as subjects reject an association with dollars with wildlife (Stevens *et al.* 1991a, 1991b). In Stevens *et al.* (1991a) a proportion of the protest bids focused on the immorality of money measurements. Further, 44 percent of the respondents felt that the level of wildlife preservation should not be determined by the funds available. A significant proportion of bidders therefore are proceeding without acknowledging the costs of conservation activity. In a survey conducted in Moyle (1991) strictly ethical reasons were the most popular motivation for species conservation over other, more utilitarian rationales.

This suggests that bids obtained under these types of surveys may not be a reliable indicator of a subjects' values. The economic approach has two problems with incorporating ethical bids. The first is the interest the 'moral' subject has in the means and not just the consequence of the bids. Such surveys may therefore be very sensitive to the payment vehicle adopted.

The second problem is the assumption of *reduction*. It is axiomatic to the neoclassical approach that a subject's welfare is able to be represented by a "utility function". Thus, all the values of a subject are reducible to this one-dimensional function (Etzioni, 1988). This utility is measured, generally in dollar terms, given knowledge of choice behaviour. However, this can be

criticised on two grounds. First, that "goal orientated" behaviour is not able to be represented by this function (Collard, 1978; Sen, 1982, 1987). The second is that "moral choices" are different from ordinary consumption choices (Etzioni, 1988; Moyle, 1991). There are fundamental differences in the way these two categories of choice affect welfare. While it is conceivable, it does not follow that a subject *can* reduce ethical values with preference values to a single outcome (Harsanyi, 1988). Subjects may instead apply "rules of behaviour". For instance, rules on tithes provide an approach to combine charitable giving with a subject's own consumption. This eliminates the need to continuously compare the welfare gains from charity versus one's own consumption.

Therefore, by their very nature, observed trade-offs between "ethical choice" and "own-consumption" may not be particularly reliable (Etzioni, 1988). If a subject is having to act against their moral values their bids might be inflated through *implicit* side payments simply to induce such a trade. That is, a money trade off with nature is a particularly unpalatable option for some subjects to consider. The bids these subjects will make will have two elements. These are the welfare change induced by the policy and the side payment the subject introduces to overcome their distaste in the trade-off. This makes it difficult to determine the welfare change of a particular *ends* as the bid incorporates a valuation of the *means*.

Problems with reduction are encountered in other areas of economics; for instance in expected utility theory (Machina, 1987,1989). Subjects find it difficult to reduce an outcome and probability measure to a single outcome consistently (for instance, the Allais Paradox). Such a difficulty might also be experienced by a subject having to reduce their preferences and moral values to a single welfare outcome.

The existence of a moral sentiment by some subjects to species conservation is attested to by the behaviour of conservationists (Anon., 1990a, 1990b), as well as writers in a variety of fields (Horsley, 1991; Rolston, 1988; Johnson, 1991). The assumption that subjects can reduce their ethical beliefs, along with preferences, to rank-welfare outcomes is tenuous. The behaviour of subjects confronted with the moral dilemma of trading off their moral beliefs with their consumption preferences makes for poor choice behaviour (Etzioni, 1988).

However, the aim of many writers in the field of environmental ethics is to go beyond this moral sentiment, which is extremely vulnerable to subjective reduction (Rolston, 1988). A moral sentiment towards the environment can theoretically be observed, even if this is difficult. Voting, lobbying and other budget allocations should permit observation of the value a subject places on species conservation even in a moral manner. It may be that subjects will show more sensitivity to payment vehicles. For instance, they may be opposed to direct charging if they feel that the benefits of species conservation are widespread publicly. It might also be the case that framing effects are important. That is, the way the survey frames its questions and background may be very important for the subject. The style of questions used may evoke more of an affirmation of moral beliefs than consumption preferences (Blamey and Common, 1993). Conceptually however, this just adds a level of complexity to the problem of observing values. It is not a sufficient reason to reject the neoclassical premise of subjectivity. That is, underlying neoclassical assumptions about the subjectivity of values are not seriously challenged.

The existence of a subjective, moral sentiment would not alter the level of conservation in the case above (**Fig. 3.1**); ie. where the marginal benefit equals the marginal cost (E1). The aim of the foray into environmental ethics is to produce objective or intrinsic values that would justify a *higher*

level of conservation without regard to the impact on human welfare. The difficulty with the subjective approach is that it is just an explanation for the *motivation* to conserve more. It carries no obligation to those who do not share a moral sentiment towards species conservation. It can not be used to require these people to achieve a higher conservation level.

The task writers in environmental ethics have set themselves is to establish that the environment can carry value in an objective or intrinsic way, so as to immunise their thesis from subjective, instrumentalist reduction (Rolston, 1988). This task has several parts. These include: establishing why a shift away from anthropocentric values is required; what makes traditional (especially subjective) ethical theories inadequate; what it is in the environment that carries value; how these values relate to other values (for instance, human preferences); and what sort of policies are justifiable under the new ethic.

It is outside the scope of this thesis to examine all proposed environmental ethical systems in detail. However, this thesis will concentrate on two to illustrate how they have attempted to cope with the various parts to the task. These will also be criticised and it will be established that the problems facing an environmental ethics approach to policy are in fact impossible to solve. This will confirm the neoclassical stance adopted for this thesis.

Environmental Crisis: evidence for and against.

A shift in the current anthropocentric ethical system is required by the current decline in the environment (Rolston 1988, Johnson 1991)[9] In Rolston (1988) it is claimed (without referencing the source) that more

[9] "As a great and growing number of people are aware, we have gone a long way toward degrading our environment, and in many ways the situation is getting worse. That needs no elaboration at this point..." (Johnson, 1991).

people are starving than ever before. This view of increasing environmental degradation can not be taken as a given. Instead, it is a rather contentious point.

For instance, Simon (1980) demonstrates that food production has been rising, food prices falling and that the incidence of famines has been falling. One estimate in Simon (1980) indicates that about a tenth as many people died of famine in the third quarter of the twentieth century as in the last quarter of the nineteenth. This is a decline in relative and absolute terms. This supports the notion that famines are declining in duration and frequency. Many famines to date, notably in the Horn of Africa have not occurred because of environmental problems, but through wars and policies of their governments.

Further, in Simon and Kahn (1984) considerable scientific evidence is mustered to demonstrate that environmental problems are improving and **not** deteriorating. The status of the environment has not checked population growth rates, availability of space, lifespan measures or agricultural output. Forest cover has been increasing in North America. Tropical deforestation has been declining (albeit it is still positive) and single satellite pictures of the Amazon do not constitute appropriate trend data. Bailey (1992) presents a rather dismal record of predictions by environmentalists. Environmental catastrophes are being continually postponed, but the myth of the pending environmental crisis is sustained.

With respect to the topic of species loss, the evidence is not clear (Simon and Kahn, 1984). Actual losses depend crucially on a number of assumptions. These typically involve tropical forest loss. To estimate current species loss it is necessary first to make assumptions about current habitat loss, the relative abundance of species in these habitats and the ability of these

species to migrate or occur elsewhere (Mann, 1991). Estimates of species loss can only be very speculative at this stage (Mann 1991).

Not only is the view of environmental degradation contentious, but it provides a weak basis for disputing anthropocentric ethics. First, environmental degradation affects human welfare but the point of a new environmental ethic is to attribute value independent of human welfare. Second, to the extent that environmental problems do occur it does not follow that anthropocentric values are a cause. It is possible to argue that it is the rejection of the neoclassical stance (with its emphasis on incentives) that prevents solutions to these problems (Moran, 1992).

Underwriting the neoclassical system of exchange and value is a set of property rights. Without rules on property, value can not be obtained from exchange. Exchange depends on clear rules on the transferring of property. One subject must be deemed to own it initially and title must pass with voluntary exchange. Without clear rules that are both enforceable and enforced, costs and benefits from production and consumption may lie in the public domain (Alchian, 1989).

For instance, it is the absence of clearly defined rules on property that contribute to over-fishing in open-access fisheries. An individual cannot own the future fish stocks they might permit to reach maturity by more prudent fishing in the present. These fish stocks may be enhanced by the individuals restraint, but all other fishers can share in this improvement. The prudent individual thus bears all of the costs (less fish caught in the present) and captures only a diluted benefit. Therefore, the incentive to use the resource wisely is absent. Property right solutions have tended to improve the incentives of private fishers; for instance, in the Chatham Island paua fishery seeding is now carried out by the quota holders (Hide and Ackroyd, 1990).

Pollution is also a problem where inadequate rules on property (ownership of discharge) reduce the incentives to clean up. Likewise, the forest owner that maintains spotted owl habitat bears the costs (foregone timber) but can not capture the benefits (welfare gains to U.S. households). Where property rights are clearly defined, agents are forced to take into account their actions on the environment, and can capture more of the benefits associated with environmental enhancement.

In consequence, the two premises "justifying" a shift in environmental ethics based on the event of "environmental crisis" are not well supported. It is perhaps more accurate to say that the belief that a new ethic is required occurs prior to the proposed new ethic. In Johnson (1991) it is conceded that this belief occurs prior to the attempt to design a new ethic. This prior belief is of course, not a sufficient ground to reject all environmental ethics approaches. However, it increases these proposed ethical systems' vulnerability to subjective reduction.

Environmental Ethics: approaches

The other premise on which a new environmental ethics approach can be erected is, that the human exclusion of the environment and its components from the moral world, is an arbitrary and perhaps discriminatory approach. Therefore, if moral value can be found in the environment it will not be possible to ignore this in setting policy.

This premise is superior to the ones concerning the impending "environmental crisis". It also requires that traditional ethical systems (especially those with subjective bias) be rejected. For instance, utilitarianism, Kantism or contractarianism must be shown to be seriously

flawed. These approaches vindicate or contribute towards, immoral outcomes, with respect to the environment. Demonstrating flaws in traditional ethical systems will make the construction of a new objective ethic much less contentious.

For a utilitarian, a moral action must improve human welfare as a consequence of that action. Actions, which do not improve human welfare, are not considered moral. Utilitarianism is therefore a consequentialist system. The atomic unit of utilitarianism is the individual. It is also a reductionist system, that is outcomes can be reduced to a single measure of welfare which is *utility*. There are a considerable number of forms of utilitarianism from the classic, hedonistic through to neoclassical preference utilitarianism.

The utilitarian approach has been adopted by the animal welfare movement. For instance, utilitarians perceive pain (and suffering) to be a bad. For the proponent of animal rights, the distinction between pain-in-humans and pain-in-animals is irrelevant. Who experiences the pain is not relevant to its badness. While this premise is not universally accepted its principal problem is it does not justify saving holistic entities (a species can not experience pain) and places no additional weight on saving endangered species (Gunn, 1990). Thus a programme to poison rats may involve an unacceptable level of suffering one the part of the rats. An endangered species predated on by these rats, may in virtue of its lower numbers and lesser nervous organisation, experience less suffering if it is predated to extinction (for instance, giant wetas make good snacks for rats).

Utilitarianism can be criticised on a number of grounds. By concentrating on consequences, an action may be judged moral even if the intent was not. The reductionist aspect is possibly too impractical to hold in all cases.

These criticisms are not so important that they refute the premise that human welfare is morally relevant and that suffering has a moral dimension. While controversy is attached to whether the suffering of animals is significant no wholesale rejection of utilitarianism is evident in the environmental ethics discussed below. For instance, the approach of Rolston (1988) is to argue that human values lie in the **category of culture**. Subjective, human value systems can lie wholly within this category without problem. The problem lies in applying these values to the **category of ecology** where objective values exist. Hence, a categorical mistake is made if subjective values are applied to the environment or objective values to human life-styles.

In Johnson (1991) the approach is also to include utilitarian insights. He argues that it is interests which are morally significant. Human preferences and welfare states are a sub-set of this larger set of interests. Prudent desires follow from interests; but these desires do not define the limits to a subject's interests. Interests are an integral part of human life processes. Utilitarian ideas therefore provide part of the moral over-view but do not define its entire scope.

Contractarianism is also criticised extensively by Johnson (1991). For the contractarian, moral rules are those that are agreeable by all members of society. That is, they are contracted into either implicitly, explicitly, tacitly or hypothetically (Gauthier, 1990). Each agent takes their welfare into account when deciding on these rules. These rules are intended to be universal in the sense they apply to all contracting agents. Unlike a utilitarian however, each agent is constrained by the rules agreed upon by others. A universal set of rules must therefore be agreeable to all. Therefore, each subject must take into account the welfare of others as they attempt to maximise their own welfare. Rules others find disagreeable are unlikely to be instituted.

Contractarianism therefore argues that moral rules are rational outcomes (mutually agreeable). However, they must also be subjective as they depend on each subject's welfare outcome. A moral agent is someone who follows these rules, which tend to enhance human welfare. This excludes most life-forms and all holistic entities as being morally significant. A species can not contract into any set of rules, and few organisms are able to appreciate moral principles. Therefore, the only moral value the environment can have is that attributed by human agents in the social contract. Thus, according to contractarian arguments, any moral value to the environment has a subjective origin.

The contractarian approach is consistent with Rolston's (1988) argument. Both would agree that the objective value for the environment is not going to be found in extant ethical systems. However, Johnson is more aggressive. Johnson musters three broad arguments against contractarianism. The first is that morality is reduced to a trade-off between contracting agents. The second is that there appears to be no reason why subjects should regard the contract as binding. The third is that the contracting group could arbitrarily decide to exclude other subjects from the contract. That is, contractarianism is subject to the charge that it promotes discrimination. Complementing this are Johnson's arguments that rationality is not a precondition for moral acts. That is, it is not necessary to know an act is moral for that act to be moral (Johnson, 1991).

The first argument is, as Johnson recognises, more of a reaction than an argument. However, Johnson (1991) does not appear able to detach himself from this point of view. For instance, in his summary he describes contractarianism as an "unsavoury system". More salient to this criticism is that Johnson (1991) commits a conceptual error by characterising the

contract as a trade-off. In this view, the social contract is a zero-sum game where the gains by "winners" are offset by losses by "losers".

However, a contract can make both agents better off. Consider a "Prisoners' Dilemma" problem.

Figure 3.2

		Player 2	
		Confess	Not Confess
Player 1	Confess	¼,¼	1,0
	Not Confess	0,1	¾,¾

In problems of the Prisoners' Dilemma type, rational choice leads to the worst possible outcome. Both players would choose to confess (the dominant strategy) which leads to the *lowest* possible payoff to each (Fudenberg and Tirole, 1991). A contractarian solution would be to have the players select a binding strategy agreeable to both. While this is not an option in the classic Prisoners' Dilemma, it is where contracts are enforceable and enforced. A contractarian solution would be for both players to "not confess". This "cooperative solution" achieves the highest combined payoff for the players. Note that the ideal strategy for an individual player would be if their opponent choose to "not confess" while they decided to "confess".

The contractarian solution in this case makes both players better off compared to the actual Nash equilibrium[10]. That is, they both gain by remaining silent as opposed to the individually rational strategy of confessing. The contract is able to make everyone better off and in consequence no one loses.

[10] "A Nash equilibrium is a profile of strategies such that each player's strategy is an optimal response to the other players' strategies". (Fudenberg and Tirole, 1991; p11).

Therefore the claim that Johnson makes that this is unsavoury is contentious. The contract acts to improve all parties' welfare. This does not justify the unsavoury label. Further, a trade-off is often dictated in real life situations. Unless a subject's set of moral duties is very small, they will find it impossible to fulfil all of their obligations (Moyle, 1991). A trade-off is thus going to be a necessary feature of choice behaviour. There are no grounds to judge this negatively.

The aspect of "being unsavoury" is not essential to Johnson's case. However, it does illustrate a poor understanding of contractarianism. This is exacerbated by the absence of any references to contractarian thought. Indeed Johnson's criticism of contractarianism is limited to explicit or tacit contractarian thought. This is despite the dominance of hypothetical contractarianism, for instance with the Leviathan (Hobbes, 1952) and A Theory of Justice (Rawls, 1974).

The second criticism by Johnson is based on the obligation to keep the contract. He suggests that this obligation has two possible sources which demonstrate the inadequacy of contractarianism. The first solution is that there must be a moral imperative outside the social contract. This exogenous imperative is not accounted for in the contract, hence limiting the contractarian approach to ethics. The second source of obligation is through expediency. Agents keep the contract through the motivation of convenience. It is convenient to adhere to the contract which implies the moral obligation is lacking. In this situation, subjects are merely simulating moral behaviour, and not demonstrating a commitment to be ethical.

Johnson therefore ignores the possibility that the obligation to keep the contract is endogenously generated. That is, altruistic strategies may be equilibrium outcomes which rewards the genuine altruist (Axelrod, 1984). This is more consistent with a socio-biological interpretation of the evolution

of cooperative and altruistic behaviour (Trivers, 1971). In a socio-biological context "virtue can not be its own reward". Virtuous acts must somehow generate benefits for individuals otherwise altruists would be selected against by egoistic strategies.

The relevant game to consider is the repeated game where players may cooperate or cheat on their opposite. Each player will be concerned to maximise the flow of payoffs through time (repeated games). The option to punish a cheating player by responding tit-for-tat lowers the payoff from cheating (Axelrod, 1984). Cheating may be the optimal strategy (as in Prisoner's Dilemma) for a one-off game. It is less than perfect in a repeated play game where punitive strategies can be applied. Indeed the threat of punitive action may be sufficient to hold both parties to a cooperative contract (Axelrod, 1984).

The social contract may have a number of devices which encourage moral acts to persist. In the Hobbesian case, players relinquish some of their power (natural rights) to a central authority who then enforces the social contract (Hobbes, 1952). Thus, the state acts to secure subject's life and property from arbitrary seizure. Further, by acting in a moral manner the player is communicating to other players that they can be trusted. This increases the confidence other players have that this particular subject will not cheat. This is necessary in a society where subjects are mutually interdependent. Exclusion from this community by committing acts contrary to the social contract reduces the gains from interaction.

In consequence, a moral sentiment is important and rational in a contractarian society. The gains from interaction can only be had if subjects are confident others reciprocate. Much of the gains from interdependence in society would be lost in *transaction costs* if subjects could not be trusted (Casson, 1990). That is, monitoring the behaviour of people who are

unlikely to act in an altruistic manner is costly to society. Therefore, those societies that do evolve moral sentiments are welfare enhancing.

Biology gives examples where cooperative behaviour is advantageous to the parties involved even though cheating would appear to be evolutionary dominant. This implies that a hypothetical contract can be very effective at generating **endogenous** incentives. Animals can't agree that there is an obligation to *not* cheat on a symbiont, but will act altruistically anyway. For instance, the groper fish and its cleaners have a reciprocal altruistic arrangement (Trivers, 1971). The groper fish could gain by consuming its cleaners but refrains from doing so. The transaction costs of finding cleaners are high and the gains from repeated interaction outweigh the one-off gain of an easy meal (Trivers, 1971).

In socio-biological terms humans have all the necessary attributes to make altruistic behaviour (in a contractarian sense) an evolutionary advantage. In an uncertain hazardous, world it is expedient for subjects to perform altruistic acts (Trivers, 1971); that is, there are gains from fulfilling the social contract. Behaving morally is a means by which a small cost or risk is borne now in order to reduce the probability that a larger risk or cost is borne later. When a subject saves someone from a hazard this reduces the probability the subject will die later from the same hazard. Someone who fails to act altruistically may be abandoned by others who can not trust them (Trivers, 1971).

Altruism is a stable strategy therefore in repeated play games. Humans have many of the attributes that would make altruism an evolutionary advantage (Trivers, 1971). For instance, people have relatively long lives (meaning the chances of being exposed to hazards in the future is high) The dispersal rate is quite low which implies others can monitor an individuals

behaviour easily. An altruist will be rewarded and a cheater punished (Trivers, 1971).

This suggests that the strategic double-cross, which Johnson fears is a flaw in the contractarian approach, may not arise or be allowed to persist. The possibility of repeated play amongst the same set of subjects reinforces adherence to particular rules. Immorality is explained as actions in opposition to the agreed social contract. In the long term, subjects who cheat will make themselves worse off.

In the contractarian view, moral rules are those mutually agreeable to a set of contracting subjects. These rules are stable, equilibrium points because the social contract generates the incentives to adhere to them. Johnson commits another conceptual error by arguing that contractarianism would justify a double cross. While unobserved cheating does generate benefits to the player, the social contract does not condone such actions. Johnson is confusing the flow of benefits from altruistic behaviour to single acts by subjects which pass a cost-benefit calculus.

To confuse matters further, Johnson argues that rationality is also not a basis for morality. This is necessary to judge the Kantian approach inadequate. In Johnson (1991), moral acts can be attributed to subjects who have no idea that what they are doing would be classed as moral. This implies moral intentions or a sentiment are not a necessary condition for a moral act. Thus, a subject simulating morality is indistinguishable from a subject intending to be moralistic.

Johnson bases his case on the behaviour of shocked rhesus monkeys (Wechkin *et al.*, 1964; Johnson, 1991). He argues that monkeys that go without food rather than shock another monkey are performing moral acts. That is, a cost is borne (hunger) even though no benefit to the monkey flows

from this act. However, the biological interpretation is that the monkey **must** benefit from this action (otherwise this behaviour could not evolve).

The benefits are likely to be that, first, the monkey reduces the probability it will be shocked later (Trivers, 1971). The possibility of being shocked is a threat all monkeys in the colony face. To reduce the odds of being shocked later, the monkey concerned goes hungry, rather than shock its partner. In game theoretic terms, this is a repeated game and not a one-off game. The second benefit is genetic. The monkey has an interest in the other monkey's welfare as they share genes in common. All monkeys in the case used are from the same colony (Wechkin *et al.*, 1964). It is a biological advantage to have an interest in relatives because they also represent a means by which the particular organisms genes will be passed on. Thus injuring a relative is ultimately injurious to the perpetrator.

Johnson is therefore not making a strong case to prove the inadequacy of the contractarian approach. His case that moral acts do not produce benefits is untenable and his case is weakened by the separation of intent from action. That is, an act could still be moral even if it was not intended to be.

His third criticism is that contractarianism could be used to exclude some subjects from the benefits of the social contract (for instance, slaves). Thus, contractarianism is unstable as it lends support to discriminatory behaviour. This criticism is inapplicable to the hypothetical contractarian stance. In this case, subjects do not know their status in society before the contract is entered into (Rawls, 1974). Each subject faces the possibility that they will be disadvantaged by discriminatory rules. Thus, as contractarianism is a rational procedure, each subject takes into account the impact of the contract on minorities. This prevents such discrimination as being justifiable in a contractarian case.

This discussion illustrates that the design of a new environmental ethic has problems with existing moral theories. There are two options available to someone wishing to derive a new environmental ethic. The first is to prove that existing moral theories are inadequate to deal with all moral issues. This goal is too ambitious as Johnson inadvertently demonstrates. The second option is to demonstrate that while existing moral theories are adequate they do not describe the whole set of moral entities. Thus while human welfare belongs within the field of morality it does not make up the complete measure of morality. Other factors are morally significant too and some of these relate in particular to the environment. Thus Rolston's approach is more convincing. For the time being, human welfare is significant within the category of culture but this does not exclude morally significant issues lying outside this measure.

Difficulties with Environmental Ethics Approach

The next task in the attempt to create an objective ethical view of the environment is to identify what it is in the environment that carries value. That is, what properties exist that justify attributing intrinsic value to the environment. This value-carrying property (or properties) exists independent and separate of humans. This separate existence guarantees that value will be objective and immune to subjective reduction (Rolston, 1988).

This separation of value and humans however, creates a dilemma. This dilemma is one of recognition. How is it possible to perceive this value from the perspective of human subjectivity? This is manifested in the variety of proposals pertaining to the value carrier. While many environmentalists wish to establish the existence of intrinsic value and moral duties to the environment there is no consensus on how this can be done (Johnson, 1991).

The dispute over the precise value carrier also increases the vulnerability of this approach to subjective reduction. That is, a subjectivist can claim that this conflict demonstrates that it is the subject's differing values producing disagreement. Hence, the underlying source of value is still subjective.

The second dilemma confronting the ethical approach is the scope of what is being valued. The environment encompasses many things for which a single property is difficult to define. It includes individuals such as members of a species as well as holistic entities such as the species themselves. It includes sentient things, such as intelligent mammals and also the non-sentient, such as plants. It includes the living, as well as the non-living, such as mineral cycles.

This scope has frustrated attempts to identify a single property that carries value. Conventional ethical theory concentrates on individuals and rational decisions. Little of the natural world can deliberate on the moral course of action and few elements environmentalists are trying to claim value for, are individuals. This implies that there are two ways to deal with the scope. This is to have many properties that carry value or a single property that is extraordinarily elastic. That is, it can be defined (stretched) to cover the entire set of environmental things.

The problem of the multiple-property approach is that it magnifies the dilemma of recognition. Instead of having to identify one property only, several properties now have to be identified. Further, it will be very much more difficult to determine the morally appropriate outcome. For instance, if there was only one property, this would have to be maximised to obtain the most valuable course of action. However, multiple-properties would require a set of weights as well so that each property could be compared to another. This will be impossible to achieve for a set of properties designed to be so aloof from human concerns.

The single property approach therefore overcomes the practical disadvantages of determining multiple properties. This is achieved in Johnson (1991) who argues that interest carry morally significant value. Rolston uses an ecological measure of value, thus is concerned with effects on the biosphere. Value for Rolston is *also* linked to sentience, the more sentient a creature is then its will possess a greater set of values relative to the non-sentient.

The problem with the broad, single property (assuming the dilemma of recognition can be solved) approach is a lack of discrimination between the various sorts of values. That is, the value carrier is so loosely defined that all values could be morally significant. Take the case of interests. A human has a large set of interests, some of which relate to individual preferences. If interests are the moral value carrier then all of a subject's interests could be considered morally significant. Thus, there is some obligation or moral duty to meet a subject's interest in the consumption of say, fast cars. While it is common to separate moral values from other values this now presents a difficulty. All of the values held by a subject could be considered of moral weight if our criterion is broad enough. Johnson's interest argument is not discriminatory enough to be operational.

Thus, the conflict here is to have a concept broad enough to cover all of the environmental concerns held but restricted enough to eliminate the unlikely and absurd. Johnson (1991) finds it necessary to adopt a more restrictive concept of interests than that adopted by Frey (1980). In essence, value has to be objectively attributed. This produces the argument that X is the value carrier because X is good for B, where B is part of the environment. Thus X is known to be valuable because doing X is good. For instance, if X are "interests" then looking after B's interests is morally good because it is good for B.

The set of objectively valuable objects is restricted to prevent moral values being a universal set. For instance, a tractor could be said to have interests (Frey, 1980) but few would agree that these were morally relevant. Johnson (1991) argues that only living things can have interests because man-made artefacts have interests attributed to them. Living things have inherent interests. In Rolston (1988) the use of the two categories culture and ecology, restricts the use of these objective values to the natural environment.

This reduces the vulnerability of this ethical approach to make all values moral. However, it is still not immune from unlikely cases. For Johnson, human interests could still have moral weight. For instance, having a fast car is good for me because it brings me pleasure. This is unlikely to have a moral weight. Society could not be obliged to meet this interest on moral grounds.

While the morally good can be considered the same as the objective good for a particular component of the environment, two major problems exist. These are the teleological and informational. To know what is going to be the most valuable course of information requires a general equilibrium model of the biosphere (or major ecosystems at least). What is good for a particular individual is not necessarily good for a population, species or ecosystem. Mutation and selection are good for populations but bad for the affected individuals. To determine what is "good" requires that all of the components and processes of an ecosystem be modelled.

This, for all but the simplest of ecosystems is impossible. The information simply is not available. Many species remain undescribed at a basic morphological level. There is no information on their population genetics (for instance, mutation rates) or ecology. Threshold effects are very difficult

to model, especially when threshold limits are not known. Hysteresis effects add considerably to the basic level of complexity, as well as stochastic effects. The informational requirements are excessively high to model what is "good" for an ecosystem.

Further the claim that human modification is a bad is difficult to sustain on objective grounds. Replacing forest ecosystems with pastoral (in New Zealand) merely changes the ecosystem in question. Some species are disadvantaged while others, for instance the native grass grub are considerably advantaged. It is impossible to judge one ecosystem as morally superior to another. Balancing the winners and losers is impossible.

The second problem with determining the "good" for an ecosystem is a teleological one. The physical and meteorological environment is always changing. Evolution brings forth new threats and possibilities all the time. These are not teleological processes. Hence it is impossible to know that what is good for a particular component of an ecosystem now will be good in the future. The future direction of evolutionary processes can not be known thus the "good" for the environment can not be realistically determined (other than the extreme case of extinction of life on this planet).

If it is assumed that the recognition dilemma and the informational requirements can be solved there still exists problems with the relation between valuable objects. This exists within the category of the environment as well as how these relate to human activity (which is also valuable).

Within the environment, a matrix of value-ratios will be needed between all the components of the environment. How much more (or less) valuable is an individual of an endangered species to individuals of threatened species? How do ecosystems relate to endangered species? How do global problems

relate to local problems? Given that there is no universal agreement as to what carries value in the environment (Johnson, 1991), it is extremely unlikely that such a matrix could be devised. Indeed, without reference to choice behaviour or environmental manifestations of value ratios such a task is impossible.

Not only does conservation require that trade-offs be made among components and the environment, there must also be a trade-off between human activity and conservation. This Johnson regards as impossible (Johnson, 1991). Rolston creates an axiological scale to indicate a lexicographic scale of value. That is, a set of implicit *quasi-indifference* curves are proposed (Gorman, 1971). The dominant concern is effects on the biosphere through to less-global concerns through to society impacts. Individual values, as manifested by market behaviour have the lowest rank (Rolston, 1988).

A lexicographic ordering at least has implications for policy, unlike the approach by Johnson. For Johnson, a trade-off between forestry and habitat protection can not be solved. For Rolston, individual preferences to have timber rank less than the value of an intact ecosystem. Hence, the forest should be preserved.

Rolston's axiological scale is very contentious. Realistically, the separation of the category of ecology and that of culture can not be maintained. Scarcity implies choice and choice requires a trade-off between culture and ecology. The two can not exist separate to each other.

No justification is provided for this lexicographic ordering. Some kind of justification is required. Such an ordering implies that the environment's "good" and societies' "good" should always take precedence over individual values. This conflicts with several views on welfare. For instance, even

Rawls, a self-confessed socialist, argues that some individual rights must have precedence over the collective (Rawls, 1974). Such an approach is not consistent with the Pareto approach. Making some people losers so others can gain can not be claimed to improve human welfare. Objective values turn out to be superior to subjective in Rolston's state-of-the-world. This would produce perverse incentive effects for the economy (Perry, 1968) and lower standards of welfare.

It does not appear to be realistic that individual subjective values can be compared realistically with any objective value system. Decisions about orderings of value or exchange-ratios must be arbitrary until the environment can be modelled realistically. This is impossible hence the policy set offered by environmentalists must be arbitrarily determined.

While the policy set is unjustifiable (except on the grounds of impacts on human welfare) this does not prevent policy being proposed. More certainty is attached to the policy recommendations than the suitability of a particular version of an environmental ethic.

Given that practically no human activity is devoid of environmental impacts environmentalists feel justified in applying their objective values to a wide range of activity. All consumption, production, trade and exchange are valid targets. Rolston (1988) argues against electric knives and despite his concessions to subjective values, doubts more than 1 percent of the purchasers of these items were made better off. That is, the remaining 99 percent of purchasers are in error if they believe they did gain welfare.

The intrinsically, morally valuable environment stance advocates keeping species entirely in the public domain. Thus the collective good takes precedence over individual values. It also has perverse consequences. The neglect of property rights has not encouraged the preservation of species in

the past. For instance, the case of elephants in Kenya has been discussed above. When the benefits of having elephants around (tourism, ivory, meat) can not be privately owned the incentives to cooperate with poachers is extremely high.

Many environmental ethics approaches reject market solutions (belonging to the realm of subjective values) in favour of collective solutions (Simon and Kahn, 1984; Bailey, 1993; Bidinotto, 1993). However, the impact is to convert a resource into a non-resource. Individuals can not capture through exchange-markets, the benefits of prudent use of the environment. Instead, accelerated depletion of the environment is likely as the benefits of prudent use are denied to individuals. Hardin (1968) attests to the problems of managing common-property resources. Further, the governments can also "fail" through public choice problems (Bennett, 1994). Replacing the market with the government management does not eliminate the perverse incentives created by common-property resources (Alchian, 1993).

Discount rates are also criticised in these approaches (Rolston, 1988). Yet, without positive discount rates there would be little reason for proceeding with any species' recovery programme. Furthermore, low or zero discount rates also reduce the rate at which new, efficient technology is adopted. In the case of endangered species, all presumably have the same "moral interest" in avoiding extinction. If a benefit-cost analysis is applied at a zero discount rate then no project could be accepted. This stems from the eventual, certain extinction of all species on this planet[11].

The cost of preventing extinction therefore is infinity; ie. it is impossible. As a result it would be impossible given the infinite time frame implied by a zero discount rate, to save any species. Any benefits must be less than

[11] Modelling of biological populations generally shows that extinction approaches certainty as time extends to infinity. It should come as no surprise to learn that most species that have occurred on this planet are already thought to be extinct.

infinity otherwise all our resources would have to be applied to it. The ratio of benefits to costs is thus zero. It is only by adopting a positive discount rate and a standard project life-cycle that the costs of a programme fall below infinity.

An objective environmental ethic has to confront the incentives generated by human, subjective values. These incentives can not be overcome, as demonstrated by the failure of socialism.[12] Therefore, these objective values have to be imposed on people by some coercive procedure. This in turn produces moral conflicts if such an environmental ethic is adopted.

That coercion is involved is evident. In New Zealand Heritage Orders and the Forest Amendment Bill 1993 represent uncompensated takings by the government. It is not usually considered to be moral behaviour to coerce others or to deprive them of property.

Further, applying an objective environmental ethic is ultimately human welfare reducing. That is, as it occurs above a level that is subjectively desirable or optimal, less resources can be applied to other human activity. With human activity at a lower level than before, human welfare is less than what it could be. An objective environmental ethic can not avoid being coercive nor can it avoid reducing human welfare. Its moral status should not be accepted uncritically.

In summary, the objective environmental ethics approach is not applicable to species conservation problems. Its moral status is questionable. It is not obvious how the moral demands of human welfare be reconciled by the "moral demands of the environment". Further, it does **not** follow that if the environment is valuable in some objective way that this value has any

[12] "Socialism...has been the tragic failure of the twentieth century. Born of a commitment to remedy the economic and moral defects of capitalism, it has far surpassed capitalism in both economic malfunction and moral cruelty." (Heilbroner, 1993).

attached moral duty. That is, even if the environment has objective value that does not imply humans have any obligation to it. Given the broadly conceived notion of the value carrier in the environment it does not appear to have any ethical force (for instance, Johnson's concept of interests).

An objective approach, if able to be devised, would simplify the species conservation problem. That is, the policy task would be to maximise this property, say "interests". However, the task of devising such a system is impossible. The independence of this value from human existence implies there is a problem in identifying the value carrier in the first place.

The informational problems are enormous and the relationships between all the "intrinsically valuable" components of the environment is unclear. Especially unclear is how to relate intrinsic value to human, subjective values.

The scope of the problem presents a major difficulty. The environment has a very large number of components. Yet, all of these components are attributed some moral value. Unfortunately, there is too little in common among the sentient and non-sentient, living and non-living, individuals and holistic for a Cartesian approach to work. The effort to attribute moral value to all of this requires very generalised, plastic value-carriers.

At the policy level such an approach is extremely non-discriminatory. This is a product of the very generalised value-carriers proposed. All species have an interest in avoiding extinction and it would be "good" for all endangered species to act to promote their persistence. This does not tell us whether the kakapo or kiwi should have priority or more funding. Indeed, many of the policies recommended by such an approach would not assist species conservation at all.

Therefore, the environmental ethics approach has a number of unsolvable problems and is inapplicable to policy or worse. It can lead to outcomes that are the very reverse of those being advocated by environmentalists. It is coercive and unsuitable as a basis to proceed. In essence, it is only possible to apply "objective values" to the environment if we are prepared to drop human welfare as a policy outcome. By way of contrast, the subjective economic approach provides a much stronger, less coercive basis to analyse policy.

CHAPTER FOUR
SPECIES VALUATION APPROACHES

4.1 Subjective Approaches: Contingent Valuation Method

The Contingent Valuation Method or CVM (Mitchell and Carson, 1989) has been the dominant economic tool to derive values for wildlife (Young, 1991). It has found application to the Wisconsin bald eagle (Boyle and Bishop, 1987), the whooping crane (Bowker and Stoll, 1988), the striped shiner (Boyle and Bishop, 1987), the spotted owl (Hagen *et al.*, 1992), the New England bald eagle and the wild turkey (Stevens *et al.*, 1991a). In the United States values derived with CVM have rebuttable presumption in the legal system (Blamey and Common, 1992; Portney, 1994). The results of a CVM survey are thus presumed to be correct in liability cases unless proved wrong. Given the prominent use of this tool it is outlined in detail below.

Contingent Valuation proceeds from the assumption that the non-use values dominate the total value of a species (Bowker and Stoll, 1988). Further, the evidence that has been collected on environmental values supports this assumption (Young, 1991). That means that use values such as hunting, observing or photographing are trivial compared to non-use values. Non-use values include option values and existence values (Boyle and Bishop, 1987; Young, 1991; Portney, 1994). Non-use values may arise through uncertainty about the future and altruistic or benevolent attitudes towards the species (Young, 1991). Given that these values will rarely manifest in behaviour it is not possible to attribute value through budget allocations. Thus the CVM techniques posits a hypothetical institution by which values may be attributed.

Actual budget allocations are typically unobservable for non-use values, hence CVM is based around a survey format. For any resource R (where R is dominated by non-use value) the subject is provided with two states of the world (S_1) and (S_2). The first state, (S_1) lacks R and the second, (S_2) has R. The subject may be asked their **willingness-to-pay** (WTP) to move from (S_1) to (S_2). That is, how much they would be willing to contribute to assist the move. Alternatively the subject may be asked how much compensation they would need to approve a move from (S_2) to (S_1). This is termed the **willingness-to-accept** (WTA).

Theoretically (S_2) is equivalent to (S_1)+WTP and (S_1) is equivalent to (S_2)-WTA. Thus, the WTP and the WTA should be equal. As the only difference between these states of the world is R, the WTP and the WTA measures the value of R. In practise this is not the case. WTA is almost always larger than WTP (Randall, 1982; Knetsch and Sinden, 1984; Coursey *et al.*, 1987). This may be a result of different budget constraints in the two states (WTP is arguably more sensitive to income distribution) of the world (Cory, 1985). The nature of R may also affect the WTP bids. If R has few substitutes or is irreplaceable, subjects may need a very high level of compensation (Hanemann, 1991).

In the CVM studies of species conservation that have been undertaken, there has been total reliance on the WTP measure of value. For a non-market good such as a species, subjects are unlikely to be familiar with the welfare loss its absence would create. Further, a species is thought to be irreplaceable if it becomes extinct. This irreplaceability implies a lack of substitutes. Thus, if a WTA question was posed, the subject may inflate their declared value in light of this irreplaceability. On practical grounds WTP is useful as a conservative measure of value. It produces a result that is less sensitive to the charge it overestimates the value of an

environmental resource. Projects that pass a WTP estimate of value must also pass a WTA estimate of value.

The CVM technique also features a payment vehicle (Mitchell and Carson, 1989). This is the means by which the subject records their WTP or WTA. Selection of an appropriate payment vehicle is important. If the subject feels that the payment vehicle is inappropriate then their declared WTP or WTA will reflect this. Distortions in the declared value or a lower response rate to the survey is the likely result. Typically payment vehicles (for wildlife) have been voluntary contributions to trusts, mandatory rises in taxes and/or changes in affected product prices. Thus saving an area of forest may result in rises in timber and other wood product prices.

Thus, the CVM survey has a number of basic elements. These are the resource, the relevant states of the world and the payment vehicle. The survey can be conducted using telephones, mail-out questionnaires or on-site surveys.

The dominance of this technique is directly attributable to its ability to reveal non-use values. As indicated above, it is reasoned that non-use values will dominate the value of an endangered species. Small populations with restricted or remote access will typically reduce viewing and photographing opportunities. Hunting is not compatible with other uses and is determined to be less than the other values (Boyle and Bishop, 1987). Thus, should an endangered species have value this is likely to be attributed to non-use values.

The two most important components of these are option value and existence value (Young, 1991). Option value relates to the uncertainty about future states of the world. The subject may feel that sometime in

the future they might wish to make use of the species. They are thus willing to bear some cost to preserve this option (Young, 1991). Alternatively, with existence value, the subject may feel that the species has a right to exist or that they have some obligation to ensure its survival (Young, 1991). This means that they are willing to contribute to the species survival even if they never see it as resource for now and/or in the future. It is doubtful that these values are additive (Boyle and Bishop, 1987). That is, the different sources of value may have substitute or complement relations to each other.

There have been several CVM studies on endangered species recently. All of these studies have attributed value to a species indigenous to the United States. In Boyle and Bishop (1987) the Wisconsin bald eagle and the striped shiner (described as an obscure fish) were surveyed. Bowker and Stoll (1988) conducted a survey on the whooping crane. Stevens *et al.* (1991a, 1991b) conducted a survey on the New England bald eagle and the wild turkey. Hagen *et al.* (1992) performed a CVM study on the spotted owl and associated old growth forests. The species above are, with the exception of the striped shiner, all bird species. All of these bird species should be well known to the subjects. The bald eagle is the national symbol of the United States and the wild turkey is also historically important (ie. the Thanksgiving turkey). The whooping crane has long been subject to a recovery programme and this programme has been organised internationally (Bowker and Stoll, 1988)

The survey format has been exclusively a WTP one. The payment vehicle has in the majority of the above studies been contributions to a private trust. The only exception was the spotted owl survey which used a combination of higher taxes and wood product prices as the payment vehicle. All questionnaires were in a dichotomous choice framework. Thus, each subject was asked a randomly determined WTP value. They

had to respond either YES or NO to the posited value. The probability any subject would answer yes was related to the subject's value of the species.

Response rates to these surveys was varied. The highest was the Boyle and Bishop (1987) study with an overall response rate of 81 percent. The lowest was the Stevens *et al.* (1991a) with a response rate of 37.5 percent. The two studies (Stevens *et al.*, 1991a; Hagen *et al.*, 1992) with the lowest response rates had additional survey questions attached to the questionnaire.

In two of the studies the questionnaires stated that the probability of saving the species was a certainty subject to sufficient funds in the trust. These were Boyle and Bishop (1987) and Stevens *et al.* (1991a). In the spotted owl survey the probability of success was not given. In the whooping crane survey the probability of success was indicated at less than one. In all studies, the State of the World associated with extinction did not specify how long extinction would take to be achieved.

The CVM technique has not been free of criticism. In the above studies several specific problems occur. The first is that neither time or risk are accurately represented in the surveys. Thus the subject only knows that the species concerned will become extinct in one state of the world. They do not know whether this will take ten years or one hundred. It is known that the expected time to extinction for species is varied and it would be reasonable to presume that this would influence the subject's valuations. The existence of financial markets demonstrate after all, that people are willing to place differing values on present and future consumption. The sense of urgency accompanying a species facing imminent extinction differs from that facing extinction in the medium

term. Such effects are likely to alter subjects' bids if they were more aware of them.

In addition, risk is poorly represented. It can not be known with certainty that a species will recover after a recovery programme is implemented. For some species this will be more difficult than others. The kakapo in New Zealand is proving very resistant to a recovery programme (Chamberlain, 1993) while the black robin has experienced a large increase from a low-point of two females. In any case, it can not be known with certainty that a species will respond to a recovery programme. A species may be quasi-extinct by the time the study is launched. For instance, the heath hen in the United States became extinct after the population had risen from fifteen to two-thousand individuals (Halliday, 1978). That is, not even this rise was sufficient to offset the subsequent inevitable extinction.

This is likely to influence the willingness of a subject to contribute to recovery programmes. A programme with a 99 percent probability of success may have greater reported value than a project with only a 40 percent chance of success. Again, the behaviour of subjects in financial markets (for instance, risk premiums) indicates that risk is not something that should be ignored.

There are also generic problems related to the implementation of a CVM survey.[1] The observed outcomes may be biased for a number of reasons (Jakobsson and Dragun, 1989). It is possible that the outcome will be biased through the subjects behaving strategically. This means that the

[1] The debate over the suitability of CVM as a tool to value the environment, is characterised by a wide divergence in opinion. Hanemann (1994) and Portney (1994) tend to be qualified supporters of the technique. Diamond and Hausman (1994) are firm critics. These references outline the range of views held on this technique. However, this research is only concerned with the suitability of CVM as a technique to value endangered species.

subject believes that by deliberately misstating a value they can influence the actual policy that is determined. Given the subject is not asked to bear the cost of their decision there is no incentive to refrain from strategic play. For instance, a conservation-minded subject could be prepared to exaggerate their reported value if they felt this would increase the probability a conservation policy was adopted. As they will not be asked to meet the cost of this policy strategic play is encouraged. However, evidence of strategic behaviour has been difficult to demonstrate (Randall *et al.*, 1983).

Further, there is the problem of self-selection bias. In CVM studies it is common to have a response rate of between 40 percent and 60 percent (Mitchell and Carson, 1989). The problem this creates is that those who choose to respond are conspicuous by their interest in the resource. This groups that responds typically has higher education and income levels relative to the general population (Stevens *et al.*, 1991a). It is not obvious that it will be possible to generalise a population's value of the resource from this sample. The decision to participate in the survey by only some subjects thus produces a bias.

The studies above are free from two other forms of bias attending CVM surveys. These are the interviewer bias and the starting point bias (Boyle *et al.*, 1985; Mitchell and Carson, 1989). Where the survey is not dichotomous choice a starting point bias may result. This is common to bidding type surveys. Here, the initial bid causes a calibration about that value. Thus, subjects may declare different values of the same resource as a result of this initial bid. Where the initial bid is high, this tends to increase declared values and where it is low it tends to decrease declared values.

Face-to-face surveys may produce a bias also. This is caused by the subject being influenced by the interviewer. If the subject is hostile then this will produce a different declared value than if the subject was seeking to impress the interviewer. A similar problem may exist with the survey design. The subject may be influenced by the survey format and its associated information content when declaring their values (Samples *et al.*, 1986).

Further, the results obtained from these surveys are sensitive to the analysis that is undertaken. Where calculation of an average is undertaken the result is sensitive to whether the mean or median is used. The argument in favour of use of the median is that it eliminates outliers which (perhaps) are inaccurate valuations because of strategic or irrational reasons. The argument in favour of the use of a mean is that it captures the valuation of a small minority that place very high values on the resource (Boyle and Bishop, 1987; Bowker and Stoll, 1988; Stevens *et al.*, 1991a; Hagen *et al.*, 1992). The problem is that there tends to be a small proportion of very high valuations in these surveys. Distinguishing between the genuine and the misreported is impossible.

The treatment of non-respondents also creates a problem. There are two extreme solutions available. The first is to assume that non-respondents hold the same value for the resource as the respondents. Unfortunately respondents tend to be atypical. That is, higher income and education than the general population (Stevens *et al.*, 1991a). Assuming non-respondents have similar values to this group is inherently unrealistic. The second approach is to assume that the non-respondents have a zero value on the resource. This eliminates all non-respondents who might have had values on the resource but did not answer the survey for other reasons. The presence of protest answers has been demonstrated in these surveys (Stevens *et al.*, 1991). The first approach is thus likely to

inflate the value of the resource and the second approach is likely to deflate the value.

Further, the choice of model to 'fit' the data also can influence the outcome. Bowker and Stoll (1988) used three models on their data. All models had similar goodness of fit but produced disparate estimates of value. Thus the analysis that is performed on the data has the potential to influence the resultant attribution of value to the resource.

Contingent Valuation also deals with large, discrete quantity changes. For instance, the states of the world are described as either having a resource R or lacking it. Marginal quantity changes are not captured. For instance a reduction in R of 10 percent or an increase in R of 10 percent. Yet recovery programmes typically produce marginal quantity changes. That is, the probability a species will persist is raised (hopefully) as a consequence of a recovery programme. Further, given that a conservation agency has many candidates for recovery programmes the valuation agency has to make decisions on the margin. *That is, they have to decide at which point they will cease supporting one species and start supporting another.* As contingent valuation does not produce marginal values its usefulness for policy is restricted.

A rather contentious problem is embedding (Blamey and Common, 1992; Hanemann, 1994; Diamond and Hausman, 1994). This phenomena is often observed in CVM studies. Embedding occurs when a subject is willing to pay the same amount for a resource R, irrespective of the quantity. Thus, a subject might value one species at the same level as two or more combined. The additional species in effect have a zero value. In Blamey and Common (1992) geographical embedding was found. At a 95 percent level of significance the respondents valued 10 percent, 50 percent and 100 percent of the area of National Estate forests equally.

The suspicion that the sum of all valuations for endangered species would be greater than if species conservation was valued as a whole has also been expressed (Jakobsson and Dragun, 1989).

Smith (1992) attributes some embedding effects to the survey design and analysis. That is, the reason embedding has been observed is a product of not specifying the good to be valued properly. Therefore the subject is unsure of what it is they are valuing. If embedding is likely where goods can not be described accurately then the usefulness of CVM for species is limited.

Species are likely to be goods for which such description is unlikely. A species is a "good" with multiple properties. Some of these properties are unlikely to be captured now. Rather, they will provide value in the future. These properties include ecological role (now and in the future), incidence of useful genes, the presence of substitutes (for instance, the number of genes that are shared with relatives), aesthetic and scientific parameters of interest, possible harvesting options, educational options, ecosystem monitoring options (is the species an indicator of ecosystem status?) etc.

Many of these properties are surrounded by uncertainty at very low levels of probability. For instance, off-the-cuff estimates of the probability a species will have genes useful for biotechnology range from 5,000-1 to 10,000-1 against (Moyle, 1994). Further, the behaviour of subjects in surveys involving probabilities of low magnitude are conspicuous with paradoxes (Machina, 1987, 1989). That is, preference reversal can readily be produced. It is scarcely credible that subjects in a survey will be able to assess the low probabilities associated with the properties listed above accurately (Ready and Bishop, 1991).

Of the particular studies above the strongest indication of an embedding effect is the Stevens *et al.* (1991a) study. Subjects were divided into three equal sized groups. The first group was asked to value the New England bald eagle. The second group was asked to value the wild turkey. The third group was asked to value the wild turkey and bald eagle together. The outcome was that the reported values of group three were less than that for group one and two. That is, the combined value of the two species is less than the value for either species considered singly.

This is an unusual example of embedding as the marginal species appear to have acquired negative values. This appears to be intuitively unlikely. Afterall, the two groups valuing "single species" attributed a positive value to the species concerned. A possible interpretation is that CVM produces a choice problem where the assumptions of consumer behaviour are unlikely to hold.

The CVM technique assumes that a subject's behaviour is going to be consistent with the neoclassical elements of consumer choice. It is therefore prescribing a utility function for which there are (at least) two types of goods. The first is the subject's normal consumption bundle of goods. The second is the bundle of endangered species. The subject is seen as allocating a budget among the bundles. This allocation is aimed at maximising something that the subject finds good. It is conventional to term this utility.

To maximise this "good" the subject must have a complete ordering of preferences between the two bundles and within each bundle (Varian, 1984). As a result of the completeness of preferences, the subject is able to reduce all states of the world into a preference ordering. As a consequence, the subject is able to trade-off between consumption bundles and within consumption bundles.

This is illustrated by the following example. In the normal consumption bundle X there are only two goods x_1 and x_2. In the species bundle S there are two species s_1 and s_2. Completeness occurs within each bundle. Thus, a feasible preference ordering would be $x_1 \succ x_2$ and $s_1 \succ s_2$. The neoclassical theory of consumer choice also requires that completeness occurs between the bundles. For instance, the following ordering may be feasible: $x_1 \succ s_1 \succ x_2 \succ s_2$.

If such an ordering exists then it should be possible for a subject to provide answers that reflect their preferences. However, it can be disputed that such an ordering does exist. That is, the subject is unwilling to make such an ordering **or** is unable to reduce the welfare of each state-of-the-world into a comparable measure. One symptom of the inability to reduce outcomes to a comparable measure is the violation of transitivity.

The CVM technique when applied to endangered species has properties which reduce the likelihood of completeness. For instance, suppose the subject regards species conservation as a moral issue. Evidence of this comes from Stevens *et al.* (1991a) and Moyle (1992). For instance in Stevens *et al.* (1991a) 44 percent of the respondents agreed with the statement that "preservation of wildlife should not be determined by how much money can be spent" and 67 percent agreed with the statement that "As much wildlife as possible should be preserved no matter what the cost". This demonstrates that many subjects (especially those concerned with conservation) do not see the choice in expedient terms. This is supported by the results of public opinion surveys (Smith, 1990).

In consequence, the subject regards species conservation as something for which trade-offs are not possible. This may violate the standard model of

consumer choice in a number of ways. First, preferences over the range of quantities in X and S may be lexicographic (Chipman, 1971; Varian, 1984). That is, the subject feels that a certain level of conservation must be achieved before all other preferences are fulfilled. This would make the subject's utility function **discontinuous**. Second, the requirement to fulfil a moral principle may defy reduction into a single welfare term. The subject is being asked to 'compare apples and oranges' (Elster, 1979). Thus, the incidence of transitivity violations may increase.

Compounding the above problems is the fact that a subject may not know the species in S sufficiently well to have a prior preference ordering. Familiarity with X is produced in the everyday business of market transactions. However, subjects may be very ignorant of S and rely much more on the information provided in the survey (Samples *et al.*, 1986). In this sense it can be argued that CVM produces values for things not previously valued. Not knowing S the subject may be applying *ad hoc* rules rather than consistent preference orderings. The subject can not make a complete ordering across X and S because they do not have an ordering within S.

Suppose the subject knows that saving species is a good. They wish to maximise this good subject to other demands on their budget. The characteristics of their utility function are that preferences between X and S are complete **and** within X preferences are complete. It is just within S that orderings are not complete because of ignorance (Harsanyi, 1988). Consider a state of almost complete ignorance where only one species is a candidate for a recovery programme. Here, this species would attract almost the entire budget the subject has allocated for S. They are in fact solving a partial equilibrium problem. That is, if there is only one species at risk from extinction what should they contribute?

The subject, because of their ignorance, discounts the claims of other endangered species.

Where the subject is informed that S is larger than one (that is, there are more species that are deserving of funds) their allocation changes. A preference ordering across one species is easy- it attracts all the budget for S. Where many species are involved the subject has to be more prudent. The number of species in S is larger than they thought and each allocation has an opportunity cost in terms of reduced funds to other species. The subject is then attempting to solve a general equilibrium problem. If the subject is risk averse they may establish a contingency allocation within S to meet the claims of future endangered species. This would reinforce the general equilibrium solution and support an embedding effect. The absence of a complete ordering across S would also increase the likelihood of transitivity violations. Reinforcing this ignorance is the fact that the changes the CVM method proposes are large and discreet. The neoclassical model of consumer choice typically deals with small marginal changes.

The problem of species conservation is thus likely to produce the effects of embedding and intransitivity. It is characterised by moral elements and a high degree of ignorance. This inhibits the subject from making a complete ordering among all relevant goods. The fact that the subject does make choices (ethics after all has not banished the problem of scarcity) does not indicate that the subject has a complete ordering. All choice demonstrates is that the subject faces scarcity. The CVM method is poorly designed to elicit prudent and informed valuations under these circumstances.

The element of ethics finds expression in the CVM technique at other levels. Ethics is likely to impact on the number of protest bids. A

proportion of protest bids result from ethical reasons (Stevens *et al.*, 1991a). In this study 25 percent of the protest bids recorded a zero bid because they claimed "that wildlife values should not be measured in dollar terms". The use of dollar values may be unethical in the mind of the subject. This would produce feelings of conflict with the survey. The consequence of this is that the incidence of protests, avoidance and intransitive bids would increase.

A further source of conflict is the implicit system of property rights within the survey. A WTP or WTA question presupposes that some group has acquired property rights over the species concerned (Vadnjal 1992, 1993). If the subject feels that this allocation is unjust or undeserved than the presence of protest bids will again increase.

Further, the CVM technique is unable to filter out misinformed preferences (Harsanyi, 1988). For instance, a subject may prefer to save species A over species B based on certain assumptions. These assumptions happen to be in error. The subject may be assuming that the rate of decline for species B is very slow. Thus, they have the ability to save species A and later, species B. What they would choose to do if their preferences were informed may be very different.

This could be a problem with the application of CVM techniques to the valuation of endangered species. Suppose subjects wish to save the maximum number of species. To do this they choose a strategy which concentrates effort on the most-at-risk and expensive candidates (a minimising loss strategy). This strategy is sub-optimal with respect to their goal. Thus, the values the CVM techniques would report would be in conflict with their preferred goal.

There are many features of the CVM technique which make it inapplicable to valuing endangered species. It is unable to deal with large numbers of species. Risk and time are unable to be presented to the subject in a realistic way. The subject's own reported preferences may be biased or misinformed. The changes it relates to are large and not marginal. This reduces applicability to policy. Thus, the CVM method is rejected as a suitable tool for revealing relative value.

4.2 Subjective Approaches: The Safe-Minimum-Standard

An alternative to CVM is the Safe-Minimum-Standard approach (Bishop, 1978). Along with CVM, the approach assumes that the society is only interested in preserving species on the grounds that the species will contribute to human welfare. The SMS is the "minimum reserve needed to avoid potentially grossly adverse consequences to human welfare" (Krutilla, 1967). The decision rule is to preserve all species subject to the social cost not being excessive (Bishop, 1978).

The SMS approach, argues that it is not possible to determine the value of a given species. If a species is preserved, it may become very valuable in the future. This value is likely to be a result of pharmaceutical or agricultural applications. However, this value may be impossible to measure. For instance, it may not be possible to predict, at a credible level, what the odds are that the species concerned, will become a valued resource in the future (Bishop, 1978). Alternatively, even if the species becomes a valuable resource in the future, it may be impossible to forecast the value, future generations will place on it (Ready and Bishop, 1991).

Given predicting the benefits of preservation will be impossible, the SMS approach perceives that preservation is analogous to an insurance policy. By preserving species, society insures itself against potentially large losses in the future. This insurance view of preservation, can be put into game theoretic terms (Ready and Bishop, 1991).

Following Ready and Bishop (1991), assume that society has two, mutually exclusive strategies to choose from. They may choose a development strategy (DEV) that will result in the extinction of a species. Alternatively, they may choose an SMS approach which will prevent extinction at the cost of development. It will be assumed that the development benefits (Bd) will be positive net of any costs (including the losses from current benefits provided by the species). The SMS carries no further costs other than the forgone development (Ready and Bishop, 1991).

If extinction occurs, there is the potential for large losses to society. The species concerned could provide a cure for some new deadly disease, like AIDS. Extinction will thus produce a large loss to society given that the cure will no longer exist. This loss will be measured as L, and is assumed to be greater than Bd. If an SMS approach is followed, a cure will be found, preventing the losses L from occurring (Ready and Bishop, 1991).

Nature generates one of two exclusive strategies also, using some unknown decision rule. The first is to generate an outbreak of some new disease (D) and the second is to generate no outbreak (N).

These strategies can be placed into a game-theoretic matrix (Table 4.1). The payoffs that are of interest are the losses to society only. A baseline loss of zero, is defined as a situation of no disease and no development. Clearly, this baseline position does include the cost of forgone

development. A negative loss below, corresponds to a benefit to society (Ready and Bishop, 1991).

Table 4.1 Loss Matrix NATURE

Strategy	D	N	Max Loss
SMS	0	0	0
DEV	L-Bd	-Bd	L-Bd

If an SMS approach is chosen, and the disease breaks out, then the loss is the base of zero. A cure for the disease is effected given that the species is extant. This eliminates the losses that otherwise would occur. Likewise, if the disease does not break out, then the loss is still the base of zero (Ready and Bishop, 1991).

If development is chosen, then the loss is equal to L offset by the benefits of preservation. However, as L is much greater than Bd, the cost to society is very high. If the disease does not break out, then the "loss" is -Bd, ie. a net gain to society of Bd (Ready and Bishop, 1991).

To minimise the maximum possible loss to society, it would be prudent to select the SMS approach. The maximum loss possible from a SMS approach is 0. The maximum loss possible under the DEV strategy is L-Bd (Bishop, 1978; Ready and Bishop, 1991).

However, this result is not robust. In Ready and Bishop (1991), changing some of the assumptions altered the outcome of the game above. While the same decision rule was applied (minimise the maximum possible loss), treating the situation as a lottery, favours development. In fact, the insurance game (Table 4.1) could also favour development, if a third outcome was included. In this case, if the species does not produce a cure, the SMS would generate the maximum loss of L+Bd.

The SMS was conceived as a decision rule to be applied where the benefits of preservation could not be calculated. The rule to favour preservation, does not have a good theoretical base (Ready and Bishop, 1991). Further, the rule favouring preservation has a high conservative bias (Luce and Raiffa, 1957; Bishop, 1978). If the probability any given species will provide large benefits in the future is extremely low, favouring preservation will not pass other welfare tests, such as expected utility.

It is unlikely that any given species will generate large benefits. For instance, by 1991 Xenova had screened over 23,000 species of plants and fungi for pharmaceutical uses with no positive results (Moyle, 1994). The odds that any species will produce new biotechnology resources, is estimated to be in the region of thousands-to-one against (Moyle, 1994).

From the perspective of the thesis, the SMS approach is of limited use. It generates only one criterion, to determine which species should be favoured with recovery programmes. As it assumes that the benefits are both uncertain and unknown, cost becomes the only feasible criterion. Thus, to minimise the maximum loss to society, the maximum number of species should be saved. This improves the odds that within the set of saved species, some will produce large future benefits.

The SMS could not recommend preserving a species, if a human engineered solution was likely to be cheaper. Technology is diminishing the need for preserving large numbers of species (Huber, 1992). Indeed, the search costs involved with screening species for pharmaceutical applications are considerable (Moyle, 1994). There is little evidence that pharmaceutical companies find screening species as a cost-effective way to generate new products (Moyle, 1994).

While costs should be an important factor to select candidates for recovery work, it can not be the only criterion. The value of a species, may be determined by factors other than cost. For instance, aesthetic factors presumably influence the values of subjects. In short, subjects place value on species for a number of reasons. The use of cost only, is a rather heroic, simplifying assumption.

4.3 Objective Approaches: the SPRS

Background to Conservation Programs in New Zealand.

Recovery programmes in New Zealand have only been instituted in the last one hundred years. Low population densities and tapu restrictions by the Maori did allow coexistence with many species prior to the arrival of Europeans (Halliday, 1978). However both Maori and European settlement has been associated with extinctions. For instance, in the Polynesian era thirty-two species of birds became extinct. During the last two hundred years another nine have become extinct (Gill and Martinson, 1991).

Concern for threatened and endangered species has been a feature of the last one hundred years (Bellamy *et al.*, 1990). There have been two phases to this concern. The first phase found expression in passive policies. This was the creation of reserves and the granting of legal protection to the species. For example, throughout the 1890s, sanctuary islands were established. Resolution Island was established in 1891, Little Barrier in 1895 and Kapiti in 1897 (Galbreath, 1993). Many native birds were placed on a protected list also during this period. These included the huia, white heron, crested grebe, bellbird, tui,

kokako, kiwi, saddleback and stitchbird. Once these policies were in place, the species was left to its own devices to recover. This was not always successful. For instance, legal protection for the huia did not prevent its demise (Bellamy *et al.*, 1990).

The second phase has featured more active human interventions. Typical of these has been pest eradications, captive breeding, supplementary feeding stations and surrogate breeding (Bellamy *et al.*, 1990). This second phase has been a largely post-World War Two phenomenon.

In the face of growing demand for preservation action (Bellamy *et al.* 1990) conservation policy was rationalised into one Government Department. This combined elements of the Department of Lands and Survey, the Wildlife Service from the Department of Internal Affairs and the Forestry Service.

The Species Priority Ranking System

One of the tasks of the Department of Conservation is to actively engage in recovery programmes for endangered species. In view of the large number of species who are possible candidates and a limited budget, the Department is trying to optimise its work. The was recently manifested in the Species Priority Ranking System (Molloy and Davis, 1992). This system (normally abbreviated to the SPRS) is inadequate for role it is intended.

The SPRS is intended to identify those species which are in need of **consideration** of a recovery programme. It is not intended to justify any particular programme; thus leaves out criteria which would result in the rejection or acceptance of any specific programme (eg. cost).

The system classes all possible candidates according to five factors. Each factor has a number of criteria. For instance, the Cultural factor has two criteria and the Population Status factor has 6 criteria. Each criterion is ranked, ordinally between 1 and 4 or 5. The higher the value of the rank, the more valuable that species is within that particular criterion. Thus a species that scored 3 on taxonomic distinctiveness is more 'taxonomically distinct' than a species that scored 1. Given the ordinal nature of these ranks it is not possible to make cardinal comparisons. That is, in the case above, the species that scored 3 is not three times more valuable (from the perspective of taxonomic distinctiveness) than the species with rank 1.

It is this ordinal nature of the ranks which contributes to the first methodological flaw. In essence, without a system of indexing or weighting the ranks and criteria, it is not possible to sum the ranks to a meaningful total. The act of summing, without weights, implies that the ranks are cardinal. Summation is therefore mathematical nonsense and the results completely meaningless. However, summing the ordinal ranks this is precisely what the SPRS does. To compare species, their ranks are summed.

To some extent it is realised that summing each species does not yield comparable totals. As an expedient, the SPRS only classifies species according to three broad bands. That is, each score is compared to a set of three ranges. If the total falls within a particular range, that species falls into that category. This does not address the root cause of the problem. It also greatly reduces the applicability of the system to policy. This is because the system produces only three outcomes. Each species belongs to either Category A (requiring immediate recovery work), Category B (requiring recovery work in the medium term) or Category C

(requiring recovery work in the long term). Determining the needs of each species within the same category is not possible under this system.

The choice of criteria is only justified as being agreeable to those involved in the design of this system. However, the quality of general agreeability is not sufficient to justify the criteria of the SPRS. The particular choices of criteria are actually contentious. The points of contention come from the implicit weighting of criterion, the interdependence of certain criteria and the aim of the SPRS.

In the SPRS there is only one criterion for taxonomic distinctiveness. Taxonomic distinctiveness is also intended to be a proxy for genetic diversity. As a proxy measure for genetic diversity it is probably acceptable. That is, there is a positive correlation between the taxonomic and genetic distinctiveness of species. However it is also possible for this correlation to be negative. That is, within a genus there is a high level of genetic diversity but between related genera there is a low level of diversity (Solow *et al.*, 1993). For instance, consider the following case. Genus A and B are closely related. Genus B has one species, species 3. Genus A has two species, species 1 & 2. According to the SPRS species 3 is preferred because it belongs to a monotypic genera. However, species 1 has adapted in isolation (perhaps for an underground habitat) and now has little genetic overlap with species 2. The single species that will contribute most to genetic diversity is thus species 1. In this case the SPRS will favour the wrong species. A discussion of the inability of a simple, linear scoring mechanism to identify taxonomically distinct species is found in Solow *et al.* (1993).

The actual choice of criteria is also not free from contention. The choice is justified in terms of general agreeability among those responsible for the report (Molloy and Davis, 1992). There is a degree of ambiguity

about the outcome the SPRS is trying to achieve. This ambiguity is evident in the choice of criteria.

Consider why a particular species should have priority over another with respect to evaluating a recovery programme. There are two possible explanations. The first, is that the species that are given priority are more likely to become extinct than the others. That is, if their programmes are delayed then there is a greater chance of irretrievable loss. The second is that the species that are selected are somehow more valuable in conservation terms. That is, they have some biological qualities which if lost would create a greater loss than if a lower priority species was to become extinct. Despite the suggestion of one of the authors (Davis *et al.*, 1992) these reasons do not overlap. That is, the SPRS is not designed to select species on either of these grounds.

With regard to the first case there are a number of criteria which have no relation to the time it will take for a species to become extinct. These include the 'taxonomic distinctiveness' criterion and the 'cultural value' criteria. The silver fern *Cyathea dealbata* for example, has significant cultural value but it is not endangered. The number of factors devoted to conservation qualities also make it unlikely that the SPRS will identify species with high conservation values. For instance, there is no ecological criterion, and genetic distinctiveness enters the SPRS only by one proxy measure. By contrast, the status of the population (size, distribution etc.) warrants 6 criteria.

The result of this is that the SPRS can be used to justify particular programmes while simultaneously **not** justifying a programme. Any species that occurs in a high priority category can have a recovery programme justified on the basis of the SPRS. However, the absence of certain criteria mean that DoC is free to supplant a high priority species

by appealing to other factors. As a discriminatory guide to decisions, the SPRS is not well designed.

The SPRS prefers species that are rare at the expense of other qualities. Unfortunately, rarity is not always associated with risk of extinction (Drury, 1974). The preference for rarity is evident in the population status factor. The criteria within this factor are correlated to each other. For instance, a species that has only one population is also likely to have low population numbers and a restricted distribution. This weights rareness much more than any other factor. For example, half the species in the high priority category have only one population (Davis *et al.*, 1992).

As a consequence, the SPRS has a selection of independent and correlated criteria. The presence of correlated criteria imply that some aspects of a species are going to be double counted. This again reduces the applicability of the SPRS unless the subject shares the same preferences as the authors.

This problem with the choice of criteria, illustrates the objective-subjective conflict of this system. The system is intended to be objective. That is, if a species has a certain set of properties it will be more valuable. Thus, it does not matter about the perceptions of subjects. An endangered weta is treated on the same basis as an endangered parrot. Only the objective properties of species may be used to discriminate between them.

However, the problem is how to determine these properties and why they should be considered valid. The authors note that the selection of criteria was necessarily a subjective process (Molloy and Davis, 1992). This

reduces the objective claim of the SPRS. The SPRS is not proven to be superior to other weighting systems.

Given the subjective basis for criteria selection it is impossible to design a non-contentious objective system. People will have many reasons for preferring some species over others. The **number** of these reasons should prevent their capture by any single system. At some point, these reasons are going to become trivial. For instance, the subject who prefers green feathers to red feathers on birds. It is **not** practically possible to design a system exhaustive enough to contain all of these motives.

The intention of the SPRS is **not** to determine a budget allocations among the various candidates for recovery programmes. It is intended to guide the budget allocation nonetheless. However, it is inadequate for this task; ie. setting priorities for conservation action. The results the system generates are non-comparable because of their ordinal nature. The grouping of results into broad categories is arbitrary and reduces the ability to discriminate between species. Not enough criteria are included to reflect biological concerns regarding high priority species. Finally, there is no recognition of any diminishing returns. That is, if twenty species of kauri snails are saved then the twenty-first is given the same consideration as the first bird species. The system does not weight the achievement of a balanced portfolio of species.

CHAPTER FIVE
A GAME THEORETIC APPROACH

5.1 Introduction

The subjective approaches discussed in the previous chapter are not well-suited as a decision tool to allocate funds among species. The merit of CVM and SMS is that they demonstrate two points: an endangered species has value (even in the absence of a real market) and this value is constrained by costs.

The shortcomings however count against these techniques. The SMS does not identify which properties of a species has the most value. It is not an approach to allocate funds where the costs are *relatively* high. The CVM finds its application in species that are reasonably well known. It does not offer as much guidance for the vast majority of species outside of public knowledge. Further, applications of CVM have not identified the properties of a species that contribute towards its valuation. CVM would also be very costly to implement on the three-hundred odd New Zealand endangered species.

A game theoretic approach addresses a number of these shortcomings. The multiple choices (with selecting candidates for recovery work) that have to be made under circumstances of risk and uncertainty fit well into a game format. The budget allocation problem can be constructed in game theory terms.

Take the case where there is only one person responsible for allocating the budget. A host of environmental and population factors, *outside the control of the manager*, may improve or diminish the success rate of this budget (Maguire, 1986). These factors can be represented by a strategy chosen by "nature".

Further, assume that the player only has two species, species A & B. They could choose to put all the budget into species A, or all of the budget into species B, or allocate part of the budget to A and the rest to B.

This situation can be put into game theoretic terms:

Figure 5.1

		Nature	
		F	U
Player	All A	X	L
	Mixed A & B	Y	M
	All B	Z	N

In the above matrix, the strategies chosen by nature are either Favourable (F) or Unfavourable (U) to recovery work. The player has three strategies; everything into A (All A), everything into B (All B) or part into A and part into B (Mixed A & B). The payoffs are those associated with the player. In this game we are not interested in the payoff to nature.

Games are either strategic or extended form. The game matrix above is in strategic form. However, an extended form game would have some advantages for players. An extended form game would allow a subject to monitor the progress of their bids. This allows feedback to a player which would be lacking in a survey method of valuation.

A game is also characterised by a "solution". A solution can be a single or mixed strategy. The solution gives the player the highest payoff taking into account the strategies chosen by opponents. The solution is described as a Nash equilibrium. A Nash equilibrium is also a prediction of what strategy a subject will choose in the environment they find themselves. Depending on the complexity of the game a Nash equilibrium may not always be immediately recognised by a player.

In the game above, there are at least two Nash equilibria. If Y > X > Z then the Nash equilibrium if nature choose strategy (F) would be (Mixed A & B). If L > M > N and nature choose strategy (U) then the Nash equilibrium would be (All A). The actual strategy chosen by a player would then be determined by their attitude towards risk. For instance, a risk-averse player may wish to choose to maximise the minimum payoff. They would choose (All A), which gives the highest payoff if conditions are unfavourable. An alternative would be an expected utility rule. This would require the player to attribute a probability distribution across F and U. The more likely F was or the higher the payoff Y, the more likely a player would choose (Mixed A & B). In this case, the expected payoff from choosing (Mixed A & B) would be rising.

Rational players will try to find Nash equilibria. This is a consequence of their rationality. Faced with a choice between exclusive states of the world a player should chose that state that has the highest value. It would be perverse to settle on a state of the world with a value lower than that which could be achieved.

The process of playing an extended form game permits the introduction of random elements. The outcomes of bids can be varied by the use of a probability distribution. Thus, outcomes are transformed from being deterministic towards stochastic. A subject's attitude towards risk and uncertainty will in part determine their bids. For the observer, bids made under these circumstances convey information on subject's valuation of risk.

5.2 Simulation Design

The form of the game used was a simulation. This had a number of advantages. The principal advantage was that simulations are extended

form games. Thus, if players started with misinformed instrumental preferences (Harsanyi, 1988) then this would be revealed. [1] The second advantage of a simulation is that player behaviour could be subject to more control. That is, money payments and other constraints could be introduced. Such control of the setting would limit the impact of "unobserved variables" on player strategies.

The objective of the simulation was to create a game[2] which permitted a number of Nash equilibria corresponding to player's utility functions. That is, from the choice available, at least one strategy in the set of Nash equilibria would match the utility function of the player. The bids players offer to each endangered species should be consistent with their utility function. Another aim was to prevent distortions to player's bids by constructing a neutral payoff table.

The simulation was constructed to represent the key elements of the problem facing New Zealand. The proportion of species (per taxonomic group) in the set of actual endangered species is approximately the same as that in the set of game endangered species. For instance, in New Zealand over half the endangered species are drawn from the plants and invertebrates. Likewise, over half the species in the game are either plants or animals.

The game was set up as a board game rather than a computer simulation. It is conceived as a one player, extended form game. The game is principally a budget allocation game with a resemblance to *Snakes and Ladders*. The progress (or lack thereof) of a species is represented as a movement of a counter on the playing board.

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[1] Misinformed preferences are discussed below in chapter 6.
[2] The rules for this simulation are enclosed as *Appendix 1* in this thesis.

The rationale for avoiding a computer simulation was to avoid confusion generated by population numbers. That is, in the board game risk of extinction is not represented by a population number but by the physical distance from the extinction cell. Population numbers do not provide a good indicator of extinction risk over a *disparate range of organisms* (Drury, 1974). The population numbers and densities of plants and invertebrates is often different to birds, and likewise reptiles. Position in a trophic level also partly determines population densities (Drury, 1974). These factors account for relative rarity or abundance but do not reflect starting risk of extinction.

In essence, population densities are not comparable across different groups of organisms. Hence, it is not possible to tell if a low population in one group is at greater risk of extinction than a high population from another group. The *rate of decline* will be more important at determining risk then the initial population density.

Further, **all** endangered species are at risk from catastrophes. These events do not distinguish between population densities to an appreciable degree. For instance, the heath hen became extinct despite a recovery programme that appeared successful. The population density of this species reached about 2000 individuals before catastrophes wiped it out (Halliday, 1978). Droughts or cyclones in New Zealand can have catastrophic effects also. Where an area is struck by a catastrophe the population density of the endangered species will be largely irrelevant.

The set of endangered species was divided into three categories. These categories corresponded to the expected time to extinction. For instance, all species that start facing imminent extinction will be extinct at the end of the game in the absence of interventions. The three categories are:

- Category A, Facing Imminent Extinction;
- Category B, Highly Endangered;

- Category C, Endangered.

These Categories also determined the technical cost of recovery.

The aim of the game was to discover the relative importance of class (what taxonomic group the species belonged to), genetic and ecological properties, risk, time and cost. Changes to the overall budget of players was also considered. No counter was identifiable as a specific organism. This was to reduce the impact of bias produced by mega-fauna. Mega-fauna are high profile species which evoke strong responses among subjects (such as the elephant in East Africa). Given that most endangered species are relatively unknown a bias could result. That is, the subject identifies all species in a group with a particularly well known representative.

The simulation allowed for the control of other factors. While including these factors would make the simulation more realistic it would be more difficult for subjects to compare outcomes. The simplifying assumptions are as follows:

1. Recovery costs are correlated with the expected time to extinction. That is, a species that is expected to persist for a long period of time will have a lower technical cost of recovery than a species that is facing imminent extinction.

2. Recovery costs do not vary for any other reason. Species that face the same time to extinction face the same technical recovery costs. A plant species in this case faces no difference in technical cost to a bird species.

   This assumption preserves neutrality in the scoring system adopted for the game. In the game, the subject's utility function is influenced by the score they obtain. Deliberate adjustment of recovery costs on

the basis of class would produce substitution effects towards "cheaper" species. This would make determining the impact of class more difficult. It would also increase the number of variables a subject must consider. This would make the subsequent reduction of variables to a single utility measure more difficult.

As no great advantage was judged to be obtainable from relaxing this assumption it was left in place. The subject is observed to make a set of marginal bidding decisions. These bids would reflect the marginal utility from preferring one group of organisms to another. Marginal utility theory would allow for subsequent adjustment of price (that is, technical cost).

3. The conservation agency has perfect mobility of resources. The budget may be freely allocated without regard to technical limits. This means for instance, that resources may be freely transferred away from a bird species towards a plant.

4. The model is deterministic to a large degree. It is not possible to experience catastrophes of a random nature in the game. The progress of a species is determined by the random "move" by Nature and the budget allocated to the species.

The "move" by nature is intended to capture the effects of high probability, low impact events. These include shifts in the rate of predation, competition, disease and weather, as well as random shifts in the birth and death rates. It does not incorporate the effect of low probability, high impact events. For instance, the effect of new virulent diseases or meteorological events or new predators.

The decision to omit these effects is based on the perception that in

the majority of games, these effects would be irrelevant. That is, these events are sufficiently low probability that their frequency is unlikely to be experienced in the time frame (about ten years) presented in the game. Its chief effect would be to drive a wedge between bids and outcomes. Thus, the relationship between a bid and a particular outcome would be more difficult to determine. In this sense, a player who adopts a bidding strategy that may otherwise lead to a good score, would be penalised.

In consequence, players are not being penalised for completing the game with relatively few *saved* species. It is the *endangered* species, with few populations, that are at then most at risk from these catastrophes. However, this presumes that saving many species yields more utility to the subject than saving few. This can not be imposed without destroying the neutrality of the scoring system. While players may recognise that saving species is "good" they may not wish to bear the opportunity cost of doing so. To obtain a high rate of recovery within a budget constraint implies an offsetting rise in extinctions. Thus, this assumption is made to ensure that a subject's strategy is not distorted by the scoring system.

5. The probability of success is not affected by progress in previous rounds. However, progress in earlier rounds may modify the cost of a recovery programme if not the probability.

Progress shifts a counter into a region of lower technical costs. In consequence, the *expected cost* of shifting a counter a constant number of cells falls. This payback is indistinguishable from a rise in the probability of success.

6. Each threatened species is linked to others in an *ecosystem*. Each

species experiences the same environmental factors as others in its ecosystem. However, success with one species does not overlap with success for another. That is, the benefits of one recovery programme have no benefits for others in the same ecosystem.

It was not seen as necessary to link recovery programmes of species. While this detail would be more realistic, it is indistinguishable from a fall in the relative price of saving a species. During the game the player experiences changes in relative prices. Their behaviour in response to this can be used to model the effects of such linkages. That is, a recovery programme on species X that has a beneficial effect on species Y is indistinguishable from a reduction in the cost of a recovery programme for species Y.

Population Model

The underlying population model is based on a modified Lotka-Volterra system. This system involves two species. This is the endangered species and another species which predates or competes with it. It is not necessary to specify whether this second species is a competitor or predator (although this would be necessary in the original system). It is only necessary that a rise in the population density of this second species lowers the population growth of the endangered species. Both a competitor or predator do this, either by direct consumption of the endangered species or through a reduction in the endangered species' resource availability.

There are two modifications to the basic Lotka-Volterra model. The first is that the predator population is *not* dependent on the density of the endangered species. This follows from the existence of substitute species a predator may also feed on in the absence of the endangered species. The second modification is that the predator population is influenced by the human effort on control.

More formally the two population equations are as follows:

$$\frac{dN}{dt} = r_1 N - aP \qquad (1)$$

$$\frac{dP}{dt} = r_2 P - \beta H \qquad (2)$$

In this system, N is the population density of the endangered species and P is the population density of the predator. Human effort is represented by H and is measured in dollars. In equation 1, r_1 is the rate of increase of the endangered species in the absence of predation (or competition). In equation 2, r_2 is the rate of increase of the predator in the absence of human intervention. For both populations, their rate of increase over time will be exponential if the growth rate approaches r_1 or r_2. That is, if the rate of increase is r_1 or r_2, the solution for population density at time t is:

$$x = x_0 e^{rt}$$

Here x is the population density at time t and x_0 is the density at time t=0.

Resource Constraints

The exponential nature of population increase here is not moderated by resource constraints. While this could be incorporated it was omitted for simplicity. At any rate, the Department of Conservation's research implicates predation and competition as dominant sources of population decline (Davis et al., 1992). Habitat loss is generally not a major cause of population decline. The implication is that an endangered species, could over a certain range of population density, increase exponentially to fill its habitat. It can also be assumed that the population density of a predator or competitor species could also rise much higher (to approximate an exponential increase).

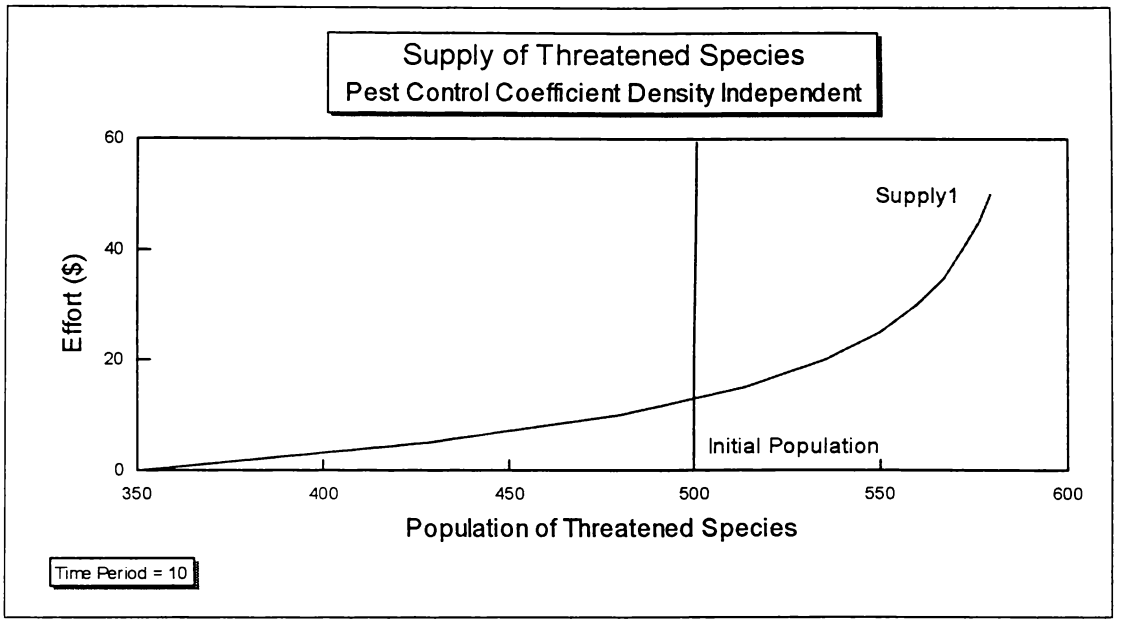


Figure 5.2: Population at t=10

The Lotka-Volterra model outlined above (equations 1 and 2) can be modelled to illustrate the rising marginal cost of increasing the population of a threatened species. This is shown above (**Fig. 5.2**). In this particular case, the initial population of the threatened species was five-hundred. As the resources committed to control the predator rose, it becomes progressively more difficult to generate a population increase in the threatened population. For instance, in this case, raising effort from 0 to $20 will generate a population rise of just under two-hundred over ten time periods. A further $20 increase will generate less than fifty new individuals for the threatened population.

This conclusion, the existence of diminishing returns, is not altered if the pest control coefficient is density dependent. The graphical impact is to make the supply schedule steeper as effort rises, in contrast to the density-independent case. In the density dependent case the marginal cost of killing the next pest is rising. This makes the resultant rise in population numbers of the endangered species more difficult to achieve.

Lotka-Volterra Model

In the two equations above there are factors which reduce the rate of population growth. In equation 1, this is the rate at which a predator (or competitor) reduces the population. This is a function of the population density of the predator and the coefficient α which describes the rate at which predation occurs. It is possible for predation to be sufficiently severe that the population growth of the endangered species is negative. The expression for predation states that losses to an endangered species will increase as the population density of the predator increases.

In equation 2, human control is able to reduce the density of predators. This is a function of the effort expended on control and the coefficient β which describes the rate at which predators are killed (that is, the kill rate per dollar). It is possible for human control to be sufficiently high that the population growth of the predator population is negative. An implication of a decline in the population density of the predator would be a rise in the population density of the endangered species. This system is a closed system also. There is no migration into the area by new members of the predator species. This is a valid assumption when island reserves are considered.

For any point in time, H may be freely varied. As H increases in this one instance, then the population density of the predator falls. This has the effect of increasing the population density of the endangered species. As P gets very small, dN/dt approaches r_i. Therefore, extra effort spent by humans has a diminishing effect on the population growth rate of the endangered species (**Fig. 5.2**).

The consequence of this model was the construction of an outcome table also subject to diminishing returns. That is, to achieve progress of one cell, the **expected** amount to be spent, kept increasing. For instance, for a species in

Category B; to shift the expected movement from a decline of one cell to no change in position would be an extra $3. To shift the expected movement from progress of two cells to progress of three cells would be an extra $7. The additional movement is subject to rising marginal cost.

In essence this captures the idea of a reproductive limit to a species in any one point in time. There are only so many juveniles that can be produced. Endangered species tend not to be able to freely vary their output upwards.

Scoring System

The scoring system for the simulation depends on the net movement of species on the players board. Each cell is ranked in a linear manner. The extinction cell has the highest score and the recovered cell has the lowest. The game has an initial score and a final score. The initial score is the sum of the cardinal number of species in each cell multiplied by the cell number. This is summed across all cells. The final score is calculated the same way. If the final score is less than the initial there is a net movement of species towards recovery. That is, the losses a player has accepted has been more than offset by the gains.

To encourage players to fully consider the opportunity costs of their strategies monetary payoffs were also introduced. Players were ranked in order of performance (that is, what their final score was). Depending on their position, a monetary payoff was allocated. This increases the concern a subject has towards their strategy. Players faced less incentive to treat the game as strictly hypothetical. A player is affected by their choices. Poor choices reduces their expected monetary payoff. Subjects face **real** changes to their welfare as a result of their bidding strategy. Hence they face increased incentives to behave consistently with their preferences.

The linear scoring system does not explicitly weight some goals more than

others. A players score will progress by the same amount whether they achieve progress with a bird instead of a reptile or with a Category A species rather than a Category C. At the margin there are bonus and penalty points however. An additional point is penalised if a species becomes extinct and a bonus point awarded if a species is recovered.

To encourage the development of a preference ordering players were allowed to differentiate some species. Four species could be classed as high priority species. The points awarded for achieving progress with these counters was doubled. Two species could be classed as low priority species. No points were awarded for the movement of these counters. That is, it did not matter whether a subject let them become extinct or saved them, no points could be gained.

This option gave players the opportunity to weight more heavily effort towards species they regarded as important. The linear scoring system does not award extra points for saving preferred species because it is not possible to have prior knowledge of what each subject will prefer. For instance, to suggest beforehand that subjects will prefer birds over other species has no prior support to qualify as a universal generalisation.

However, this means the incentives facing a player to weight effort towards their preferences is diminished. A subject's utility function is determined by psychic rewards, the score and possible monetary payoffs[3]. Should the attraction of the final score dominate the utility function, then the psychic rewards for playing consistently with their utility function is discounted.

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[3]The simulation uses the following formal description of a subjects utility function:

$U = f(sp^*, S, M)$

In this model U is the subject's utility, sp* is the value attached to the set of endangered species, S is the subject's final score and M is the monetary rewards available.

This opportunity to select high and low priority species offers a corrective. Given that the species they prefer will be high priority, the scoring system better reflects their preferences. It pays to achieve greater progress with these species to capture the higher score that can be obtained. The process of prioritising species also gives the observer additional information about the subject's utility function.

The progress of a counter depends on the budget allocated to it and the environmental counter move by nature. The player has to allocate their budget prior to the response by nature. Nature is playing against the player in this simulation. That is, the host of environmental conditions that will reduce the odds of species persistence. These effects are always, to a varying degree, deleterious. The progress a species makes depends ultimately on the change in population. The simulation concentrates on the rate of change, rather than initial population numbers.

All species on the board have the same population growth rate function:

$$\frac{dN}{dt} = (b_1 - d_1)N - gE \qquad \textbf{(3)}$$

In this function, N is the population density of the endangered species, b_1 the birth rate (propagation rate for plants), d_1 is the death rate, E is an index of environmental factors and g is the coefficient describing the rate at which these factors affect the birth and death rates.

This rather simplistic function ignores resource constraints and density dependence. E is always positive, indicating that there will always be some environmental impacts that adversely affect a species' chances of

persistence. Given that the progress of a counter was represented discretely, that is as a movement along cells, it was not judged necessary to have a more sophisticated population growth function.

To represent the limits to population growth in any time interval all counters could not progress more than three cells in any one turn. The decline rate was also set at a maximum of three cells. This ensured that all critically endangered species (Category A) would be extinct within "ten years" (the end of the game) if nothing was done. This made constructing the outcome table somewhat simpler as well. While the range of movement was between a decline of three cells to progress of three cells, the probability distribution (see above) skewed these outcomes.

It was necessary to assume that the rate of population change for all species was the same (in the absence of human intervention). This did not imply that the birth rate or the death rate was the same. This implied that all species would experience the same range of decline to progress. The need for this assumption follows from the influence the scoring system has on the player's utility function. Part of the player's utility derives from their final score. To differentiate the rates of progress between, for example, birds and beetles would produce a distortion. If the potential change to the population of beetles is much higher than birds at every level of bids, then a subject may spend more on beetles. This would not be a product of the extra utility they derive from saving beetles over birds. It would be produced by the gain to the final score caused by favouring beetles. As such, it would limit the conclusions that could be made about the subject's preferences.

The counter-move by nature which produces the environmental index E is done randomly. This is achieved through the use of a percentage dice which generates a score of between 1 and 100. This ensures that the environmental impacts always have a negative impact on the population of

the endangered species.

To determine the outcome of the budget and this environmental index, a ratio system was used. A range system was abandoned as it created bid-distortion. In a range system, a specific outcome depends on the move by nature and the range the bid falls within. For instance, a range of between 1 and 5 might produce a 'decline of two cells' outcome. The starting point of this range (and other ranges) are Nash equilibria. Bidding more than the starting point does not shift the probability of success.

On the other hand, a ratio system is sensitive to bids. Changes to bids alters the probabilities of achieving certain ratios. Thus, the subject achieves a payoff for small increases in the bids towards species. This payoff is absent in the range system unless the increase takes it to a new range. The ratio system is much more sensitive to marginal valuation decisions. In the simulation, the ratio is expressed as the environmental score over the bid offered to the species concerned.

All subjects start the first turn with exactly the same set up. In subsequent turns, a subject's real budget and the state of the world they are in will differ from others. Hence, the first turn is the only turn where bids between players are strictly comparable.

Subjects have to allocate a restricted budget among the candidate species. This constraint accurately communicates to the subject the opportunity cost of preferring some species to others. An increase in the bid offers to some species must be offset with decreases to others. Indeed, the game puts the player in budgetary stress. A player would have to be abnormally fortunate to save the entire set of endangered species. Given the recommended technical costs of saving the species it is impossible to save all the species with the budget granted.

While the players could achieve similar scores with different sets of preferences some kind of optimising strategy needed to be adopted. The characteristic of this strategy is the initial, deliberate abandoning of several, high technical cost species. This prevents some subjects from achieving a low score by acting consistently with their utility functions.

Player Utility Functions

There are three general utility functions that are penalised in the game as a result of the resource constraint. The first is if there are species in the high technical cost range that have more utility than several others in the low cost region. For instance, a species in Category A might yield twice or greater the utility of a species in Category B (or four times the utility of a species in Category C). That is, the opportunity cost of preferring a Category A species is to abandon two category B or four category C species. It is possible that a player may be willing to make this valuation.

The counter design however should eliminate this possibility. Counters are very substitutable because a subject does not know what species a counter represents (assuming the scientific name of this species would mean anything anyway). The critically endangered species are well represented by species with low population densities. In part, this is because part of this pool comes from isolated sub-species of mainland species. Hence, the degree of genetic distinctiveness of this group is not pronounced. Further, low population densities count against conspicuous ecosystem roles.

In consequence, for each species in Category A there are several in less-at-risk categories that have biological values equal to or greater than the corresponding Category A species. It would be irrational for a player to prefer a species at higher technical cost **and** with the same or lower biodiversity measures (relative to substitutes in lower cost regions). The

subject would have to suffer considerable regret at experiencing initial extinctions to prefer these species. This point is developed below.

The second utility function would be that subject that had both a very high disutility from extinctions and a high discount rate (that is, is very myopic) with respect to time. This subject would be very keen to avoid initial extinctions and would discount the prospect of extinctions in the future. This would weight effort towards species with high technical cost (as this is strongly correlated to time to extinction).

A subject that experienced disutility from extinctions but did not have a very high discount rate would not prefer species with high technical costs. Overall, the number of extinctions can be reduced by accepting some initial ones in exchange for less in subsequent turns. The problem with refusing to accept initial extinctions from the set of species in Category A is that funding for less at risk species is crowded out. These species are also in a state of decline. As they decline, the pool of species that are critically endangered rises. This tends to overwhelm the players budget and increase the number of extinctions at the end of the game.

There is a third group of subjects which may find it difficult to score well in the game by favouring a particular utility function. This is if the subject feels that a large number of the species has zero value relative to another set. For instance, the subject may only have an interest in birds and thus place no value on saving invertebrates.

Explicit to the game is the rule that no player is allowed more than two low priority species. This means that a subject could not have more than two zero value species on the board. This player is still penalised for neglecting species that have zero value to them. The game puts the same value on these species as any other species.

It is thus necessary to assume that the subject that plays this game has a number of constraints on their utility function:

1. Species tend to be substitutable for each other. Some species may be marginally more valuable than others but this margin is not of a significant magnitude.

2. The subject's discount rate is not extraordinarily high. As the game is set up the subject would have to discount additional extinctions to zero from ten years in the future. It is judged unlikely that anyone who experiences utility losses from extinctions would be unaffected by these occurring ten years in the future.

3. The subject places positive values on the majority of species in the endangered set. That is, their values make them unwilling to accept more than two extinctions in the course of the game. An interest in saving species extends to the majority of species requiring saving. Additional species must be neglected because the opportunity cost of saving them is too high rather than zero values being attributed to them.

The simulation generates a large number of marginal valuations. The process of selecting species as either high priority or low priority species provides information on a subject's valuations. Specifically, an observer is able to note what parameters are common to species selected as high or low priority.

The bids reflect marginal valuations of a variety of factors. Players can be observed reacting to cost, risk, organism class and biological parameters. Further, the bids actually offered to each counter provides cardinal values of

these factors. Thus, the simulation provides information on the relative value of these factors.

The simulation thus provides answers to a number of policy questions. These include the relative importance of biological factors, risk and cost. Further, it points to likely policy direction caused by technical costs. The results that are obtainable and the policy implications of the results are presented in the following chapters.

CHAPTER 6
RESULTS

This chapter is divided into three sections. The first section considers the data and the analyses that will be undertaken. The second section discusses the results that are obtained from the statistical analysis. The third section presents the statistical results in tabular form.

6.1 Statistical Analysis

A discussion of the independent variables follows. It is noted that the simulation purposely restricts the factors influencing subject bidding behaviour. Diagnostic tests demonstrate that the data is heteroscedastic in nature. This prompts the use of the PROBIT, LOGIT and TOBIT regression techniques. Statistical analysis is undertaken on three models of bidding behaviour.

Variables

The simulation purposely limits the number of independent variables affecting a subject's bidding behaviour. There is for instance, no variable for mega-fauna effects in the explanatory variable set. Likewise, by only providing a 'code' for the species and not a name, the effect of aesthetic factors is also limited. The subjects do not know which bird has "green feathers" and which has "red feathers". The subjects therefore can only respond to the variables presented to them during the game.

The one economic independent variable each subject faces is cost. For each species, the subject is told what minimum viable bid is necessary to obtain an increase in population status, in expectational terms. In the presence of the mean strategy selected by "nature", this bid would be just sufficient to generate a rise in numbers. These posted costs varied from

five to twenty "resources" out of a budget of two-hundred. This independent variable was coded in the analysis below as C (for costs). There is a perfect correlation between cost and risk of extinction in this simulation.

The subjects also had to face a number of biological variables. This set was limited to first, the taxonomic group to which the species came from and second, two biodiversity measures. Each species was drawn from one of five taxonomic groups. Taxonomic group was represented as a dummy variable. The groups that a species could be drawn from were plants, invertebrates, reptiles (includes amphibians), fresh-water fish and birds. These variables were coded as PL, IN, RP, FS and BD.

The biodiversity measures were genetic and ecological. These had values ranging from one (lowest) through to three (highest). It was presumed that these values were generated by some process that the scientific community had agreed upon. For instance, it was presumed that a phylogenetic approach could generate genetic values (Vane-Wright *et al.*, 1991; Solow *et al.*, 1993). Ecological value was coded as ECOL and genetic value as GEN.

The dependent variable, used throughout the analysis, was the bids offered by the subjects. Thus, the basic model of bidding behaviour, with n independent variables was:

$$Y_t = \alpha + \beta_{1t} X_{1t} + \ldots \beta_{nt} X_{nt} + \mu_t \qquad \textbf{(1)}$$

where the dependent variable Y_t is the bid offered by a subject, X_{it} is the ith independent variable and μ_t is the unobservable error term.

The independent variables used in this analysis showed little tendency to be correlated (Table 6.1). In only two cases, did the correlation statistic

exceed $|0.4|$. The first case, the genetic value and cost has a correlation of -0.407 and in the second case, genetic and ecological value have a correlation of 0.501.

Sample

The analysis below is based on the games played by a group of nine economics graduate and senior students from the University of Waikato. The bids for the initial turn only were compared. This had the effect of eliminating income effects, generated by competent players in subsequent rounds. Some subjects were more successful at promoting species than others. This lowered the "expected cost" of saving a species for these competent players. Thus, they experienced a positive income effect. In consequence, only in the first round did all players start on the same "level playing field".

Other samples were tested, including MBA students at the University of Waikato. However, it was not possible to give this sample a trial game. Performance of this sample was highly erratic (and related to whether the subject had read the game rules before the exercise). This limited the usefulness of the results produced by this group. It also raised the suspicion that in general, subjects would improve their "performance" as a result of learning.

The sample of economics students, produced two sets of data. The first was the set generated during the trial game. This game did not feature money payments. The second set of data came from the real game. This was played subsequent to the trial game and included money payoffs.

With a set of nine players and twenty-four species, each game produced 216 bids. These bids are strongly comparable between players. Each

subject faced exactly the same conditions as the other players. For instance, their budgets and their species' set were identical.

The number of bids produced by the game was relatively high. As a result, the regression analysis could assume that the independent variables were normally distributed. Basically, the central limit theorem and the law of large numbers are invoked (Ramanathan, 1992).

Heteroscedasticity

It is desirable to have a regression model, that is the best linear unbiased estimator of the parameters. However, if the error structures are heteroscedastic, then the ordinary least squares regression technique will not be *efficient* (Ramanathan, 1992).[1] It was thus necessary to conduct diagnostic tests for heteroscedasticity on the error structure. For these tests the bids became the dependent variable and the cost became the independent variable. The null hypothesis used in this test was that the data was homoscedastic. However, the χ^2 tests were all significant at a 99 percent level of confidence (Table 6.2). Thus, it was necessary to reject the hypothesis that the data was homoscedastic.

Heteroscedasticity is not an unusual occurrence in cross-sectional data sets (Ramanathan, 1992). This effect is caused by subjects willing to increase bids as costs rise to some threshold point. Past this threshold bidding abruptly stops. For instance, high cost candidates were in serious risk of getting no funding at all by many of the subjects. This caused the error term in equation 1, to increase, as costs rose.

Qualitative and Limited Dependent Variable Regressions

[1] Ignoring heteroscedasticity and using the OLS approach, generates an estimator of greater variance than an approach that accounts for heteroscedasticity. (Ramanathan, 1992).

Given the presence of heteroscedasticity, an OLS regression was not appropriate. As a result, a limited dependent variable regression technique was selected to overcome this problem. The OLS approach is also inappropriate when the dependent variable is qualitative. For instance, if the dependent variable is either YES or NO, then the OLS technique can not be employed.

Qualitative dependent variables can be expressed as a 0-1 dummy variable. In this situation, there are two regression techniques that can be employed. These are the PROBIT and LOGIT regressions. These are typically used in dichotomous choice, CVM studies. For instance, all of the CVM studies of species, outlined above (Chapter 4), used PROBIT and LOGIT analysis.

In these models an index I is created which is a linear function of the independent variables. The index is translated into a 0-1 range by the use of a cumulative density function. The two approaches are distinguished by the cumulative density function used. The PROBIT model uses the normal distribution while the LOGIT model uses the logistic, cumulative density function (Amemiya, 1981).

There are a number of ways of calculating R-square in these models. The output in Section 6.3 gives four of these. These are the Maddala R-square (Maddala, 1983; *eq. 2.44*); the Cragg-Uhler R-square (Maddala, 1983; *eq. 2.50*); the McFadden R-square (Hensher and Johnson, 1981; *eq. 3.32-3.33*) and the Chow R-square (Chow, 1983; *eq. 47*).

The coefficients are also amenable to analysis. One test of interest is the likelihood ratio test. This tests the null hypothesis that all the coefficients (except for the intercept) are zero. The statistic has an asymptotic $\chi^2_{(k-1)}$ distribution.

The coefficients in these approaches, do not show the effect of a change in the independent variable on the dependent variable. Rather, they show the effect of a change in the independent variable on the *index*. For this reason, the output in Section 6.3 also includes elasticity measures. These do show the impact of a change in the independent variable on the dependent. Since the elasticity is different for all observations, the elasticity values are for the mean values of the variables.

The output in Section 6.3 includes prediction success tables for these models. This approach uses a decision rule where, an observation is predicted to be 1, only if the index value for that observation is equal to zero or positive. These predictions are compared to the actual observations.

The TOBIT approach is an alternative regression technique. It is used when the dependent variable, has a discrete jump at some threshold value or limit. The particular problem here is that the dependent variable is only observable for data above the limit.

Like the PROBIT and LOGIT methods, the TOBIT approach constructs an index. If the observation is less-than or equal-to the limit, the index is set at zero. Otherwise, the index takes on the value of the observation. The data is thus divided into two sub-samples. A probability distribution is derived for each of the two sub-samples. Then, a joint probability distribution for the entire sample is derived. The regression technique uses a maximum likelihood procedure (Amemiya, 1981; Ramanathan, 1992). The goodness of fit of the model, is calculated by the *squared correlation between the observed and expected values* (Section 6.3). Output from this model includes coefficients and elasticities for the independent variables.

Models

The statistical analysis was based around three different models of bidding behaviour. Each model had a unique, dependent variable.

In the first model, the dependent variable was the bid offered to the species. This bid was expressed as a proportion of the suggested minimum cost. For instance, a ratio of one implies that the subject places no premium or discount on that particular species. A ratio greater than one, implied that the species had a premium attributed to it. The data thus comprised two sub-samples. The first, was all the species that attracted a zero bid (the limit). The second sub-sample was all the species that received some level of funding. In this model, the TOBIT approach was used.

The independent variables used in this approach were C (cost), GEN (genetic value), ECOL (ecological value), BD (bird), PL (plant) and NO. The variable NO is the number of representative counters in a particular group. This was intended to identify the presence or absence of diminishing returns from saving species from the same taxonomic group. If diminishing returns were a significant factor, groups with a large number of candidates, such as birds, may have a lesser value than groups with a low number of candidates, such as fish.

The two taxonomic groups were introduced to identify any aesthetic impact. It was assumed that plants, generally, had less aesthetic appeal than birds. If this was the case, then these coefficients would be statistically significant. In addition, the coefficient for plant would have a lesser value (lower elasticity) than the coefficient for bird. These two

groups had a similar number of candidates, thus the presence of diminishing returns would not impact on the data.

The results from this model are given in tables 6.3 and 6.4. Table 6.3 reports the results from the real game while table 6.4 reports the results from the trial game. These results are discussed below in section 6.2.

The second model, accounted for those species for which the bid was above the minimum posted cost. The cardinal value of this set of species was smaller than in the first model. Thus, this model is attempting to account for bidding behaviour, where a *premium* is attached to a species. The dependent variable ranged from zero to one. A score of one for the dependent variable was noted only if the species had attracted a premium bid. If a premium was not attributed, the dependent variable took on a value of zero.

The independent variables in this model were both economic and biological. The economic variable was the cost and the biological variables included ECOL, GEN, FS, BD, RP and IN. It was not judged necessary to include the dummy variable for plants, as any species that did not belong to one of the animal groups listed above, had to be a plant.

The data was analysed using both PROBIT and LOGIT approaches. Again, the trial game and the real game were tested separately. The results are given in tables 6.5, 6.6, 6.7 and 6.8. These results are discussed in Section 6.2.

The third model deals with the selection of high and low priority species. These species are on the very margins of bidding behaviour. In this case, the dependent variable is simply whether or not the species was selected as a high priority candidate, or alternatively whether or not the species

was selected as a low priority candidate. The independent variables were unchanged from the second model above. The results are given in tables 6.9 through to 6.16. Both LOGIT and PROBIT analysis was done on both priority cases, for both games. These results are discussed below.

6.2 Discussion

The results of the statistical analysis are discussed below. The impact of the independent variables on bidding behaviour is described. The presence of learning behaviour and reversed instrumental preferences is identified. The results are partly corroborated by a study of actual bidding behaviour at the Otorohanga Kiwi House and Native Bird Park.

Independent Variables

Cost was found to be a significant factor influencing the bidding behaviour of the subjects. For instance, in the TOBIT analysis of model one, cost was statistically significant in both the real and trial games. Subjects, were deterred from offering high bids relative to recovery costs, if the posted recovery cost was high.

The elasticity of costs however, changed between games. In the trial game, the bid offered had an inelastic response (-0.58) relative to costs. In the real game, the bid offered had become elastic (-1.27). The subjects increased their sensitivity to this particular variable.

In the PROBIT and LOGIT tests of the premium bids, costs again were statistically significant. In these tests, it was also observed that the elasticity measure rose between the trial and real game. For instance, in the PROBIT analysis, the elasticity of cost, was -1.64 in the real game

and -1.02 in the practise. This is consistent with the preferences revealed in the previous, TOBIT model.

However the third model, which considered the selection of priority species, revealed that cost had no deterrent effect. In the real game, the selection of high priority species, was not observed to be influenced by their costs. However, in the practise game, the selection of high priority species was influenced by cost (Table 6.11 and 6.15). In this case, the coefficient was *positive*. This implied that for the practise round, subjects were willing to place a high priority on those species most at risk from extinction. This preference, did not survive into the real game. Thus, rather like the two models above, the initial position of players was to put a relatively low weight on costs. In the real game, this weight had been revised, and a preference away from high cost species was noted.

The two biodiversity measures also influenced bidding behaviour. In the first model, the ecological value of the species was statistically significant, for both real and trial games. However, the genetic value, was only important in the trial game. The reduction in the importance of genetic value, occurs simultaneously to a reduction in the importance of cost. However, this is extremely unlikely to be the explanation for the diminished concern regarding genetic value. After all, cost has a negative correlation to genetic value (-4.07). Hence, if the fall in importance of genetic value was been driven by the increased concern about cost, genetic value would be increasing in value rather than decreasing.

The elasticity measure of ecological value, increases between the trial and real games. This is indicative of its increasing weight and influence. The sign of the coefficient illustrates that rising ecological value has a positive impact on the subject's bidding behaviour. The second model is

consistent with this result also. The PROBIT and LOGIT analysis show, that in the real game only ecological value is important. Further, this importance increases from the practise round through to the real round.

In fact, in the trial game, ecological value starts with no statistically observable effect. Conversely, genetic value is initially important, but by the time the real game is played, has no effect on the subjects willingness to offer a premium to any particular species.

In the third model, the selection of high priority species is also influenced by the biodiversity measures. However, despite cost not having an observable effect on bids, genetic value still falls in importance relative to ecological. For instance, in the trial game, PROBIT model, the elasticity of genetic value is 3.52 while for ecological value, the elasticity is 1.39. The same tests, in the real game produce elasticity measures of 3.43 for ecological value and 1.95 for genetic value.

These elasticity measures have also risen, as we have progressed through our model. If we concentrate on the real game only, then in model one, which accounts for all non-zero bids, the ecological value had an elasticity of 0.57. In model two, which accounts for all bids with an attached premium, ecological value had an elasticity measure of 1.55. In model three, which accounts for the highest priority species, the elasticity measure is 3.43. This indicates, as subjects moved to the margins of their budget, biodiversity measures became more important as a means to discriminate between candidates.

Taxonomic group had less impact than was anticipated. In the first two models, there was no statistically observable effect. Subjects were not using the particular taxonomic group a species was drawn from, to discriminate between candidates.

However, taxonomic group, was employed as a selection criteria for high priority species. In the real game, all groups except for the birds, had an observable, positive impact on selection. Thus, animals appear to be preferable to plants, for this sample. In the practise game, the fact that a species was a bird or invertebrate, had no observable effect.

The final independent variable used was the number of species in each particular taxonomic group. This had no effect in the real game of model one. It was statistically significant in the practise game. The effect was negative on bidding behaviour in this round. Thus, the more species in a particular group, the less likely a particular species was to receive a non-zero bid. As noted above, this effect did not survive into the real game.

Learning and Preference Reversal

In a discussion on rationality Harsanyi (1988) identifies misinformed preferences as a source of irrational behaviour. One category of these was misinformed instrumental preferences. The subject has clear, well-informed preferences towards an outcome. However, they may select an inappropriate means (or instrument) to fulfil this preference. Their behaviour appears irrational, because they are unable to achieve the outcome they value the most (Harsanyi, 1988).

One source of misinformed preferences, is the lack of knowledge about the set of feasible outcomes. A subject may not know, what it is they can achieve for a given budget. Expenditure on conservation, produces small incremental improvements in the probability of persistence. Within a short time frame, this improvement may not be perceptible. The subject must therefore form expectations about the feasible outcomes, prior to any good knowledge on the odds.

The subjects may start with expectations that are too high. They may, in a situation of optimism, expect to achieve a goal outside the feasible (or likely) set.[2] The simulation is designed to correct for this, by running a trial game first. This gives the subjects a source of data to identify more realistic expectations. Harsanyi (1988) also argues that better information is the best way to overcome this problem.

Learning behaviour does seem to be exhibited by the players. Of the nine subjects concerned, seven managed to improve their scores in the real game relative to the practise (Table 6.18). The presumption that players would begin the game with misinformed preferences appeared to be correct. Players were able to improve their association of a particular bid with a particular outcome. As a result it is not surprising that there were marked improvements in the R-square correlations in the real game relative to the practise game.[3]

For instance, in the TOBIT model the R-square value increased from 32.3% to 48.7%. The second series of statistical tests were on the willingness of a subject to bid more than the posted, minimum recovery cost. That is, what motivated a subject to bid a positive premium. For the PROBIT model, improvements in the McFadden R-square were from 16.9% to 28.6%. In the LOGIT model, this improvement was from 17.1% to 28.5%. The Cragg-Uhler R-square improved from about 25% to over 42%. The third series of tests were on the willingness of a subject to select a high (or low) priority species. Taking the High Priority Case first: the McFadden R-square rose from 24.1% in the practise game to 32.8% in the real game for the PROBIT model. Similar rises occurred for the LOGIT model.

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[2] This is discussed in greater detail in chapter 7.

[3] It is worth noting that the R-square fell for the low -priority species. This may be a result of the "regret" experienced at losing species. A lottery process may have been used by players to select their doomed species. The lottery could be perceived as "procedurally fair" (Ng, 1988).

Learning is also revealed by preference reversals between the games. In the TOBIT model there is preference reversal with the genetic and ecological variables. In the practise game, both variables are statistically significant (Table 6.4). However, the genetic coefficient at 0.48 is greater than the ecological coefficient at 0.37. Thus, in the practise game, genetics is a factor that is preferred to ecological rank. In the real game, genetics is no longer statistically significant. Further, the ecological coefficient is now over twice the value of the genetic (Table 6.3). The subjects have determined that it is preferable to rank ecology ahead of genetics.

In the PROBIT and LOGIT models of bids of a positive premium this reversal of preferences between ecology and genetics is still present. In the practise game only genetics is statistically significant (Tables 6.7 and 6.8). Likewise, its coefficient is greater than the ecological coefficient. This is reversed when the same subjects play the real game (Tables 6.5 and 6.6). As with the TOBIT model, ecological value is now greater than genetic value.

This reversal is carried through right to the high priority margin. Between the practise game (Table 6.11) and the real game (Table 6.9) players exchange a preference for genetic value with ecological value. The coefficient and elasticity of each variable also displays this swing. Ecological value in the real game ends up with about twice the weight as genetic value.

Another preference reversal shown at the margin is the impact of cost. Subjects in the practise game have a positive coefficient with cost that is also statistically significant. In effect, the players were more willing to make a costly species high priority in the practise game. This

willingness is lost by the time the players attempt the real game. Costs now have a negative coefficient and the variable is now longer statistically significant.

Strategies

This preference reversal is related to a shift in the strategies adopted by players. From the TOBIT model above, it was observed that there was an increase in the number of species abandoned by players between the practise and real games. In the practise game, players abandoned 43 species to extinction. In the real game this was increased to 57. Thus, a shift away from a fire-brigade strategy towards an optimising strategy was observed.

A fire-brigade strategy is one where the species that are at most risk are targeted first. That is, species that are at greater risk of extinction (and correspondingly have higher recovery costs) are prioritised. Symptomatic of this is low sensitivity to costs and in some cases, a positive link between costs and priorities. An optimising strategy is one where species that are relatively low cost, low risk prospects are prioritised. A player adopting this strategy has a greater willingness to abandon species and is statistically, more sensitive to costs.

A shift towards an optimising strategy was observed. Players became more sensitive to costs and were increasingly deterred from backing costly species. This is evidenced in the increase in statistical significance of costs and the rise in the coefficient and elasticity of this variable, between the practise and real game.

Likewise, the choice of high priority species was, in the *practise* game, positively related to cost. A high cost species, all things being equal, was *more likely* to be selected as a high priority counter. This implies a fire-brigade type strategy in this game.

The positive effect of costs on priorities did not survive into the real game. Costs became statistically insignificant with a negative coefficient (Table 6.10) overall. Again, this is indicative that the strategies adopted by subjects changed between the two games. A fire-brigade bidding strategy did not produce the outcome subjects had the highest preference for. This is consistent with misinformed preferences.

Weights

In terms of rank preferences the PROBIT-LOGIT tests on premiums and PROBIT-LOGIT tests on priorities supply data. For instance, the higher the coefficient (or elasticity), the higher the relative value of the taxonomic group concerned. The coefficients provide a source of weights that could be applied to policy.

Following the real game (Table 6.6), the preference ranks (highest to lowest) and coefficients are: Reptiles (0.41); Fish (0.30); Invertebrates (0.29); Birds (0.22). These preference orderings are maintained through the LOGIT test (Table 6.7).

In the high priority cases the preference rankings have reptiles and fish exchange ranks. In the real game (Table 6.10), the preference ranks (highest to lowest) and coefficients are: Fish (1.17); Reptiles (0.92); Invertebrates (0.68); Birds (0.41).

In this example the coefficients are statistically significant for all taxons except the birds.

In fact, these ranks are not the same as the practise rounds conducted by the subjects. In the case of the practise round for premium bids, birds replace invertebrates in ranking while, as is the case in the real round, reptiles are preferred to fish (Table 6.6).

In the priority tests (Table 6.10) birds again rank ahead of invertebrates, although in both cases the coefficients are not statistically significant from zero. Fish are still preferred to reptiles in the practise game. These preference ranks are maintained for the LOGIT tests.

From a policy perspective, the fact that these tests generates cardinal weights or elasticities is of enormous interest. It provides a means by which weights can be attributed to these independent variables. A simulation in this context, provides detailed data on the relative importance of the variables being considered.

Corroboration: bids from Otorohanga

The importance of cost was corroborated with an investigation of the "Adopt-A-Bird" programme at the Otorohanga Kiwi House. Visitors are invited to make a donation towards a particular species. This donation varies according to the species and can not be altered by the bidder. The range of prices posted to the visitor is a minimum of $20 for the NZ Scaup and a maximum of $500 for the Little Spotted Kiwi. The data produced from these adoptions is not hypothetical.

The sample involved 81 adoptions across 24 species. Each species was described by the following parameters: cost (the listed price supplied by the Otorohanga Zoological Society); endangered (whether the species was listed as endangered in the SPRS (Molloy and Davis 1992)); endemic

(whether the species was unique to NZ or found elsewhere) and bird (whether the species was a bird (1) or reptile (0)).

The dependent variable in this model was the number of times each species was adopted. The limit was set at one. Thus, if the species was adopted only once, the dependent variable was set at zero for that observation. In consequence, this model is analysing which variables contribute to a species attracting two or more adoptions. This set comprised 75 percent of the total sample. The results are given in Table 6.19.

These results indicate that only two factors are statistically significant at a level greater than 5 percent. These are cost, which has a negative impact and endemism, which has a positive impact. Risk of extinction, captured by the endangered 'dummy variable', is not statistically significant. Likewise, taxonomic group, captured by the bird 'dummy variable', is not statistically significant.

The variable bird has a positive coefficient. This is probably caused by the sample characteristics. Visitors to a Kiwi House and Native Bird Park are intuitively more likely to be enthusiastic about birds than other taxonomic groups. It is therefore a little surprising that the variable bird did not have a statistically significant effect.

Summary

The game the subjects played was useful to uncover their values. While it is not possible to generalise to a wider population, the game did demonstrate many preferences. This implies that the game theory technique, could be used to produce detailed utility functions for subjects, whether they be experts in conservation biology or ill-informed students.

The sample tested showed a preference for saving "more" species rather than "less". An optimising strategy will save in the longer run, more species than a fire brigade approach. Players were willing to shift their strategy away from the fire brigade to the optimising one. The emphasis on a large set of saved species, irrespective of their scientific properties, reflects the degree to which endangered species are "substitutable" for each other.

It also reflects the dominance of cost and ecology in selecting a strategy. Taxonomic group did not have an impact on subject's decisions until the "marginal dollar" had to be allocated. If one does not know what a species looks like then aesthetic factors will play a lower role. No preference for the most-at-risk species survived into the real game.

In short, both the hypothetical market and the real market at Otorohanga revealed concern over costs relative to taxonomic group and risk of extinction.

6.3 Results- Tables

The results generated by the statistical tests (outlined in section 6.1) are presented.

Table 6.1: Correlation matrix of variables

VAR	C	PL	BD	IN	RP	FS	GEN	ECOL
C	1							
PL	**0.196**	1						
BD	**-0.195**	-0.370	1					
IN	**-0.052**	-0.412	-0.370	1				
RP	**-0.100**	-0.243	-0.218	-0.243	1			
FS	**-0.070**	-0.134	-0.120	-0.134	-0.079	1		
GEN	**-0.407**	**-0.142**	**-0.149**	**0.142**	**0.195**	**0.000**	1	
ECOL	**-0.182**	**-0.066**	**-0.112**	**-0.077**	**0.122**	**0.067**	**0.501**	1

Key: C= cost; PL= plant; BD= bird; IN= invertebrate; RP= reptile or amphibian; FS= fish; GEN= genetic value; ECOL= ecological value.

Table 6.2: Heteroscedasticity Tests

Test	χ^2 statistic
$e^2 on\ \hat{Y}$	22.15
$e^2 on\ \hat{Y}^2$	21.76
$e^2 on\ \log \hat{Y}^2$	22.33

Model 1- all bids that were above zero.

Table 6.3: Real Game/TOBIT

Variable	Coefficient	T-Ratio	Elasticity of Index
C	-0.15	-9.16**	-1.27
GEN	0.19	1.32	0.28
ECOL	0.45	3.30**	0.57
NO	-0.04	-0.89	-0.18
PL	-0.16	-0.79	-0.03
BD	-0.29	-1.56	-0.05
CONSTANT	2.16	4.28	
Square Correlation	48.66%		

Results with double asterisk (**) are significant at 1%.

Table 6.4: Practise Round/TOBIT

Variable	Coefficient	T-Ratio	Elasticity of Index
C	-0.07	-4.52**	-0.58
GEN	0.48	3.33**	0.70
ECOL	0.37	2.84**	0.48
NO	-0.08	-1.67*	-0.34
PL	0.16	0.83	0.03
BD	-0.05	-0.28	-0.01
CONSTANT	0.85	1.73	
Square Correlation	32.27%		

Results with asterisk(*) are significant at 10%; with double asterisk(**) are significant at 1%.

NOTE: the Likelihood Ratio Test is a chi-square test where the null hypothesis is that all non-intercept parameters are zero. If the test-statistic is marked * it is significant at 10% If the test-statistic is marked ** it is significant at 1%.

Model 2- all bids that were above the minimum posted recovery cost.

Table 6.5: Bid Exceeds Minimum/Real Game/PROBIT

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	-0.11	-4.52**	-1.64
BD	0.22	0.77	0.068
IN	0.29	0.97	0.10
RP	0.41	1.08	0.06
FS	0.30	0.65	0.02
GEN	0.27	1.21	0.65
ECOL	0.71	3.41**	1.55
CONSTANT	-1.27	-2.16	-1.54
MAD R2	0.31		
CU R2	0.42		
MCF R2	0.29		
CHOW R2	0.32		

Results with double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 79.29** with 7 D.F.

Table 6.5a: Prediction Success Table /Real Game/PROBIT

		Actual	
		0	1
Predicted	0	127	35
	1	15	39

Percentage of Right Predictions= 76.85

Table 6.6: Bid Exceeds Minimum/Real Game/LOGIT

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	-0.20	-4.15**	-1.77
BD	0.49	0.95	0.09
IN	0.58	1.12	0.13
RP	0.76	1.19	0.07
FS	0.64	0.83	0.02
GEN	0.45	1.18	0.67
ECOL	1.20	3.31**	1.60
CONSTANT	-2.22	-2.03	-1.61
MAD R2	0.31		
CU R2	0.42		
MCF R2	0.29		
CHOW R2	0.33		

Results with double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 79.2** with 7 D.F.

Table 6.6a: Prediction Success Table/Real Game/LOGIT

		Actual	
		0	1
Predicted	0	127	35
	1	15	39

Percentage of Right Predictions= 76.85

Table 6.7: Bid Exceeds Minimum/Practise Game/PROBIT

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	-0.06	-2.69**	-1.02
BD	0.05	0.17	0.02
IN	-0.23	-0.80	-0.09
RP	0.23	0.69	0.04
FS	0.20	0.42	0.01
GEN	0.63	2.94**	1.71
ECOL	0.15	0.80	0.37
CONSTANT	-1.56	-2.63	-2.13
MAD R2	0.18		
CU R2	0.26		
MCF R2	0.17		
CHOW R2	0.19		

Results with double asterisk(**) significant at 1%.
MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square
MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 41.66** with 7 D.F.

Table 6.7a: Prediction Success Table/Practise Game/PROBIT

		Actual	
		0	1
Predicted	0	150	39
	1	10	17

Percentage of Right Predictions= 77.32

Table 6.8: Bid Exceeds Minimum/Practise Game/LOGIT

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	-0.12	-2.66**	-1.12
BD	0.12	0.24	0.03
IN	-0.38	-0.75	-0.09
RP	0.45	0.78	0.05
FS	0.43	0.54	0.01
GEN	1.11	2.97**	1.77
ECOL	0.23	0.72	0.34
CONSTANT	-2.63	-2.52	-2.10
MAD R2	0.18		
CU R2	0.26		
MCF R2	0.17		
CHOW R2	0.19		

Results with double asterisk(**) significant at 1%.
MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square
MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 49.3** with 7 D.F.

Table 6.8a: Prediction Success Table /Practise Game/LOGIT

		Actual	
		0	1
Predicted	0	150	39
	1	10	17

Percentage of Right Predictions= 77.32

Model 3- all species selected as high or low priority candidates

Table 6.9: High Priority/PROBIT/Real Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	-0.001	-0.03	-0.02
BD	0.41	1.06	0.19
IN	0.68	1.90*	0.36
RP	0.92	2.18**	0.21
FS	1.17	2.28**	0.09
GEN	0.53	1.96**	1.95
ECOL	1.05	4.10**	3.43
CONSTANT	-4.77	-5.35	-8.68
MAD R2	0.26		
CU R2	0.43		
MCF R2	0.33		
CHOW R2	0.35		

Results with single asterisk significant at 10%; double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 63.9** with 7 D.F.

Table 6.9a: Prediction Success Table/Real Game/PROBIT

		Actual	
		0	1
Predicted	0	177	21
	1	3	15

Percentage of Right Predictions= 88.89

Table 6.10: Low Priority/PROBIT/Real Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	0.05	1.62	1.15
BD	0.52	1.43	0.28
IN	0.24	0.70	0.15
RP	-0.36	-0.61	-0.95
FS	-4.79	-0.001	-0.42
GEN	-0.05	-0.18	-0.20
ECOL	-0.04	-1.36	-1.43
CONSTANT	-1.45		
MAD R2	0.05		
CU R2	0.12		
MCF R2	0.10		
CHOW R2	0.05		

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square
MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 12.01 with 7 D.F.

Table 6.10a: Prediction Success Table /Real Game/PROBIT

		Actual	
		0	1
Predicted	0	198	18
	1	0	0

Percentage of Right Predictions= 91.67

Table 6.11: High Priority/PROBIT/Practise Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	0.05	1.89*	1.09
BD	0.22	0.63	0.10
IN	-0.03	-0.08	-0.01
RP	0.70	1.98**	0.15
FS	1.03	2.05**	0.08
GEN	1.00	3.99**	3.52
ECOL	0.44	1.95*	1.39
CONSTANT	-4.88	-5.34	-8.54
MAD R2	0.20		
CU R2	0.33		
MCF R2	0.24		
CHOW R2	0.22		

Results with single asterisk significant at 10%; double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 47.1** with 7 D.F.

Table 6.11a: Prediction Success Table /Practise Game/PROBIT

	Actual		
		0	1
Predicted	0	178	29
	1	2	7

Percentage of Right Predictions= 85.65

Table 6.12: Low Priority/PROBIT/Practise Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	0.008	0.22	0.24
BD	0.23	0.55	0.15
IN	0.75	1.69*	0.57
RP	0.47	0.63	0.15
FS	-4.39	-0.0002	-0.47
GEN	-1.26	-2.92**	-6.51
ECOL	-0.62	-1.63	-2.89
CONSTANT	1.16	1.37	3.00
MAD R2	0.16		
CU R2	0.37		
MCF R2	0.31		
CHOW R2	0.21		

Results with single asterisk significant at 10%; double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 38.4** with 7 D.F.

Table 6.12a: Prediction Success Table/Practise Game/PROBIT

	Actual		
		0	1
Predicted	0	194	13
	1	4	5

Percentage of Right Predictions= 92.13

Table 6.13: High Priority/LOGIT/Real Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	-0.001	-0.02	-0.01
BD	0.84	1.13	0.19
IN	1.16	1.68*	0.31
RP	1.65	2.13**	0.19
FS	2.19	2.46**	0.084
GEN	0.95	1.94*	1.74
ECOL	1.99	3.91**	3.28
CONSTANT	-8.74	-4.96	-8.04
MAD R2	0.26		
CU R2	0.44		
MCF R2	0.33		
CHOW R2	0.36		

Results with single asterisk significant at 10%; double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 64.7** with 7 D.F.

Table 6.13a: Prediction Success Table/Practise Game/LOGIT

		Actual	
		0	1
Predicted	0	177	21
	1	3	15

Percentage of Right Predictions= 88.89

Table 6.14: Low Priority/LOGIT/Real Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	0.09	1.54	0.99
BD	1.11	1.51	0.27
IN	0.57	0.82	0.17
RP	-0.55	-0.46	-0.07
FS	-25.2	-0.00008	-1.02
GEN	-0.18	-0.34	-0.36
ECOL	-0.69	-1.28	-1.21
CONSTANT	-2.49	-1.63	-2.43
MAD R2	0.05		
CU R2	0.12		
MCF R2	0.10		
CHOW R2	0.06		

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square
MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 12.05* with 7 D.F.

Table 6.14a: Prediction Success Table /Practise Game/LOGIT

		Actual	
		0	1
Predicted	0	198	18
	1	0	0

Percentage of Right Predictions= 91.67

Table 6.15: High Priority/LOGIT/Practise Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	0.09	1.66*	0.93
BD	0.35	0.53	0.08
IN	-0.07	-0.11	-0.02
RP	1.22	1.95*	0.14
FS	1.84	2.13**	0.07
GEN	1.81	3.82**	3.27
ECOL	0.70	1.64	1.14
CONSTANT	-8.42	-5.04	-7.59
MAD R2	0.19		
CU R2	0.33		
MCF R2	0.24		
CHOW R2	0.22		

Results with single asterisk significant at 10%; double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 46.3** with 7 D.F.

Table 6.15a: Prediction Success Table /Practise Game/LOGIT

	Actual		
		0	1
Predicted	0	178	29
	1	2	7

Percentage of Right Predictions= 85.65

Table 6.16: Low Priority/LOGIT/Practise Game

VARIABLE	COEFFICIENT	T-RATIO	Elasticity at Mean
C	0.009	0.13	0.10
BD	0.31	0.41	0.08
IN	1.29	1.52	0.32
RP	0.90	0.61	0.11
FS	-24.04	-0.00008	-0.99
GEN	-2.37	-2.75**	-4.70
ECOL	-1.11	-1.46	-1.97
CONSTANT	2.38	1.50	2.36
MAD R2	0.16		
CU R2	0.37		
MCF R2	0.30		
CHOW R2	0.21		

Results with single asterisk significant at 10%; double asterisk(**) significant at 1%.

MAD R2= Maddala R-Square CU R2= Cragg-Uhler R-Square

MCF R2= McFadden R-Square CHOW R2= Chow R-Square

Likelihood Ratio Test = 37.6** with 7 D.F.

Table 6.16a: Prediction Success Table /Practise Game/LOGIT

		Actual	
		0	1
Predicted	0	194	13
	1	4	5

Percentage of Right Predictions= 92.13

Table 6.17: Summary of Results

	TOBIT Model- Premium Bids	PROBIT Model- Premium Bids	PROBIT Model- High Priority
Observations > 0	159	74	36
C	Y	Y	N
GEN	N	N	Y
ECOL	Y	Y	Y
PL	N	-	-
BD	N	N	N
IN	-	N	Y
RP	-	N	Y
FS	-	N	Y
MCF R2 Real	na	28.56%	32.83%
MCF R2 Practise	na	16.85%	24.12%

KEY: Y= statistically significant variable; N= variable not statistically significant; - variable not included in analysis.

Table 6.18: Game Scores

Player	Practise	Real
1	164	185
2	228	180
3	210	215
4	152	151
5	173	141
6	237	173
7	201	178
8	211	204
9	194	183

Note 1: In the simulation, players were trying to achieve the lowest score possible.

Table 6.19:Adoptions/TOBIT

Variable	Coefficient	T-Ratio
COST	-0.02	-2.20†
ENDANGER	-0.26	-0.46
ENDEMIC	1.60	2.77*
BIRD	0.95	1.58
CONSTANT	0.50	0.73
Squared Correlation	49.81%	

Key: * is significant at 1% level; † is significant at 5% level

CHAPTER 7
CONCLUSION

7.1 Evaluation of Simulation Method

The simulation approach to valuing endangered species has a number of merits and shortcomings. One of these shortcomings is that the technique does not eliminate bias. However, it does limit the effects of some sources of bias. There are three sources of bias remaining in this simulated market approach. These biases are those caused by the information provided in the game, reaction to the interviewer and finally, self-selection bias.

It has been demonstrated in Contingent Valuation studies that the level of information provided to subjects has a positive impact on their bidding behaviour (Samples *et al.*, 1986). In the simulation adopted, background information on species was provided to the subjects in the *Introduction* to the game. For each taxonomic group, players were told the proportion of that group that were endangered as well as the "taxonomic" distinctiveness of that group. For instance, players were told that generally, NZ invertebrates had fewer overlap with Australian species than NZ birds.

This may account for the lower coefficient attached to birds than to invertebrates in many of the "real game" bids (chapter 6). That is, players may have decided that the number of "substitutes" to NZ birds in Australia was relatively high on the basis of the information provided. This information may have encouraged lower bids towards the taxonomic group comprising the birds.

The interviewer may also have produced a reaction among players. That is, players may have responded to what they believed the interviewer's preferences were. These subjects would have been aware that the interviewer was previously trained as a zoologist, with a background in invertebrates. Hence, they may have lifted their bids towards invertebrates in response to this knowledge.

Further, the players may also have been aware through seminars and lectures that the interviewer has been critical of conservation policy in New Zealand. The bias towards birds has been identified in these presentations. Hence, players may have been willing to lower their bids towards birds either because they wished to redress an historical imbalance and/or because they were trying to please the "interviewer". That is, a knowledge or interviewer bias was produced.

Self-selection bias was introduced with the sample selection process. The sample was not selected according to some random process. Rather, a sample from the set of graduate students volunteered to play the game. This introduced the bias to the game. On the other hand, evidence from experimental markets suggests that students do not behave differently to other members of the community (Roth, 1988). In part, this relates to the ability of the simulation designer to control aspects of player behaviour. The use of money payments and the restriction of the variables facing the subject, make the particular background of the player less relevant.

From a statistical point of view the results are not inferior to CVM studies. The study of the Whooping Crane (Bowker and Stoll, 1988) achieved McFadden R-squares (Amemiya, 1981) of 13% to 21%. In the simulation above, the same test achieved R-squares of about 28% for the "bid exceeds minimum" and about 33% for "high priority test".

Likewise, the likelihood ratio tests are similar in performance to CVM studies. In Hagen *et al.* (1992) the Chi-square was 14.9 while a range of 120.7 to 192.4 was recorded by Bowker and Stoll (1988). These were significant at the 1 percent level. Likewise, for all but two tests, the chi-squares for the simulation were well above the 1 percent level. The two tests that failed to reach this outcome were the tests in the Real Game of low priority species. This is further suggestive of a lottery process involved in these doomed counters.

Relative to a CVM study there are some practical difficulties. Simulations are costly to run and are time consuming. The material needed to run a simulation is greater than a survey. Simulations that are computer based and those that are counter based require more preparation than mail-out surveys. Further, the addition of money payments to subjects adds a cost not normally encountered with mail-out, CVM studies.

Both the subject and interviewer have to be prepared to give up much more of their time as compared to a "dichotomous choice" survey. Gathering sample data is thus a more complex procedure.

On the other hand, the approach does eliminate some of the problems found in CVM studies. For instance, embedding effects are eliminated by setting up a General Equilibrium problem. The player knows that the entire set of endangered species is in front of them.

It reduces the impact of strategic bidding by subjects. Over-bidding on one species has a cost to the player, both psychic and monetary. The impact of over-bidding is to leave the player with less funds to spread

over the remaining species. The psychic cost occurs with the higher than optimal number of extinctions. The monetary cost is the lowered "game" earnings. A less-than optimal strategy gives a lower score overall. This lowers the expected earnings of the player.

The players propensity to act strategically is thus lowered. In contrast, in a survey there is no direct cost to the subject from over-bidding. The effect on other species is unknown and thus discounted. The absence of money payments does not give the interviewer a mechanism to penalise the player for overbidding.

One of the major gains of using the simulation is to firmly relate consequence to bids. The player has an interest in linking a particular budget allocation with a particular "set of outcomes". This is developed first by practise. By trying out strategies in the practise round, they can at little cost, discover the effect of changing spending on a particular species. Further, their interest in doing so is influenced by the prospect of monetary reward. Players in other words learn. If they are mistaken about a particular strategy (have misinformed preferences) then the real game gives them the opportunity to correct this.

There are two aspects to learning behaviour that need to be discussed. The first is whether or not learning behaviour was revealed by the subjects. Did players change their bidding behaviour between the practise and real rounds? The second aspect is whether, at this stage, a strategy change can be regarded as a rational or an irrational action.

In the previous chapter a shift from what was termed a "fire-brigade" strategy to an "optimisation" strategy was noted. There was in the real game more weight attached to programme costs. The number of species

rejected from the "funded set" also increased. These species, as evidenced by the rising concern about costs, were drawn from the most expensive categories.

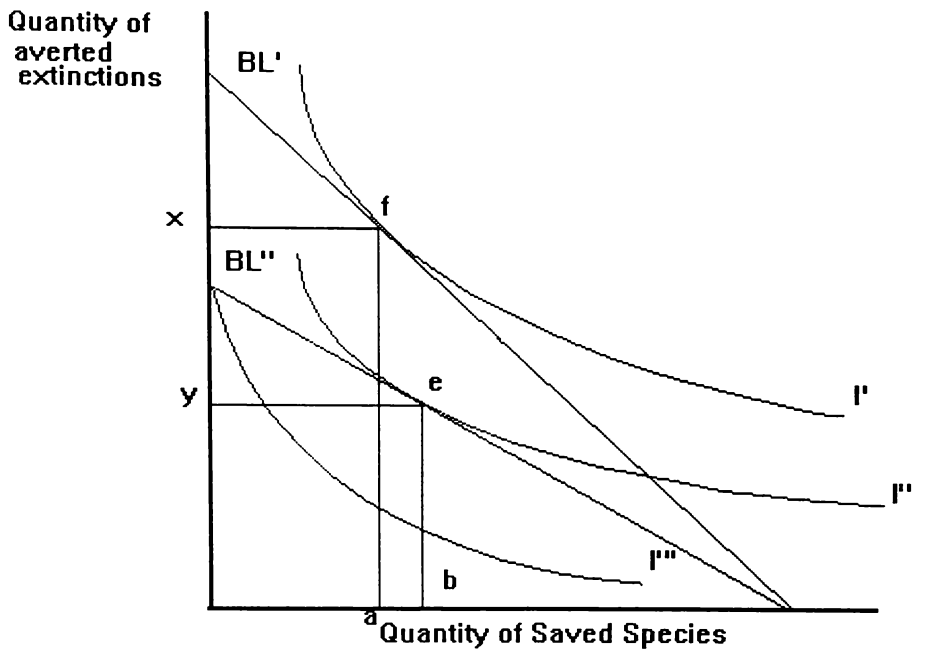
In the analysis of data an increase in the coefficient of multiple determination, or R-square, was noted between the real and practise games (table 6.19). This increase indicates that the variables were having an increased impact on subject bidding behaviour. The shift to an optimising strategy reveals that the players were trying to maximise some combination of species numbers and biological parameters.

This shift, when players had the option of continuing their old strategy, appears to be a rational change. The lower R-square in the practise game demonstrated that players did not initially have stable preferences. There would be little point in changing from the old strategy to the new unless there were welfare gains to be had. Hence, players recognised that their original strategy was the irrational (or misinformed) approach.

This shift in "instrumental preferences" can be modelled in neoclassical terms. For instance, the subject is in a position where they have two, conflicting goals. The first is to "save the maximum number of species" possible. This is achieved by improving species status so that they will no longer be endangered. Species in this category are promoted off the endangered lists. The second is to "minimise the number of extinctions". This is achieved by targeting those species most at risk. The aim is to prevent irreversible extinctions from occurring, even though the species will remain endangered. A strategy that minimises extinctions, produces very few saved species. Thus, a tradeoff between these goals is mandatory.

However, initially the subjects do not know what the set of feasible outcomes is. Suppose players consistently underprice the goal of averting extinctions. The result would be that the subjects would believe that their budget line was at a higher position on their indifference curve map (figure 7.1).

Figure 7.1: Indifference Curve Map of Subject



In this case, the subject believes that the budget line they face is BL'. This makes point *f* on indifference curve I' the optimal point. Thus their budget is allocated in the belief that they can achieve quantity x averted extinctions. In fact, the set of feasible outcomes is described by the budget line BL".

However, trying to achieve quantity x averted extinctions is impossible. Even if the subject allocated all of their budget to preventing extinctions the best result they could achieve would be the intersection of BL" and I'". In this case, no species would make it off the endangered list, and the quantity of averted extinctions would be less than x.

This outcome, results in the subject revising their strategy. They had started with misinformed preferences. The trial game, produced the realisation that the cost of averting extinctions was higher than they believed. In the real game, the subject has realised that their budget line is best described by BL" and not BL'. This causes a substitution away from averting extinctions, towards getting species off the endangered lists. Point *e* on indifference curve I" is thus the optimal point, when the subject has better knowledge on feasible outcomes.

It is interesting to note, that in this sample, players were generally optimistic about the costs of saving species initially. Such optimism is also present in some endangered species recovery work (Williams, 1986). For instance, in the latest review of the kakapo recovery programme, the reviewers felt there was nothing wrong that doubling the budget wouldn't fix. A consistent bias towards optimism, is perhaps accounted for by the irreversible nature of extinctions. Subjects are unwilling to abandon any species on the grounds that its extinction will be permanent.

Perhaps the most important feature of this simulation is that it identifies the variables that influence players bids. A CVM study typically reports that species x is doomed to extinction unless something is done about it. It is not known what characteristics of that species determine that bid. CVM studies do identify education and income among bidders as being important but this provides no useful information to a species manager (Stevens *et al.*, 1991a). As an aside, it does provide ammunition for critics of environmentalists who argue than concern for nature is a luxury good.

The simulation approach can introduce more variables, in a controlled manner, for subjects to respond to. In this simulation, taxonomic group, cost (synonymous with risk) and biological parameters were targeted. The results showed that subjects by and large were motivated by costs and ecological value (that is, how important a species is for other associated species). It is this identification of important variables that gives the simulation approach its strength. It provides more information about a group's utility function than a CVM survey. If the cost of saving a species rises or it becomes a riskier prospect, it is clear that this group would prefer putting a lower premium on it. Conversely, there is a strong preference for ecologically important species. These revealed preferences could form the basis for a more optimal budget allocation among species.

The control of the variables allows the simulation designer to properly "describe" the good concerned. A major problem with CVM surveys is defining to the subjects what precisely it is that is being valued (Smith, 1992). Where this is difficult, as in the case of endangered species (Chapter 5), it is much harder to conclude what the values of the subjects are. The simulation approach may therefore be more useful in cases where the subject is unfamiliar with the good concerned. For reasons of economy, CVM is likely to be more practical where the good is relatively familiar to the subjects.

The simulation approach is suited to answering questions about the relative value of species. It further has an educational aspect not present with CVM. It is best suited for the study of small groups. The linkage of biological impacts to economic allocations makes this a fruitful area of research.

7.2 Policy Implications of Research

The principal policy question of the research has been how to achieve an efficient allocation of resources towards a set of endangered species? The use of a simulation, appears to offer some assistance. Nonetheless, there are obstacles to achieving efficient species conservation policy.

To begin with, the importance of cost needs to be examined. Survey data (Smith, 1990; Stevens *et al.*, 1991a) suggested that the public had little concern over the costs of wildlife preservation. Available empirical data does not support this view. For instance, the CVM studies, discussed in chapter 4, all show that costs are a deterrent to increased wildlife preservation. The study at Otorohanga Kiwi House again identified costs as a strong deterrent to bidding behaviour.

The simulation is to an extent, supportive of both views. In the initial practise rounds, costs had about half the impact it did in the real rounds. Thus, to begin with, costs play a lower role in decision making. However, this strategy was regarded as inefficient, in the sense that it lead to a greater decline in species status than the optimising strategy.

It is perhaps not surprising that individuals are willing to ignore costs. There could be considerable regret experienced by someone if they learn a species has become extinct (Ready and Bishop, 1991). Given that extinction is, given current technology, irreversible, species loss is permanent.

However, if this regret is allowed to influence policy, it may lead to a rather more drastic loss in species in the future. A policy designed to prevent extinctions may differ from one designed to maximise species

persistence. It would be tempting to over-invest in species on the brink of extinction. However, given that these candidates consume, on average, more resources than less-at-risk candidates, these more persistent species may experience under-investment. Long term, this strategy may lead to a more serious loss of species.

In the United Kingdom, the fire-brigade approach to ecosystems is causing these types of inefficiencies. In the case of Sites of Special Scientific Interest (SSSI), funds are allocated without regard to the overall conservation outcome (Spash and Simpson, 1993). Rather too many resources are allocated to sites that first attract interest. This leaves very little in the way of resources to sites that attract interest later in the year (Spash and Simpson, 1993).

The Department of Conservation has signalled that it finds a fire-brigade strategy tempting. In Molly and Davis (1992) the SPRS does not include a cost component. The goal of the Department is to prevent extinctions (Department of Conservation, 1992). One consequence of this has been criticism that their approach is often not cost-effective (Craig, 1992).

The perception by Department of Conservation staff that species recovery work is not an optimisation problem is supported by the kakapo recovery programme. The annual cost of the kakapo recovery plan is about $NZ 250,000. This makes it the most expensive species currently undergoing recovery work in New Zealand. The Protected Species Director at the Department of Conservation justified this effort recently with the comment:

> *"If we give up on the kakapo, we give up on all threatened species. We lose a unique species..."* (Chamberlain, 1993).

Giving up on the kakapo is more likely to mean that there is an additional $NZ 250,000 to spend on other species rather than all other threatened species will experience a demise. Again, this reflects a lack of consistent policy for setting priorities.

In essence, it will be difficult to judge the effectiveness of any particular conservation strategy unless an attempt is made to predict its outcome. Some kind of risk analysis or simulation approach offers some advantages here. At the very least, conservation planners will have an idea of the sorts of scenarios they are likely to face. An optimistic attitude is less likely to have use for conservation policy.

The basic inefficiency of a fire-brigade strategy rests on three assumptions. The first is that the conservation agency is unable to adequately fund all candidates for recovery work. If the conservation agency is able to achieve this, then the fire brigade strategy will be successful. In this particular case the conservation agency does not have a scarcity problem. Thus, rationing of resources towards the most valuable candidates is no longer appropriate.

The second assumption concerns the rate of increase in the conservation agency's budget. The fire-brigade strategy is supposed to be inefficient because the rise in cost of programmes eventually overwhelms the agencies' (fixed) budget. However, if the fixed budget assumption is dropped, the agency might find its budget rising at a rate sufficient to offset the rise in costs.

To some extent, this has occurred in New Zealand. The initial (1988/89) budget allocation for the Department of Conservation was about $NZ 2m for species recovery work. In 1993/94, this was about $NZ 15m.

However, since the early nineties, the budget allocated for species recovery work has stabilised. Relying on continual increases in the Department's budget is not going to be prudent in the longer term.

The third assumption relates to the relative rates of decline and progress of species. This may not hold if the rate of progress is much greater than the rate of decline. In this sense, the conservation agency would be able to upgrade the status of the critically endangered candidates faster than the neglected set will decline. For instance, if three critically endangered species could be upgraded for every two less-at-risk species that decline to critical levels, a fire-brigade strategy would be more successful.

However, these exceptions are unlikely to occur. Limited resources imply that there is indeed a scarcity problem. Some species have already been downgraded in priorities given funding difficulties. The desire to obtain corporate sponsorship is not normally exhibited by agencies free of funding difficulties. Endangered species are also typically have relatively low reproductive rates. Relying on rapid reproductive success may therefore be overly optimistic.

The Value of Species

The simulation basically forces players to trade-off between two goals. These are to have the most species recovered irrespective of their biological values or to maximise some kind of biological function. Most players managed to do this by in the first instance, rejecting all species judged to be too costly to justify a recovery programme. Only then did they place premiums on species with higher, on average, biological values.

This strategy is consistent with one that maximises biodiversity saved. The more species that are saved, the greater the biodiversity that will be preserved. This is of course subject to a representative sample of taxonomic groups being preserved. Saving forty species of kauri snails would preserve less biodiversity than some combination of forty birds, lizards and invertebrates. Player bids were directed towards a mixture of taxonomic groups. After all, there was no preference for taxonomic group until the high priority species were selected.

This bidding behaviour is based on the principle that all species, are too some degree, substitutes for each other. The claim that a particular species is unique is often made by conservation agencies. However, this appellation applies to all species, in the sense they must differ somehow from their relatives in order to justify species status. Hence, the view that each species is unique and irreplaceable is both true and worthless for decision making.

Endangered species and their relatives have in common, genetic material (Solow *et al.*, 1993). Thus, saving one particular species is a way of also preserving a high proportion of genes from its relatives. Recognising that species can be substituted for another is a necessary step to take to achieve an optimal preservation outcome. Otherwise, a recovery programme will be judging species independently of other possible candidates. Optimisation requires that all proposed programmes be judged in light of their costs to other species.

Part of the justification for treating species as substitutable is the shared-gene rationale. Another rationale is the generally high level of ignorance about endangered species. The actual species known to be endangered by the public is a very small sub-set of the total. It makes

more sense to describe these species in terms of their biological parameters than to name them in a simulation. The assumption that any two obscure plants, from the same genus, are equally preferable as candidates for recovery work, is thus realistic.

The use of simulations can identify which properties of species carry value and which preservation outcomes are preferred. On the other hand, putting into place the policy insights from such a valuation approach is likely to be difficult. As noted above (Chapter 2) species preservation occurs in a human institution. This can create incentives that diminish conservation effectiveness. Government failure is another aspect of the conservation problem.

Work in public choice theory identifies a preference for bureaucracies to expand their budgets. This is also the same as a preference for defending their budget in times of public sector fiscal stress. This institutional preference leads to an interest in species of high profile. It is assumed that taxpayers prefer to save species that are aesthetic, for instance birds and dolphins rather than snakes or wetas.

Targeting high profile species is intended to maintain public support for the conservation agency. This can be illustrated with the following (Australian) excerpt:

> *"As a public sector employee, it is not unexpected that recovery programmes would be targeted at appealing creatures rather than creatures that would not catch voters attention..."* (Sullivan, 1994).

Similar patterns have been observed in both New Zealand and the United States. In the USA a study by Metrick and Weitzman (1994) demonstrated a bias towards large animals and birds. This was despite the Fish and Wildlife Service having objective criteria to select

candidates. In New Zealand, about half the recovery programmes in 1992 were for birds (Department of Conservation, 1992) whereas this group made up only one quarter of the total endangered species (Molloy and Davis, 1992). In 1993, there was in fact a greater shift towards recovery programmes for birds (Department of Conservation, 1993).

A preference for high profile species is inefficient if it is reinforcing a fire-brigade strategy. This results in the by now familiar problem: under-investment in less-at-risk candidates. It is perhaps not unreasonable to assume the public prefers aesthetic species to others. It does not follow that this preference will dominate a strategy that saves the largest number of species.

The second area government failure may manifest itself is within the conservation agency. Species conservation lends itself to a number of principal-agent problems. To begin with, there is a high degree of difficulty in monitoring the performance of agents. Species recovery programmes, especially in New Zealand, take place in island refuges. The population of the species and the staff on the ground, are a long distance geographically from head offices.

This geographic separation of staff from the species manager makes it difficult for the manager to monitor their performance. Changes in species status may take a long time to eventuate or notice. If work on the sanctuaries is not being carried out properly, there may be a significant lag between agent-shirking and when this is actually observed. As breeding failure can be attributed to stochastic factors, discerning shirking is also going to be difficult.

Such a monitoring problem may have led to inadequate rat control for the kakapo on Little Barrier Island. In 1993 a chick disappeared, possibly through rat predation. Subsequent trapping by the technical director of the kakapo recovery programme caught seven rats at the nest site (Chamberlain, 1993).

There is unlikely to be a serious principal-agent problem if all parties have similar preferences. In the case of different preferences, the lack of monitoring ability is going to contribute to an inefficient use of resources. Preferences may diverge in two ways. The first is instrumental. This occurs when the parties disagree about the proper instrument to achieve their policy goal. The second divergence is when the parties have different preferences regarding the policy outcome.

Differing instrumental preferences are observable in both the USA and in New Zealand. In the United States it was thought that the Californian condor was a sensitive animal whose survival depended on habitat protection and being left alone. It was not until the nineteen-eighties that captive breeding started. Right from the outset, conflict occurred. This was a product of the various governmental and private organisations that were determined to play a role here. The conflict between the leave-the birds-be and the capture-and-breed camps was intense. The National Audubon Society even ended up suing the Fish and Wildlife Service (Cohn, 1993).

In New Zealand, differing instrumental preferences have influenced the course of conservation strategy in the case of the kakapo. For instance, staff in the Auckland region conservancy were removing the nuts from supplementary feeding stations. They had a concern that the high fat content of nuts would lead to heart-disease in the birds. The national

director of the programme felt otherwise. Nuts were necessary to build up the weight required for lekking and nesting (Chamberlain, 1993).

There are two errors that can be made when conservation policy is put in place. The first is that a programme adopts techniques that lowers the odds of survival rather than increases it. The second is if techniques that would be successful are not adopted. It is probably the case that no one, involved in species conservation, wishes to be known as the person who implemented a policy directly responsible for a species' demise. In this case, the second error may be more likely, as perhaps the examples above illustrate. In short, there may be an institutional bias against innovative, but effective conservation policy.

The principal-agent problem can be quite pronounced when several, largely autonomous organisations, are involved. Within the Department of Conservation the regional conservancies enjoy considerable autonomy. The director of a particular recovery programme has little management authority over them. When private conservation groups are added, over which there are few sanctions or rewards, differences in the preferences of the various parties can be a serious impediment.

A contributing factor to different preferences among staff is the specialised skills held by Department of Conservation staff. A high skill level in breeding endangered parrots for instance, is not easily transferable to other public sector bureaucracies. For a given conservation agency, there are few ways for staff to advance other than being associated with successful programmes. It pays to belong to relatively more successful programmes and have other staff associated with relatively less successful.

This leads to parties involved in conservation effort, having slightly differing policy outcome preferences. Endangered species are normally a nationally important target. The manager has a rather simple goal, ensure the unqualified success of the programme. Policy may also be slanted in a way to attract personal kudos and enhance personal promotion. However, regional staff and field workers may not share this preference. They may have regional goals they also feel are important and are unwilling to downgrade their own projects. Lowering the likelihood of success of their own projects to advance the project headed by someone else, is going to be particularly unrewarding.

Private conservation groups are also involved in breeding programmes. While they can offer a high level of support for conservation work, their antipathy to the political goals of the species manager is likely to result in the under-employment of their services. This can be illustrated with an excerpt from the Otorohanga Zoological Society's Presidential report:

> *"...From the Society's view we were disappointed in the project. We received little or no credit fro the contribution that has been made by our society over the years to the breeding of Kiwis. This is strange as we are the only institution that has the three species in captivity. We had no input into the programme and a lot of work and research that is to be undertaken has already been researched and recorded by us."* (Vellenoweth, 1992).

The kakapo recovery plan also provides evidence that regional implementation of the plan has been, at times, less than adequate (Chamberlain, 1993).

One policy question that seriously needs consideration is the respective involvement of the public and private sectors in species conservation. While the public sector provides the bulk of the resources, there is no *a*

priori reason why this should be so (Bennett, 1994). Species preservation generates both public and private benefits.

In fact, the public sectors involvement extends beyond the resources it commits. The breeding of endangered species in wildlife sanctuaries and zoological parks can not occur unless the public sector grants a permit. In the case of keas, these permits are being revoked in an attempt to discourage the captive breeding of the species (Rooney, 1995). The gains from this move appear uncertain. Thus, the regulatory environment species conservation takes place in, also needs investigation.

Existing policies and institutions show that:

1. Valuation of species, for budget allocation purposes, is poor. Quasi-scientific criteria and political acumen seem to be important. The simulation approach outlined in this thesis would address this problem.
2. Species recovery programmes currently in place appear to be prone to inefficiencies. Recovery work is rather dependent on the goodwill and dedication of Department of Conservation staff.
3. The near-monopoly powers of the public sector, restrictions on the breeding and trading of species, and current environmental law, provide an unsatisfactory environment for private sector support.

Finally, the problem of determining relative values for species, can be addressed by a hypothetical market approach. The approach is based on economic concepts of value and optimisation, as well as biological modelling of populations of endangered species. This provides a methodology that can account for both the scientific and economic aspects of species conservation. The data such an approach generates provides

very detailed data on the utility functions of subjects. Allocating resources to the most valuable species, is a task made feasible by this methodology. This approach makes a significant contribution to achieving efficient conservation policy and the valuation of non-market resources.

APPENDIX ONE

SPECIES MANAGER

Brendan Moyle
Department of Economics, University of Waikato,
Private Bag 3105, Hamilton, New Zealand

Introduction

This game is a simulation exercise. The player assumes the role of an organisation responsible for threatened species' recovery programs. The game can therefore be described as a budget allocation exercise.

The player has as an opponent an unpredictable 'Nature'. The response Nature makes to the player's strategy is determined randomly. The player is playing against this Nature only (that is, not other players).

The game is analogous to "Snakes and Ladders". The player will be moving counters around a board. These counters represent endangered species. The aim is to get the counters moving towards a state of "recovery" and away from a state of "extinction".

The player attempts to influence Nature with their budget. The more that is spent on a particular species the more likely the success of a recovery program. Like the real world, there is no such thing as a sure bet. Some species will be resistant to recovery programs right to the end (no matter what is spent).

This exercise is simplified compared to the actual problem. For example, in New Zealand there are about 300 threatened species known but this game has only 24. However, the types of problems facing a species manager in the reality are duplicated here. The principal problem is that the budget is insufficient to save all of the species.

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1 Equipment

1.1 This game comes with a number of components. These include the *Players Board,* 24 counters representing the *Species* that are candidates for recovery programs, *dice* and plastic *caps* in two colours. The player will find that *Budget Sheet* is also attached. This is used as a guide for your budget allocation. It is **highly** likely that a pocket calculator will also be useful.

I.2 *PLAYERS BOARD*-The board is divided into twenty cells. As a general rule, each cell has an identification code (for example C6 is on the extreme right of the bottom row of cells). *However two cells will be referred to in name only*. On the upper left of the board is the *Extinct Species* cell. On the opposite corner of the board is the *Recovered Species* cell.

1.2.1 There are eighteen cells which have identification codes. These are divided into three groups and six sub-boards. A sub-board always consists of three cells *joined together*. The top row of six cells is the *Facing Imminent Extinction* group. These cells are A1 through to A6. This row is divided into two sub-boards. These sub-boards are 'A1 to A3' and 'A4 to A6'.

The middle row of cells is the *Highly Endangered* group. These cells are B1 through to B6. This row is divided into two sub-boards. These are 'B1 to B3' and 'B4 to B6'.

The bottom row of cells is the *Endangered* group. These cells are C1 through to C6. This row is divided into two sub-boards. These are 'C1 to C3' and 'C4 to C6'.

1.3 *CELLS*- Each cell is also given a value. This value is recorded using an *arabic numeral*. For example, in cell A3 this value is 17 while in the Recovered Species cell it is 0. While a species is in that cell it is given that value. If these values are summed this produces a *conservation score.* If by the end of the game there is a fall in this score the player has produced a trend away from extinction.

1.4 *COUNTERS*- Each counter has a *face* and a *back*. The face has *four* items of information. The first is a symbol indicating from which group the species is drawn from. For example, a counter with a small lizard on it belongs to the *Reptile* group. Underneath this symbol is an identification code for this species. This is given as a capital letter and a number. The letter indicates what group the species is drawn from. In the above case, the reptile's code letter will be "R". This letter is followed by a number which indicates which species it is within that group. For example, "R.1" would be the first reptile species. No counters share the same identification code.

1.5 The following letters are used to identify each group:

"R" = Reptiles
"I" = Invertebrates
"B" = Birds
"F" = Fish
"P" = Plants

1.6 The counter also has a *Roman numeral* in each bottom corner. These refer to the species genetic diversity and ecological value. The lower *left* numeral is the rank of genetic diversity. The lower *right* numeral is the rank of ecological value.

1.6.1 These values are given in ranks. The ranks are organised in ascending order. Thus, the higher the rank, the more important that species is using that measure. These ranks are *not relative measures* of importance. A higher ranked species is just more important than a lower rank, according to that measure. For example, a species with rank 3 is more important than a species with rank 1. These are not relative measures so it is *not three times more important*.

1.6.2 On the back of each counter is a capital letter which indicates what group the species belongs to. These letters may be "I", "P", "B", "R" or "F". These refer to the groups above (*sec. 1.5)*.

1.7 *DICE*- This simulation comes with two 10 sided die. This is termed the *Percentage Dice*. The red die is used to represent the tens and the green die is used to represent the ones.

1.7.1 The two die are rolled together. The result is found by putting the score of the red die in front of the green die. For example, if you rolled 3 with the red die and 6 with your green die your percentage dice would have rolled 36.

1.7.2 EXCEPTION: if you get a result of 00 this is treated as 100 and not 0.

1.8 *CAPS*- This game comes with a number of caps. These plastic caps are red or yellow in colour. These caps are used to indicate high and low priority species.

The scoring system is structured to favour success with your high priority species. The penalty for failure with low priority species is reduced.

2 Background

There are about three hundred species known to be threatened with extinction in New Zealand. One problem is that all of this species are facing decline. Without human intervention they are likely to become extinct. All information below is based on New Zealand.

The threats facing each species are diverse. The species most affected by habitat loss have been plants. For the other groups, predation and competition are major threats.

The effort to save endangered species has been increasing. However, there is no clear reason why some species should be saved over others. Risk and limited budgets mean that it will be near-impossible to save all species.

There are a number of different groups of species. However, each group has a different percentage of species endangered.

The Reptiles (for administrative purposes this group includes the amphibians) are a very old group of animals. Most reptiles have been long established in New Zealand. Of the known species, 60% are threatened.

Most Bird species have been established since New Zealand was connected to other land masses. Thus, they tend to be more recent arrivals and related to species in Australia. Australian species are still emigrating to New Zealand. There is a high percentage of endangered bird species with 42% recognised as threatened.

The Fish group is restricted to Freshwater Fish. This is a small group, long isolated from other land masses. Of this group, 22% are recognised as threatened.

The Invertebrates make up the majority of species in New Zealand. Some 20,000 species out of a total of 23,000 are thought to be insects. The limited dispersal power of invertebrates (across seas) has tended to reduce the relation to Australian species. Some groups of invertebrates, such as those found in freshwater, have yet to be studied to determine if any are endangered. This is the largest group of animals in New Zealand. The terrestrial insects have 0.2% of their species endangered. The land snails have 11% of their species endangered.

The Plants of New Zealand are represented by few species relative to similar land masses. This again reflects New Zealand's isolation. There are more endangered plants than any other group. However, 5% of the plant species are known to be threatened. Mosses and liverworts have not been studied to discover which are threatened.

This means that the number of threatened species in a group will not equal the percentage threatened. The number of endangered species (300) should also be compared to the total number of species in New Zealand (23,000).

People have many reasons for wanting to save species. This includes ethical motivations. It includes a desire to preserve aesthetic species. The genes or ecological role a species has also motivates people. The number of these reasons makes it difficult to design a conservation strategy. As the player, your concern is to achieve a mix that you think is best. The interests of others or your attitude to the environment can play a role in this.

3 Rules

3.1 The first step in this game is setting up the board. This depends on the scenario you are playing. The description of each scenario will specify which counters are placed in which cells. Counters are initially placed with their faces up. More than one counter can be present in a single cell.

3.2 The second step is to set your priorities in the game. You must select *four* high priority species and *two* low priority species. To indicate whether a counter is a high priority, place a red cap on the counter. To indicate whether a species is a low priority, place a yellow cap on the counter.

3.2.1 **Record** the separate scores for your two priority groups. This is done by summing the value of the cells the species start in. This is termed the initial value. You will produce two initial values. One will be for the low priority group and one for the high priority group.

EXAMPLE- you select as high priority species counters in B2 (12), B5 (9), B6 (8) and C2 (6). The initial value, for this group, would be 35. Your low priority species are in cells A4 (16) and A5 (15). The initial value for the low priority group would be 31.

After the counters are on the board and the caps are placed the game can commence.

3.3 The game has three turns, which represent about ten years in real life. Each turn has three distinct phases.

3.4 *THE BUDGET PHASE-* The player allocates the budget permitted by the scenario to the species available. This is recorded on the budget sheet. Note that the budget sheet is organised into ecosystems. The position of a counter on the board is not related to the ecosystem it comes from.

A calculator will make it easy to keep track of your budget.

Note that species that are "Facing Imminent Extinction" are more costly to save than those that are "Highly Endangered". Further, those that are "Highly Endangered" are more costly than those that are "Endangered".

3.4.1 No part of your budget may be allocated to species in the *Extinct Species* cell or the *Recovered Species* cell. Once a counter is in either of these two cells it can not be moved again. That is, extinct species remain extinct and recovered species remain recovered irrespective of budgets and "Nature's moves" (*sec. 3.5.2*).

3.5 *THE OUTCOME PHASE-*

After the player has completed their budget they will need to determine the outcome.

3.5.1 For each ecosystem, the player rolls a percentage dice (*sec. 1.7*). The score should be noted on the budget sheet. The blank space under the ecosystem name is ideal.

3.5.2 The player should then consult *TABLE 1.* Cross reference the dice roll with the group the species belongs to. This will produce a score that is equal to "Nature's move".

EXAMPLE- you roll '62' with your percentage dice. Within that ecosystem you have two counters that are "Facing Imminent Extinction" and three counters that are "Highly Endangered". For the first two counters, the score would be '11' while for the last three, the score would be '5'. You may wish to record this result in the shaded column of the budget sheet.

3.5.3 Once you have a result, compare this to your budget allocation. Express this as a ratio of the "resources devoted to that species" to the "score generated by the dice".

EXAMPLE- in the case above, if for the first species you had devoted 20 resources, your ratio would be 20:11.

3.5.4 This ratio should be simplified. Use *TABLE 2* select the appropriate ratio.

NOTE- a ratio is always **rounded down** to the next lowest ratio if it fails to match the specified ratio.

EXAMPLE- In 3.5.3, the ratio 20:11 would be classed as 3:2. If the ratio was 22:11 it would be classed as 2:1. If the ratio was 16:11 it would be classed as 1:1. If the ratio was 17:11 it would be classed as 3:2.

3.5.5 The final ratio will determine whether and how far a counter moves. If your ratio is less than 1:3 it will only move three cells to the left. If your ratio is greater than 3:1 it will only move three cells to the right.

3.6 *MOVEMENT PHASE-* After you have determined how far a counter will move shift it the requisite number of cells. After you have moved it (which includes 'no change to position') then *turn the counter over.* This will eliminate any confusion you might have over whether a counter has yet been moved.

3.6.1 When the movement of a counter takes it beyond a group, the counter should be moved into another group. Groups are "joined" to other groups at

their opposite ends. For instance, if you have to move a counter past cell B6, in a rightward direction, its move would go through B6 then C1.

EXCEPTION- a counter that moves any distance left of cell A1 is put into the *Extinct Species* cell. A counter that moves any distance right of cell C6 is put into the *Recovered Species* cell.

3.6.2 After all the counters have been moved (that is, all have their backs facing the player) the turn is at an end. Turn the counters over and start the budget phase for the next turn. NOTE- the game has only three turns so do not continue past the third movement phase.

3.7 *CONSERVATION SCORE-*
Each scenario starts with an initial score (*sec. 1.3*; *4*). The scenario being played will specify what the starting score is. For instance, in Scenario One, the initial conservation score is 264. The score is determined by multiplying the number of counters in each cell by the value of that cell (*sec. 1.3*).

At the end of the game, the conservation score depends on the overall movement of the counters, adjusted for the priority species.

The first component of the final score is the overall movement. This is determined by summing for *ALL* the counters, the value of the cell they finished up in.

The second component, is the weighting given to the high priority species. Take the starting value of the High Priority species. Compute the final value of the High Priority species. Subtract from this *final value* your *starting value*. Hopefully the result will be a negative number!

EXAMPLE- the initial value of the four high priority counters was 34. At the end of the game the final score was 9. In consequence, you would subtract 25 from the overall movement score.

The third component is the discount given to the low priority species. Take the starting value of the low priority species. Compute the low priority species' final value. Subtract from your *starting value* the *final value.* Hopefully the result will be a negative number!

EXAMPLE- your initial value of the two low priority species was 31. At the end of the game the value for these species was 42 (note: this means they have become extinct). In consequence you would subtract 11 from your overall movement score.

Clearly you will be hoping to produce the lowest score possible.

EXAMPLE- you are playing Scenario One. You started with a score of 264. At the end of the game your overall movement score was 211. You adjust for your high priority species which was (-25) and your high priority species (-11). The final score for the game is 175.

TABLES

TABLE ONE
Score Generation

% ROLL	IMMINENT EXTINCTION (1)	HIGHLY ENDANGERED (2)	ENDANGERED (3)
01-05	28	15	7
06 - 10	25	13	6
11 - 15	22	11	5
16 - 20	20	10	5
21 - 25	18	9	5
26 - 30	16	8	4
31 - 35	16	8	4
36 - 40	16	8	4
41 - 45	15	7	3
46 - 50	14	7	3
51 - 55	13	6	3
56 - 60	12	6	3
61 - 65	11	5	2
66 - 70	10	5	2
71 - 75	9	4	2
76 - 80	8	4	2
81 - 85	7	3	1
86 - 90	6	3	1
91 - 95	5	2	1
96 - 100	4	2	1

(1) These are counters in cells "A1 to A6"
(2) These are counters in cells "B1 to B6"
(3) These are counters in cells "C1 to C6"

TABLES cont.

TABLE 2
Ratios and Cell Movement

BUDGET : SCORE RATIO	RESULT
1 : 3+ a	Left 3 cells
1 : 2 b	Left 2 cells
2 : 3 c	Left 1 cell
1 : 1 d	No change to position
3 : 2 e	Right 1 cell
2 : 1 f	Right 2 cells
3+ : 1 g	Right 3 cells

Notes: (Decimal equivalent to ranges)

a result greater than or equal to '0' but less than '0.5'
b result greater than or equal to '0.5' but less than '0.66667'
c result greater than or equal to '0.66667' but less than '1'
d result greater than or equal to '1' but less than '1.5'
e result greater than or equal to '1.5' but less than '2'
f result greater than or equal to '2' but less than 3
g result greater than or equal to '3'

Scenario One:

Budget: **200 per *TURN***
Initial Conservation Score: **264**

Set Up:

Cells	**Counters**
A3	B5
A4	P2 I5 R2
A5	P3 P5 I1
B2	P1 I2 B1
B3	P6 B6 R1
B4	P4 I7 F1
B5	I3 B3 R3
C2	P7 I6 B2
C3	I4 B4

5 Advice on Play

The goal of the game is to achieve the lowest score possible. However, there is **no single combination** of species that will achieve this. This means the player is free to choose any combination they like. It is acceptable to weight birds ahead of invertebrates or to weight genetic diversity ahead of ecological role.

Do not be afraid of rearranging your budget if you think a better result can be achieved. There is enough space on the budget sheet to make one or two revisions.

The allocation of funds is critical. This is because it is the one thing you have the most control over. As a very rough yard-stick, allocate 5 per 'C group' counter, 10 per 'B group' counter and 20 per 'A group' counter.

Spending more on species you especially value is a good strategy. This is a result of the *ratio system*. An extra resource or two devoted to your favourite species will often increase the rate at which it moves.

For instance, say you had two counters and 22 left in the budget. If you simply gave both 11 each, then either both would move or neither would move. Say your score was 8. Both counters would be stationary. Yet, if you had allocated 12 to the first counter and 10 to the second, the first counter would have moved. The budget is the same, but the net move is greater. This would be a very good outcome if the first counter was a high priority species.

HINT- when allocating your budget, it is useful to start with the top two or three species you want to save. Keep working out you budget in terms of these groups until you have exhausted your budget. It is a good idea to consult *TABLE 1* when allocating your budget.

Watch the progress of your counters. This will give you feedback from your choices. You may be unduly optimistic or pessimistic about the chances of some of the species.

To free up funds there are only two approaches you can use. These are to push some counters into the *Recovered Species* cell or abandon others. Either way this gives you more funds to spread among the other species. You should find it difficult to "save" all the species on the board. Be prepared to make some hard decisions.

APPENDIX TWO

COUNTER LIST-

Matrix Format

Key: a_1 = position on board; a_2 = genetic rank; a_3 = ecological rank

Taxonomic Group **Plants**

$$Plant\ 1 = \begin{bmatrix} 12 \\ I \\ I \end{bmatrix} \quad Plant\ 2 = \begin{bmatrix} 16 \\ II \\ I \end{bmatrix} \quad Plant\ 3 = \begin{bmatrix} 15 \\ II \\ II \end{bmatrix}$$

$$Plant\ 4 = \begin{bmatrix} 10 \\ II \\ II \end{bmatrix} \quad Plant\ 5 = \begin{bmatrix} 15 \\ I \\ II \end{bmatrix} \quad Plant\ 6 = \begin{bmatrix} 11 \\ III \\ II \end{bmatrix}$$

$$Plant\ 7 = \begin{bmatrix} 6 \\ II \\ III \end{bmatrix}$$

Taxonomic Group **Birds**

$$Bird\ 1 = \begin{bmatrix} 12 \\ I \\ I \end{bmatrix} \quad Bird\ 2 = \begin{bmatrix} 6 \\ II \\ I \end{bmatrix} \quad Bird\ 3 = \begin{bmatrix} 9 \\ II \\ II \end{bmatrix}$$

$$Bird\ 4 = \begin{bmatrix} 5 \\ II \\ II \end{bmatrix} \quad Bird\ 5 = \begin{bmatrix} 17 \\ I \\ II \end{bmatrix} \quad Bird\ 6 = \begin{bmatrix} 11 \\ III \\ II \end{bmatrix}$$

Taxonomic Group **Reptiles**

$$Reptile\ 1 = \begin{bmatrix} 11 \\ II \\ II \end{bmatrix} \quad Reptile\ 2 = \begin{bmatrix} 16 \\ II \\ I \end{bmatrix} \quad Reptile\ 3 = \begin{bmatrix} 9 \\ III \\ III \end{bmatrix}$$

Taxonomic Group **Fish**

$$Fish\ 1 = \begin{bmatrix} 10 \\ II \\ II \end{bmatrix}$$

Taxonomic Group **Invertebrates**

$$Invertebrate\ 1 = \begin{bmatrix} 15 \\ II \\ II \end{bmatrix} \quad Invertebrate\ 2 = \begin{bmatrix} 12 \\ II \\ I \end{bmatrix}$$

$$Invertebrate\ 3 = \begin{bmatrix} 9 \\ III \\ III \end{bmatrix} \quad Invertebrate\ 4 = \begin{bmatrix} 5 \\ III \\ II \end{bmatrix}$$

$$Invertebrate\ 5 = \begin{bmatrix} 16 \\ I \\ I \end{bmatrix} \quad Invertebrate\ 6 = \begin{bmatrix} 6 \\ II \\ I \end{bmatrix}$$

$$Invertebrate\ 7 = \begin{bmatrix} 10 \\ II \\ II \end{bmatrix}$$

APPENDIX THREE

NZ EXTINCT AND THREATENED SPECIES

A Extinctions in New Zealand since Human Settlement

Birds

Anomalopteryx didiformis	Little Bush Moa
Megalapteryx didinus	Upland Moa
Pachyornis elephantopus	Heavy-footed Moa
P. australis	Crested Moa
P. mappini	Mappin's Moa
Emeus crassus	Eastern Moa
Euryapteryx geranoides	Stout-legged Moa
E. curtus	Coastal Moa
Dinornis struthoides	Slender Bush Moa
D. novaezealandiae	Large Bush Moa
D. giganteus	Giant Moa
Pelecanus novaezealandiae	New Zealand Pelican
Cygnus sumnerensis	New Zealand Swan
Cnemiornis calcitrans	South Island Goose
C. gracilis	North Island Goose
Pachyanas chathamica	Chatham Island Duck
Euryanas finschi	Finsch's Duck
Malacorhynchus scarletti	Scarlett's Duck
Oxyura australis	Blue-billed Duck[1]
Biziura delautouri	de Lautour's Duck
Circus eylesi	Eyles' Harrier
Haliaeetus australis	Chatham Island Sea-eagle
Harpagornis moorei	New Zealand Eagle
Capellirallus karamu	Snipe-rail
Diaphorapteryx hawkinsi	Giant Chatham Island Rail
Gallinula hodgeni	Hodgen's Rail
Porphyrio mantelli mantelli	North Island Takahe
Fulica chathamensis prisca	New Zealand Coot
F. chathamensis chathamensis	Chatham Island Coot
Aptornis otidiformis otidiformis	North Island Adzebill
A. otidiformis defossor	South Island Adzebill
Coenocorypha chathamica	Giant Chatham Island Snipe
Aegotheles novaezealandiae	New Zealand Owlet-nightjar
Pachyplichas yaldwyni	Yaldwyn's Wren
P. jagmi	Grant-Mackie's Wren
Palaeocorax moriorum	New Zealand Crow

[1] This species is extant in Australia, but is extinct in New Zealand. It is known from subfossils collected in Hawke's Bay.

Ixobrychus novaezealandiae	New Zealand Little Bittern
Merganser australis	Auckland Island Merganser
Coturnix novaezealandiae novazealndiae	New Zealand Quail (Koreke)[2]
Rallus philippensis dieffenbachii	Dieffenbach's Rail
R. modestus	Chatham island Rail
Coenocorypha aucklandica iredalei	Stewart Island Snipe
C. aucklandica barrierensis	Little Barrier Snipe
Sceloglaux albifacies	Laughing Owl
Xenicus longipes	Bush Wren
Traversia lyalli	Stephens Island Wren
Bowdleria rufescens	Chatham Island Fernbird
Anthornis melanura melanocephala	Chatham Island Bellbird
Callaeas cinera cinera	South Island Kokako
Heteralocha acutirostris	Huia
Turnagra capensis tanagra	North Island Piopio
T. capensis capensis	South Island Piopio

B Endangered Species of New Zealand (As listed 1992)

Plants

Acaena pallida
Acaena rorida
Aciphylla dieffenbachii
Aciphylla traversii
Asplenium pauperequitum
Astelia chathamica
Atriplex billardierei
Atriplex cinerea
Australopyrum sp. nov.
Baumea complanata
Boehmeria australis var. *dealbata*
Brachyglottis pentacopa
Brachyglottis perdicioides
Calystegia marginata
Carex inopinata
Carmichaelia kirkii
Cheesemania 'Chalk Range'
Chionochloa spiralis
Chordospartium muritai

[2] The New Zealand Quail belongs to the same species as the Stubble Quail (*Coturnix novaezealandiae pectoralis*). This species is extant in Australia.

Chordospratium stevensonii
Christella 'basket'
Christella 'N.Z. dentata'
Clianthus puniceus
Coprosma spathulata sub. sp. 'Surville'
Coprosma waima
Cortaderia turbaria
Corybas carsei
Cyathea kermadecensis
Dactylanthus taylorii
Davallia 'Puketi'
Eleocharis neozelandica
Elingamita johnsonii
Embergeria grandifolia
Euphorbia glauca
Gunnera hamiltonii
Hebe acutifolia
Hebe armstrongii
Hebe barkeri
Hebe breviracemosa
Hebe cupressoides
Hebe elliptica var. *crassiflora*
Hebe Adamsii
Hebe speciosa
Helichrysum dimorphum
Hydatella inconspicua
Iti lacustris
Kunzea sinclairii
Lepidum banksii
Lepidum kirkii
Lepidum oleraceum
Lepidum sisymbriodes matau
Leptinella dioica monoica
Leptinella featherstonii
Leptinella nana
Leptinella rotundata
Lycopodium serpentium
Macropiper 'Three Kings'
Melicytus 'Egmont'
Metrosideros barletti
Muehlenbeckia astonii
Myosurus minimus novae-zelandiae
Myosotidium hortensia
Myosotis albosericea
Myosotis colensoi
Myriophyllum robustum
Olearia 'Glenhope'

Olearia hectorii sub. sp. 'North Island'
Olearia hectorii sub. sp. 'South Island'
Olearia pachyphylla
Olearia 'Waima'
Pennantia baylisiana
Pimelea 'Three Kings'
Pittosporum dallii
Pittosporum obcordatum
Pittosporum 'Surville'
Pittosporum turneri
Plantago spathulata picta
Pomaderris hamiltonii
Pomaderris phylicifolia var. *polifolia*
Prasophyllum sp. aff. *patens*
Pseudopanax gilliesii
Pterostylis micromega
Ranunculus viridis
Rhopalostylis 'Chathams'
Rorippa divaricata
Scutellaria novae-zelandiae
Sebaea ovata
Stilbocarpa lyallii
Stilbocarpa robusta
Tecomanthe speciosa
Thelymitra 'Ahipara'
Thelymitra malvina
Thelymitra matthewsii
Thelypteris confluens
Utricularia lateriflora
'Xit xit'- unknown species

Birds
Anarhynchus frontalis
Anas aucklandica aucklandica
Anas aucklandica chlorotis
Anas aucklandica nesiotis
Apteryx australis australis
Apteryx australis lawryi
Apteryx australis mantelli
Apteryx haastii
Apteryx owenii
Bowdleria punctata caudata
Bowdleria punctata stewartiana
Bowdleria punctata wilsoni
Callaeas cinerea wilsoni
Charadrius bicinctus bicinctus
Charadrius bicinctus exilis

Charadrius obscurus
Coenocorypha aucklandica aucklandica
Coenocorypha aucklandica huegeli
Coenocorypha aucklandica meinertzhagenae
Coenocorypha pusilla
Cyanoramphus auriceps auriceps
Cyanoramphus auriceps forbesi
Cyanoramphus malherbi
Cyanoramphus unicolor
Diomedea eremita
Diomedea exulans
Eudyptes pachyrhynchus
Eudyptes robustus
Falco novaeseelandiae
Gallirallus australis australis
Gallirallus australis hectori
Gallirallus australis scotti
Haematopus chathamensis
Haematopus unicolor
Hemiphaga novaeseelandiae chathamensis
Hemiphaga novaeseelandiae novaseelandiae
Himantopus novaezealandiae
Hymenolaimus malacorhynchos
Leucocarbo carunculatus
Leucocarbo chalconotus
Leucocarbo onslowi
Megadyptes antipodes
Mohoua ochrocephala
Nestor meridionalis meridionalis
Nestor meridionalis septentrionalis
Nestor notabilis
Notiomystis cincta
Pelagodrama marina albiclunis
Petroica macrocephala chathamensis
Petroica macrocephala dannefaerdi
Petroica traversi
Philesturnus carunculatus carunculatus
Philesturnus carunculatus rufusater
Podiceps rufopectus
Porphyrio mantelli hochstetteri
Procellaria parkinsoni
Procellaria westlandica
Prosthemadera novaeseelandiae chathamensis
Pterodroma axillaris
Pterodroma cervicalis cervicalis
Pterodroma magentae
Pterodroma pycrofti

Puffinus bulleri
Puffinus huttoni
Rallus pectoralis muelleri
Sterna albostriata
Sterna nereis davisae
Strigops habroptilus
Thinornis novaeseelandiae
Xenicus gilviventris

Fish
Galaxias argenteus
Galaxias brevipinnis
Galaxias fasciatus
Galaxias gracillis
Galaxias postvectis
Geotria australis
Gobiomorphus hubbsi
Neochanna apoda
Neochanna burrowsius
Neochanna diversus

Reptiles
Cyclodina alani
Cyclodina macgregori
Cyclodina 'Mokohinau Island'
Cyclodina 'Poor Knights'
Cyclodina whitakeri
Hoplodactylus chrysosireticus
Hoplodactylus kahutarae
Hoplodactylus rakiurae
Hoplodactylus stephensi
Leiolopisma grande
Leiolopisma homalonotum
Leiolopisma 'long-toe'
Leiolopisma microlepis
Leiolopisma nigriplantare nigriplantare
Leiolopisma notosaurus
Leiolopisma otagense form otagense
Leiolopisma otagense form waimatense
Leiolopisma 'small ear-holed'
Leiolopisma striatum
Sphenodon guntheri
Sphenodon punctatus
Sphenodon punctatus punctatus

Mammals
Cephalorhynchus hectori

Chalinolobus tuberculata
Mystacina tuberculata aupourica
Mystacina tuberculata rhyacobia
Mystacina tuberculata tuberculata
Phocarctos hookeri

Frogs
Leiopelma hamiltoni 'Stephens'
Leiopelma hamiltoni 'Maud'
Leiopelma archeyi
Leiopelma hochstetteri

Invertebrates
Amychus candezi
Amychus granulatus
Anagotus fairburni
Anagotus stephenensis
Anagotus turbotti
Asaphodes stinaria
Deinacrida fallai
Deinacrida heteracantha
Deinacrida 'Mahoenui'
Deinacrida rugosa/parva
Deinacrida tibiospina
Dorcus auriculanus
Dorcus helmsii
Dorcus ithaginus
Dorcus 'North Coromandel'
Hadramphus spinipennis
Hadramphus stilbocarpae
'Hemiandrus' sp.
'Hemidandrus monstrosus'
Hemideina ricta
Heterexis seticostatus
Lyperobius huttoni
Mecodema chiltoni
Megadromus sp.
Oclandius laeviusculus
Parphanta busbyi busbyi
Parphanta busbyi watti
Placostylus ambagiosus ambagiosus
Placostylus ambagiosus annectens
Placostylus ambagiosus consobrinus
Placostylus ambagiosus keenorum
Placostylus ambagiosus lesleyae
Placostylus ambagiosus michiei

Placostylus ambagiosus pandora
Placostylus ambagiosus paraspiritus
Placostylus ambagiosus watti
Placostylus ambagiosus whareana
Placostylus bollonsi bollonsi
Placostylus bollonsi caparatus
Placostylus hongii
Powelliphanta annectens
Powelliphanta 'Buller River'
Powelliphanta 'Egmont'
Powelliphanta fiordlandica
Powelliphanta gilliesi aurea
Powelliphanta gilliesi brunnea
Powelliphanta gilliesi compta
Powelliphanta gilliesi fallax
Powelliphanta gilliesi gilliesi
Powelliphanta gilliesi jamesoni
Powelliphanta gilliesi kahurangica
Powelliphanta gilliesi montana
Powelliphanta gilliesi subfusca
Powelliphanta hochstetteri anatokiensis
Powelliphanta hochstetteri bicolor
Powelliphanta hochstetteri consobrina
Powelliphanta hochstetteri hochstetteri
Powelliphanta hochstetteri obscura
Powelliphanta lignaria johnstoni
Powelliphanta lignaria lignaria
Powelliphanta lignaria lusca
Powelliphanta lignaria oconnori
Powelliphanta lignaria ruforadiata
Powelliphanta lignaria unicolorata
Powelliphanta marchanti
Powelliphanta rossiana fletcheri
Powelliphanta rossiana gagei
Powelliphanta rossiana patrickensis
Powelliphanta spedenti lateumbilicata
Powelliphanta spedenti spedenti
Powelliphanta superba
Powelliphanta traversi otakia
Powelliphanta traversi traversi
Powelliphanta 'Urewera'
Powelliphanta 'Wolf River'
Prodontria bicolorata
Prodontria lewisi
Prodontria modesta
Spelungula cavernicola
Triamescaptor aotea

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