INVESTING IN BIOLOGICAL DIVERSITY: ECONOMIC VALUATION AND PRIORITIES FOR DEVELOPMENT

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Abstract

By all informed scientific accounts the world's biological diversity is currently in a critical condition. Biodiversity is vital for the continued existence of the global biosphere and, by extension, human wellbeing and development. It is inconceivable that a discipline predicated on the issues of scarcity and choice has nothing to contribute in terms of an understanding of either the causes and consequences of biodiversity loss, or in proposing solutions to the crisis. This thesis examines some of the economic parameters of the issue. Alongside the acknowledged root problems of market and institutional failure lies the question of economic valuation. Valuation of biodiversity puts conservation on a more level playing field with the economic forces which threaten its demise. Provided economic values can be appropriated (i.e. converted to flows of real economic resources) it becomes worthwhile for countries to invest in valuable biological assets. But the practice of economic valuation and the quantification of biodiversity are in their infancy and the complexity of the latter hinders the precise application of the former. Much of this thesis focuses on the use and development of the contingent valuation method (CV) as a flexible approach to valuing biodiversity. The method has a useful role to play in resource allocation, and, for valuing biological resources. Faced by the irreducible complexity of life which is the essence of biodiversity, CV does have its limitations. It is possible to conclude that existing valuation methods are a vital part of a "holding operation" alongside other surrogate approaches to setting priorities for global conservation. Nevertheless, the development of an interface between economic (preference-based) values, and biological values, which together can comprehensively inform conservation decisions remains the objective for the future.

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Declaration

Parts of this thesis have appeared or are due to appear in published form and the author claims at least fifty percent of the work stated as jointly produced. Parts of chapter one are partially adapted from Pearce and Moran (1994) and Moran and Pearce (1997). Parts of chapter two are published in Moran and Pearce (1997 in press). Chapters three and four are forthcoming in Moran and Steffens-Moraes (1997 a,b,c). Parts of chapter five are published in Moran (1994). The travel cost element of this chapter is drawn from a paper by Brown *et al* (1995) for which the author conducted the data preparation and estimation. Chapter six has been published in adapted form in Moran *et al* (1996; 1997).

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Thanks are due for the collaborative efforts of Andre Steffens-Moraes of the Centro de Pesquisa Agropecuaria do Pantanal for the study presented in chapter four. Both researchers acknowledge the financial assistance of the British Council (Brazilia) and the helpful assistance of the Companhia Independente de Policia Florestal do Estado do Mato Grosso do Sul - (CIPFlo/MS). Much of Chapter five was inspired by Gardner Brown of the University of Washington who I also wish to thank.

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Chapter one

Biodiversity Loss, Conservation and Economic Valuation

1.1 Introduction

The world's biological diversity is being lost at an unprecedented rate (Pimm *et al* 1995). At the dawn of the 80s, E.O. Wilson memorably put the irreparable loss of genetic and species diversity above other problems[1] as the "folly our descendants are least likely to forgive us". In the words of Norman Myers, a "mass-extinction episode" is under way, while similar apocalyptic scenarios have been painted by countless other distinguished conservation biologists and ecologists. With the benefit of hindsight what makes this episode so bad is that we are, more than ever before, aware of what we are doing. This implies a deliberate choice to continue eroding the globe's biological patrimony. In economic terms this suggests an immediate conservation-development trade-off which has to be rationalised, perhaps in terms of economic or other values. But the near hysterical and forthright tone belies a great deal of uncertainty and conjecture about all aspects of the biodiversity problem. What is biodiversity? Where is it, and how much should we keep? More immediately, while the effects of its decline do not yet impinge on the majority of every day affairs, what is the incentive for allocating scarce resources to conservation? In a nutshell, these questions sum up the essence of the biodiversity problem. Answering them is a multi-disciplinary task in which economics has a role. This chapter sets out some of the basic issues to be dealt with in subsequent chapters of this thesis. The first section outlines the meaning of biodiversity and offers a discussion of the evidence of biodiversity loss. The second section reviews the causal factors (of loss) and the conservation - development nexus. This leads directly to the last section introducing valuation and setting the scene for remaining chapters.

The contribution of this thesis is to examine the extent to which economics can contribute to the area of biological conservation. In considering the costs and benefits of conservation versus loss the approach is unashamedly utilitarian. Many of the more mundane conclusions arising from this analysis are nevertheless frequently overlooked in the growing literature on the economics of conservation. These can be summarised in the following terms. We do not and probably never will know enough about biodiversity to make decisions which are both scientifically and economically

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[1] The other possibilities according to Wilson (1980) being catastrophes such as energy depletion, economic collapse, limited nuclear war and conquest by totalitarian government.

efficient. Given scientific advances, a cardinal approach to biodiversity is a desirable ultimate objective. But while this ideal is constrained by the best efforts of scientific endeavour, extinction occurs and the only rational response to the crisis involves identifying and correcting proximate and underlying causes and associated market and institutional failures. Yet even here there is a knowledge hurdle in the form of imprecise preferences and potential inconsistency between a utilitarian and anything remotely approximating a 'preference-free' scientific approach. Elements of both agendas cab be evaluated and their prescriptions compared, but the remaining ad hoc approach essentially boils down to the maximisation of any habitats at minimum cost. Even within such a crude objective economic approaches have a contribution to make.

1.2 What is Biodiversity?

Biodiversity may be described in terms of genes, species, and ecosystems, corresponding to three fundamental and hierarchically-related levels of biological organisation.

Genetic Diversity

Genetic diversity is the sum of genetic information contained in the genes of individuals of plants, animals and micro-organisms. Each species is the repository of an immense amount of genetic information. The number of genes range from about 1000 in bacteria, up to 400 000 or more in many flowering plants. Homo sapiens has approximately 200,000. Each species is made up of many organisms, and virtually no two members of the same species are genetically identical. This means for example that even if an endangered species is saved from extinction, it will probably have lost much of its internal diversity. When the populations are allowed to expand again, they will be more genetically uniform than their ancestral populations. For example, the bison herds of today are biologically not the same in terms of their genetic diversity, as the bison herds of the early 18th century (McClenagham *et al*, 1990).

Population geneticists have developed mathematical formulae to express a genetically effective population size. These explain the genetic effects on populations which have passed through a 'bottleneck' of a small population size, such as the North American bison, or African cheetah (WCMC, 1992). The resultant inbreeding may have a number of detrimental effects such as lowered fertility and increased susceptibility to disease. This is termed 'inbreeding depression'. The effects of small population size depend on the breeding system of the species and the duration of the bottleneck. If the bottleneck lasts for many generations, or population recovery is very slow, a great deal of variation can be lost. The converse, 'outbreeding depression', occurs when species become genetically differentiated across their range, and then individuals from different parts of the range breed.

Genetic differentiation within species occurs as a result of either sexual reproduction, in which genetic differences from individuals may be combined in their offspring to produce new combinations of genes, or from mutations which cause changes in the DNA.

The significance of genetic diversity is often highlighted with reference to global agriculture and food security. This stresses the reliance of the majority of the world's human population on a small number of staple food species, which in turn rely on supply of genes from their wild relatives to supply new characteristics, for example to improve resistance to pests and diseases (Cooper *et al*, 1992).

Species Diversity

Species are regarded as populations within which gene flow occurs under natural conditions. Within a species, all normal individuals are capable of breeding with the other individuals of the opposite sex belonging to the same species, or at least they are capable of being genetically linked with them through chains of other breeding individuals. By definition, members of one species do not breed freely with members of other species. Although this definition works well for many animal and plant species, it is more difficult to delineate species in populations where hybridisation, or self-fertilisation or parthenogenesis occur. Arbitrary divisions must be made, and indeed this is an area where scientists often disagree.

New species may be established through the process of polyploidy, the multiplication of the number of gene bearing chromosomes, or more commonly, as a result of geographic speciation. This is the process by which isolated populations diverge by evolution as a result of being subjected to different environmental conditions. Over a long period of time, differences between populations may become great enough to reduce interbreeding and eventually populations may be able to co-exist as newly formed, separate species.

Within the hierarchical system used by scientists to classify organisms, species represent the lowest rung on this ladder of classification. In ascending order, the main categories, or taxa, of living things are: species, genus, family, order, class, phylum, kingdom.

We do not know the true number of species on earth, *even to the nearest order of magnitude* (see below). The best catalogued groups include vertebrates and flowering plants, with other groups relatively under-researched, for example, lichens, bacteria, fungi and roundworms. Likewise, some habitats are better researched than others, and coral reefs, deep ocean floor and tropical soils are not well studied. This lack of knowledge has considerable implications for the economics of biodiversity conservation, particularly in defining priorities for cost-effective conservation interventions.

The single most obvious pattern in the global distribution of species is that overall species richness increases with decreasing latitude. Not only does this apply as a general rule, it also holds within the great majority of higher taxa, at order level or higher. However, this overall pattern masks a large number of minor trends. Species richness in particular taxonomic groups, or in particular habitats, may show no significant latitudinal variation, or may actually decrease with decreasing latitudes. In addition, in terrestrial ecosystems, diversity generally decreases with increasing altitude. This phenomenon is most apparent at extremes of altitude, with the highest regions at all latitudes having very low species diversity (although these areas also tend to be of limited size, which may be one factor resulting in lower species numbers). In terms of marine systems, depth is the analogue of altitude in terrestrial systems and biodiversity tends to be negatively correlated with depth. Gradients and changes in species richness are also noticeably correlated to precipitation, nutrient levels, and salinity, as well as other climatic variations and available energy.

Ecosystem Diversity

Ecosystem diversity relates to the variety of habitats, biotic communities and ecological processes in the biosphere as well as the diversity within ecosystems. Diversity can be described at a number of different levels and scales. Functional diversity is the relative abundance of functionally different kinds of organisms. Community diversity is the number sizes and spatial distribution of communities, and is sometimes referred to as patchiness. Landscape diversity is the diversity of scales of patchiness.

No simple relationship exists between the diversity of an ecosystem and ecological processes such as productivity, hydrology, and soil generation. Neither does diversity correlate neatly with ecosystem stability; its resistance to disturbance and its speed of recovery. There is no simple relationship within any ecosystem between a change in its diversity and the resulting change in the system's processes. For example, the loss of a species from a particular area or region (local extinction or extirpation) may have little or no effect on net primary productivity if competitors take its place in the community. The converse may be true in other cases. For example, if herbivores such as zebra and wildebeest are removed from the African savanna, net primary productivity of the ecosystem decreases.

Despite these anomalies, Reid and Miller (1989) suggest six general rules of ecosystem dynamics which link environmental changes, biodiversity and ecosystem processes.

1. The mix of species making up communities and ecosystems changes continually.
2. Species diversity increases as environmental heterogeneity or the patchiness of a habitat does, but increasing patchiness does not necessarily result in increased species richness.
3. Habitat patchiness influences not only the composition of species in an ecosystem, but also the interactions among species.
4. Periodic disturbances play an important role in creating the patchy environments that foster high species richness. They help to keep an array of habitat patches in various successional states.
5. Both size and isolation of habitat patches can influence species richness, as can the extent of the transition zones between habitats. These transitional zones, or 'ecotones', support species which would not occur in continuous habitats. In temperate zones, ecotones are often more species rich than continuous habitats, although the reverse may be true in tropical forests.
6. Certain species have disproportionate influences on the characteristics of an ecosystem. These include keystone species, whose loss would transform or undermine the ecological processes or fundamentally change the species composition of the community.

The discussion has shown how biodiversity is a very complex and all embracing concept which can be interpreted and analyzed on a number of levels and scales. The debate about the appropriate scale of conservation has some economic significance since ecosystem conservation provides more obvious targets which are also amenable to cost-benefit studies. The next section examines some approaches to measuring these concepts.

1.3 Measurement of Biodiversity

Following on from the various scales, assessing the extent of biological life on earth is a sobering task as the numbers quickly become mind-boggling. This is hardly surprising since in the extreme biodiversity can be characterised as the sheer irreducible complexity of life. From an economic perspective such an all-embracing summation places a potentially insurmountable lack of correspondence between perception/preference-based values and those dictated by biodiversity measures. In other words, while people can perceive some exact traits and not others their preferences will be bounded accordingly.

It is thought that there are somewhere between 5 to 80 million species on earth. A conservative estimate is 13-14 million of which only 1.75 million have been described some in only rudimentary detail (UNEP 1995; Stork 1993) with little known of their historical relationships biological characteristics or distributions within the earth's habitats or ecosystems. Below the species level the figures are even more awesome with estimates of genetic variability ranging from 10^9 (Pellew 1995) to as high as 10^{29} (Vane-Wright [1996] *personal communication*), 99% of which we almost certainly know nothing.

A better understanding of biodiversity can be obtained when we examine exactly what we measure in order to assess biological diversity. However, this also serves to highlight further the range of interpretations, and the importance placed on different hierarchical levels of biodiversity by scholars of different disciplines, and by policy makers. Reid *et al* (1992) have commented that there is even now no clear consensus about how biodiversity should be measured. Indeed, debates on the measurement of biodiversity have filled a substantial part of the ecological literature since the 1950s. This lack of consensus also has important implications for the economics of biodiversity conservation. At its most basic level, any measure of cost-effectiveness used to guide investments in conservation must have some index or set of indices of biodiversity change. In the following sections, some aspects of measurement of biodiversity are examined, distinguishing the same components of biodiversity; genetic diversity, species diversity, and ecosystem diversity. The measurement problem will also be dealt with in more detail in chapter two.

Measurement of Genetic Diversity

The analysis and conceptualisation of differences within and among populations is in principle identical regardless of whether we are considering a 'population' to be a local collection of individuals, geographical race, subspecies, species, or higher taxonomic group. Genetic differences can be measured in terms of phenotypic traits, allelic frequencies, or DNA sequences.

Phenetic diversity is based on measures of phenotypes, individuals which share the same characteristics. This method avoids examination of the underlying allelic structure (see below). It is usually concerned with measurement of the variance of a particular trait, and often involves readily measurable morphological and physiological characteristics. Phenetic traits can be easily measured, and their ecological or practical utility is either obvious or can be readily inferred. However, their genetic basis is often difficult to assess, and standardised comparisons are difficult when populations or taxa are measured for qualitatively different traits.

Allelic diversity: The same gene can exist in a number of variants and these variants are called alleles. Measures of allelic diversity require knowledge of the allelic composition at individual loci. This information is generally obtained using protein electrophoresis, which analyses the migration of enzymes under the influence of electric field. Allelic diversity may be measured at the individual level, or at the population level. In general, the more alleles, the more equitable their frequencies, and the more loci that are polymorphic, the greater the genetic diversity. Average expected heterozygousity (the probability that two alleles sampled at random will be different) is commonly used as an overall measure. A number of different indices and coefficients can be applied to the measurements to assess genetic distance (see Antonovic [1990]). The detection of allelic variation by electrophoresis has the advantage that it can be precisely quantified to provide comparative measures of genetic variation. However, the disadvantages are that it may not be representative of variation in the genome as a whole, and does not take account of functional significance or selective importance of particular alleles.

Sequence variation: A portion of DNA is sequenced using the polymerase chain reaction technique (PCR). This technique means that only a very small amount of material, perhaps one cell, is required to obtain the DNA sequence data, so that only a drop of blood or single hair is required as a sample. Closely related species may share 95 percent or more of their nuclear DNA sequences, implying a great similarity in the overall genetic information.

Measurement of Species Diversity

Species diversity is a function of the distribution and abundance of species. Often, species richness - the number of species within a region or given area - is used almost synonymously with species diversity. However, technically, species diversity includes some consideration of evenness of species abundances. Let us first consider species richness as a proxy measure of species diversity.

In its ideal form, species richness would consist of a complete catalogue of all species occurring in the area under consideration, but this is not usually possible, unless it is a very small area. Species richness measures in practice therefore tend to based on samples. Such samples consist of a complete catalogue of all organisms within a taxa found in a particular area, or it may consist of a measure of species density in a given sample plot, or a numerical species richness defined as the number of species per specified number of individuals or biomass.

A more informative measure of diversity would also incorporate the 'relatedness' of the species in a fauna (Williams *et al* [1991], Reid *et al* [1992]). Using a measure of species richness might imply that a region containing many closely related species is as valuable as one containing a fractionally smaller number of distantly related or genealogically unrelated species. Alternative measures being developed augment species richness with measures of the degree of genealogical difference. Derived from cladistic (family tree) methods, these measures include the weighting of close-to-root species, higher-taxon richness, spanning-tree length and taxonomic dispersion (Williams *et al* 1991). Close-to-root species and higher-taxon richness explicitly use polarity from the root of the tree to weight higher-ranking taxa or 'relic' species as distinct survivors of long-independent lineages and original conduits of genetic information. In contrast, spanning tree length and taxonomic dispersion are more general tree measures of sub-tree 'representativeness'. Polarity from the root of the tree is less important than the amount of the cladogram represented by a fauna or the choice of a fauna to evenly cover the diversity of subgroups found in the cladogram. There is considerable disagreement as to which measure best characterises the pattern of difference consistent with the popular concept of biodiversity, although considerable support for taxonomic dispersion as a method of selecting faunas which most evenly represent a variety of cladogram sub-groups. For the time being, difficulties in actual implementation of cladistic measures suggest reliance on cruder indicators of richness of genera or families for rapid assessment of species diversity. However the framework provided by this type of thinking does open up a potentially rich avenue of economic research which is revisited in chapter two.

Measurement of Community Diversity

Many environmentalists and ecologists put emphasis on conservation of biodiversity at the community level. There are a number of factors which make measurement and assessment of diversity at this level more nebulous and less clearly defined. Many different 'units' of diversity are involved at the supra-species level, including the pattern of habitats in the community, relative abundance of species, age structure of populations, patterns of communities on the landscape, trophic structure, and patch dynamics. At these levels, unambiguous boundaries delineating units of biodiversity do not exist.

By conserving biodiversity at the ecosystem level, not only are the constituent species preserved, but also the ecosystem functions and services protected. These include pollutant cycling, nutrient cycling, climate control, as well as non-consumptive recreation, scientific and aesthetic values (see for example, Norton and Ulanowicz [1992]).

Given the complexities of defining biodiversity at community or ecosystem level already described, there is a range of different approaches to measuring ecosystem diversity. As Reid *et al* [1992] explain, any number of community attributes are components of biodiversity and may deserve monitoring for specific objectives. There are several generic measures of community level diversity. These include biogeographical realms or provinces, based on the distribution of species, and ecoregions or ecozones, based on physical attributes such as soils and climate. These definitions may differ according to scale. For example, the world has been divided into biogeographical provinces, or more fine-grained classifications which may be more useful for policy-making. More policy orientated measures include the definition of 'hotspots', based on the number of endemic species, and 'megadiversity' states.

Some of these concepts will be discussed in the context of using indicators for assessing and monitoring biodiversity (chapters two and six). The following section looks the estimates of recent rates of species extinction. These estimates are subject to some controversy which bear on the urgency of global conservation spending.

1.4 The rate of biodiversity loss

The actual rate of diversity loss (or more correctly species loss), has been the subject of acrimonious debate between conservationists and a few contrarian scientists (see Mann 1991). The uncertainties raised in the debate have in some ways successfully attenuated the immediacy of mitigatory action.

Quantifying biodiversity loss raises two issues: at what rate is biodiversity disappearing? And, whatever the rate, why does it matter ? The latter question relates to the proximity of the ultimate economic consequences of the loss, and is deferred until later. A legitimate focus on the weight of non-economic (e.g. ethical) motives would also provide a dimension to the discussion of extinction although track record on stemming habitat loss and a blunt appeal to pragmatism will be the unapologetic caveat provided at the outset (see Pearce and Moran 1994). Thus, estimates of rates of biodiversity loss are very uncertain because knowledge about species is limited and because the remains of extinct species exist in only a limited number of cases. Thus inferences have to be made from past extinction rates based on fossil records, or on some assumption about the fate of existing threatened species, or about the relationship between species and land area. The last approach links species to land area and then uses records of historical land conversions to determine extinction rates. As previously mentioned the species-area relationship can only be determined by limited sampling. Nevertheless, this approach gives rise to predictions over the next century that the projected loss of species might be expected to be as high as 20 to 50% of the world's total which represents a rate between 1000 to 10000 times the historical rate of extinction (Wilson [1988]).

Table 1 Estimates of the Current Rates of Species Extinction.

Estimate of Loss	Basis	Source
33-50% of species by 2000	forest area loss	Lovejoy (1980)
50% of species by 2000	forest area loss	Ehrlich (1981)
25-30% of species in 21st century	forest area loss	Myers (1989)
33% of species in 21st century	forest area loss	Simberloff (1986)

WCMC (1992) and references

Pimm *et al* (1995) suggest recent extinction rates of around 20 to 200 species per 1 million species years. (For example, if a species lasts 1 million years, its extinction rate would be 1 per million species years). This rate is substantially above that in 'pre-human' times. Projections of species loss are fraught with difficulty, but Pimm *et al.* (1995) suggest that if all species listed today as being threatened become extinct in the next 100 years, then future extinction rates will exceed current rates by a factor of ten. While some authors doubt these orders of magnitude (some of them producing absurdly low figures - see Simon and Wildavsky (1995)) it is hard to doubt rapid rates of current extinction, whilst the case for accelerating rates in the future seems persuasive. Table 2 summarises some contrary views.

1.5 Biodiversity Loss: The Causal Factors

The economic theory of species extinction has its origins in the theory of the fishery (Gordon, 1954). Various models suggest two sets of factors may give rise to extinction. The first is the property rights regime.

Regimes may be 'open access', common property, single private owner, or single state owner. Open access differs from common property in that the former regime has no owners and hence no rules for restricting or managing access to the fishery[2]. Common property regimes involve sets of rules and regulations limiting access and catch rates, these rules being enforced by communal law or communal custom. Under single ownership, access is restricted to any outsider not part of the ownership regime, and the community in question is now a private organisation or the nation state.

Table 2	Species extinction: contrary views	
	Assumption	Criticisms
Habitat loss	Most predictions of species loss are based on using islands as a model	Mainland territories behave differently from islands - if original habitat is lost, species may escape into bordering areas. Data on habitat loss are frequently misleading, and do not allow for the function patch diversity and its importance as a species refuge.
Species-area curve	Current models of the relation between species and geographic area imply that an infinite increase in area implies an infinite increase in the number of species	Critics argue that in fact the curve levels off at its upper reaches. Therefore habitats on the upper part of the species-area curve can be reduced without substantial species loss.
The number of species	During the 1960s, researchers realized the incredible biological diversity of tropical forests and estimates of the number of species shot up leading Wilson and Ehrlich to posit that 100 million species may live on Earth.	In fact taxonomists have managed to name only 1.4 million species and the actual total is a matter of speculation. Catastrophical estimates of extinction are based in large part on species no one has ever seen.

Source: Adapted from Mann (1991)

Fairly self-evidently, the greater the restrictions on access the less the fishing effort applied to the fishery and hence the more likely it is that fish stocks will be large. With open access, however, effort is unconstrained until the newest new entrant finds that fish stocks have depleted to the point where his effort is only just rewarded by the extra revenues obtained from the sale of the fish: the 'zero rent' point. Zero rents provide the condition for the equilibrium amount of effort - see Figure

[2]As the E.U. Common Fisheries Policy shows, within common property regimes imperfect management may give rise to forms of open access exploitation.

1. In itself, then, open access does not generate extinction of the resource because further entry beyond the open access equilibrium does not occur. As long as the costs of catch are positive, some of the resource is left intact, though with low sustainable yields[3]. Nonetheless, population dynamics are ill understood and hence the risk of extinction is higher the lower are the stocks. It is quite possible, therefore, for open access to create the conditions in which catch levels give rise to stock levels that are below minimum viable sizes. This will be especially true in contexts where minimum viable size is influenced by family or group dynamics, as is the case for some land based animals. Removing critical family members may render the remaining population non-viable. Note also that, while common property regimes are far more likely to protect the resource against extinction, common property tends itself to be vulnerable to external influences, including human population growth which places pressure on the regime to continue managing the resource in the interests of the whole community.

Particular combinations of cost of effort and resource price will also make the open access situation more risky still. In terms of Figure 1, the initial open access equilibrium is at E_{OA}. Now suppose that technological change occurs such that the costs of effort are reduced substantially from TC to TC'. Suppose too that the resource attracts a higher price, so that, for the same level of effort, total revenue increases. The new open access equilibrium (at the intersection of the new total revenue and the reduced cost schedules) is now much closer to the zero stock level. This 'high price, low cost' combination appears to fit certain land-based species well, e.g. elephants. Ivory prices can be very high and the use of vehicles, high velocity rifles plus the frequently low probability times the fine (for getting caught) makes costs low.

The first cause of extinction on this model, then, is the property rights regime, with open access having the highest risk of extinction for the species, and common property having a far lower but non-zero level of risk. Species attracting exotic demand, such as for ivory, and being subject to advances in technological change which reduce effort costs are particularly at risk. As we shall see, specific policy actions to correct these situations automatically follow.

The second cause of extinction advanced in the Clark model (Clark 1973ab) arises in the single owner, profit-maximising context. Whereas extinction under open access can be regarded partly as

[3]Strictly speaking, what matters is price relative to the cost at zero stock size.

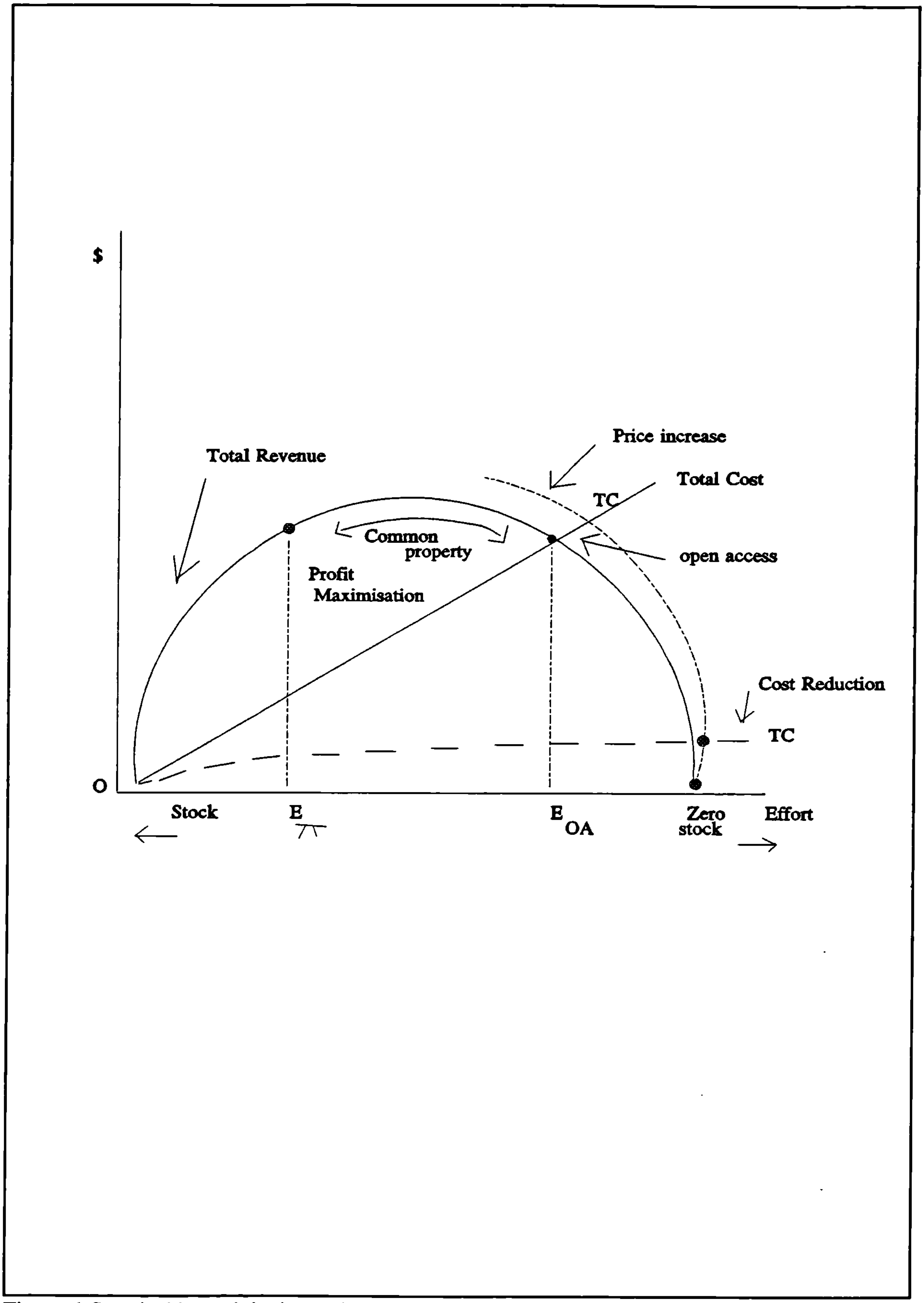

Figure 1 Sustainable exploitation and extinction.

the outcome of ignorance about population dynamics and partly the outcome of a failure to comprehend the risks inherent in unrestricted access, extinction under single ownership can be the outcome of deliberate planning. In terms of Figure 1 the private owner equilibrium is seen to be E_π where profits are maximised. The high price, low cost combination no longer threatens the species, although it will lead to changes in the profit maximising level of effort. It is when the picture ischanged from the (essentially unrealistic) static one of Figure 1 to a more dynamic context that the risks arise. The essential condition for dynamic profit maximisation is that the growth rate of the species stock should equal the single owner's discount rate[4]. The growth rate of the resource is effectively the rate of return on the resource. It then follows fairly obviously that if this rate of return is less than the rate of return the single owner can get by investing elsewhere (his discount rate is equal to the opportunity cost of capital) the resource will be run down to zero. It will pay to 'mine' the resource to extinction. Again, the analysis is suggestive since slow growth species will almost automatically be at risk on this model: their 'own rate of return' will be low. And this is how it tends to be in practice. Elephants, rhinoceros and whales are endangered whilst most (but not all) species of deer or seals are not.

Until recently, it was assumed that the fishery model was applicable to land-based species without much adaptation. Hence the analysis has tended to be in terms of property rights regimes, the price/cost ratio, and the discount rate/own growth comparison. For an application to the African elephant, see Barbier *et al.* (1990).

Swanson (1994) suggests some alternative angles to the Clark model once the resource in question is land-based and subject to greater human management. As assets they are just part of an asset portfolio. Importantly, whereas fisheries can be argued to have low opportunity costs - there are few competing uses of the seas - this is categorically untrue of land based biodiversity. This is because land based biodiversity depends on a base resource, the land itself, and this land does have alternative uses. The most obvious conflict is between land for conservation and land for 'development' uses such as agriculture. The critical point here is that whereas marine resources may be depleted because their growth fails to exceed the harvester's discount rate, in the land-based case they may be depleted because the returns from conservation fail to compete with the returns from land conversion to agriculture, roads, urban expansion etc. The second feature of the Swanson model is that whereas

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[4] This is true only if harvesting costs are not functionally related to the size of the stock. We assume they are not for the sake of simplicity. For more detail see Clark (1990).

marine resources have a (generally) fixed carrying capacity, this is not true of land -based resources. Carrying capacity is no longer a 'given': it is something that is determined by choices about the level of base resources allocated to biodiversity.

There are two ways of seeing the differences between these models. The first consists of a contrast between the profit maximising equations. In the Clark model what is being maximised is the difference between revenues from the sale of the fish and the costs of harvesting them. In the Swanson model what is being maximised is the difference between the proceeds of sale and the *sum* of the costs of 'harvest' and the foregone return to the alternative use of the land. Put another way, the net benefits of land-based conservation have to be greater than the total opportunity costs of conservation, ie the foregone net returns from developing the land. This is the basic condition examined in Pearce and Moran (1994), and, accounting for a time path for preservation benefits, the case originally laid out by Fisher, Krutilla and Cicchetti (1972) in their discussion of optimal preservation decisions. This basic cost-benefit decision rule motivates much of the subsequent discussion.

The second way of viewing the difference between the two models is to look at the policy implications. Taking the elephant example again, the Clark model blames high prices for ivory for the decline of elephant populations. The Swanson model can give the opposite interpretation, ie that ivory prices need to be kept high to encourage investment in sustainable management of elephant populations rather than in other assets. As a further illustration, the range states that invested in elephant conservation in a significant way were the southern African states - South Africa, Botswana, Zimbabwe and Namibia. These were the states where ivory sales were (largely) controlled and authorised, and where elephant populations grew dramatically. States that 'under-invested' in elephants, including even a country like Kenya where the tourist value of the elephant was high, were the ones that lost elephants to poachers.

The insights from the land-based alternative to the traditional extinction model are therefore important. It is not enough to take the property rights regime as given, as in the Clark models. For the property rights regime is a matter of choice and management. It is incorrect to cite open access, say, as a 'cause' of environmental degradation, for the question should be why open access is allowed to prevail. Put another way, why has an open access state not invested in changing the rights regime so as to allocate the base resource - land of suitable characteristics - to biodiversity ?

Once it is understood that biodiversity is competing with alternative uses of the land many things fall into place. First, population growth becomes immediately relevant because population growth simply intensifies the conflict as humans demand 'niches' to occupy as residences or as locations for crops, roads etc. Second, if markets in the 'products' of biodiversity are non-existent - as they are for the vast proportion of life on earth - the rate of return to conservation will almost certainly fail to compete with the rate of return to the alternative uses of land. Biodiversity is doomed. It is in the creation of markets for biodiversity that hope for conservation resides. While this conclusion offends the instincts of many environmentalists, it is the perpetuation of myths about moral obligations to, and rights of other species that reinforces the fate of biodiversity. By stripping the ability to compete in economic terms, emotive conservation arguments unwittingly contribute to the problem. A central element of the economic approach to changing government and popular perceptions about biological resources is to show that the sustainable use of biodiversity has positive economic value, and that this economic value will often be higher than the value of alternative resource uses which threaten biodiversity. This is no more than the prescriptions of standard welfare theory which has been influential from the earliest applications of environmental economics (Fisher *et al* op. cit). Prior to addressing this process of giving value to biological resources, some additional factors tilting the niche competition against biological resources require attention.

Fundamental and proximate causes of loss

In welfare terms the main reason that biodiversity is being eroded is that there is an underlying disparity between the private and social costs and benefits of biodiversity use and conservation (Dixon and Sherman 1990; Perrings and Pearce 1992). Market failure and institutional failure in all countries of the world contribute to a global externality problem which must typically be addressed or internalised on a location-specific basis using an array of policies and instruments. Notwithstanding the elegance of the foregoing theoretical model of extinction dynamics, the basic fact is that neither markets nor institutions can regulate the unknown. While that resulting prescriptions may be predicted on an incomplete understanding of the natural world, they do offer some powerful insights which rarely emerge from alternative arguments.

Local and global market failure, government intervention failure, plus the forces of social change such as population growth, comprise the fundamental causes of biodiversity loss.

It is also possible to identify proximate causes which show up in different patterns of resource use. The main proximate cause of loss is *land conversion*, i.e. the conversion from one land use to another, where land use includes sustainable management systems or even doing nothing with the land at all (wilderness). Much of the species-area story is based on land use change, and it is possible to skip a wealth of data on global land use change without losing the essential point (see Pearce and Moran 1994).

All these forms of failure can co-exist. Moreover, they exist very often in a context of rapidly changing population as far as developing countries are concerned. But these forms of failure are not peculiar to developing countries. Local market failure and government failure are present to some degree in most countries.

Figure 2 taken from Brown *et al* (1993) introduces a diagrammatic exposition of the types of economic failure. The horizontal axis shows the amount of land converted to, say, agriculture. The vertical axis shows money. The downward sloping line MPB_i is the 'marginal private benefits' of land conversion, i.e. the extra revenue obtained by the farmer by converting the land from forest to agriculture. The line MC_i is the marginal cost to the farmer of making the conversion. The 'rational farmer' will equate MC_i and MPB_i in order to maximise profits[5]. Hence the amount of land conversion that actually takes place is L_P.

Now suppose the farmer is subsidised to convert the land. The effect can be shown as a lowering of MC_i to MC_i - SUB, where SUB refers to the subsidy. That is, private costs are lowered. This induces the landowner to expand the level of land conversion to L_{P+S}. The distance L_P - L_{P+S} is a measure of *government failure* (GF).

To determine the socially optimal level of land conversion we need to estimate the value of the two externality components: the local and global externality. This involves *valuation*. If we know the

[5] To see this, profits, π, equal PB(L) - C(L), i.e. the private revenues from conversion less the costs of conversion. Maximising profits and differentiating gives

$$d\pi/dL = dPB/dL - dC/dL = 0$$

or

$$dPB/dL = dC/dL$$

But the left hand expression is marginal private benefits (MPB_i) and the right hand expression is marginal cost (MC_i).

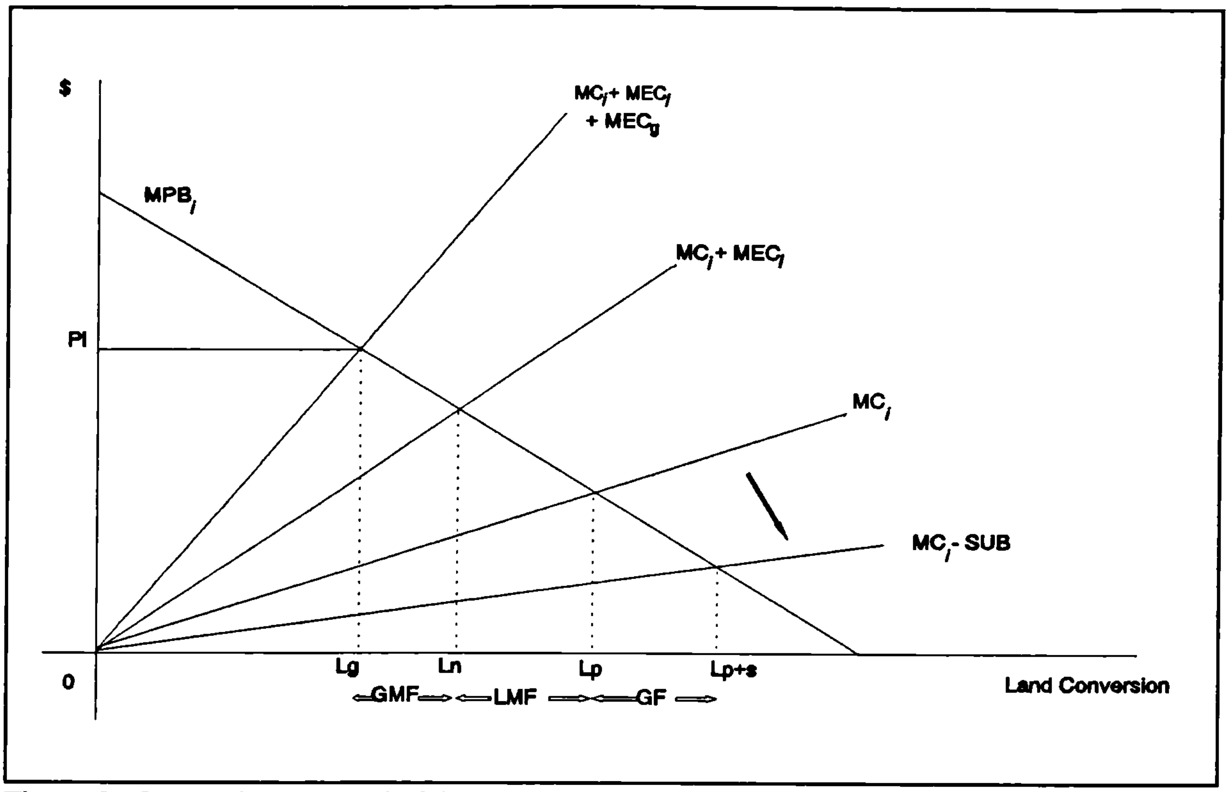

Figure 2 Measuring economic failure

value of the damage done to the nation from such land conversion -e.g. lost indirect and direct use values - then some estimate of the local external cost can be made. The diagram shows this as MEC_i, i.e. the marginal external cost imposed on the nation. If this externality is 'internalised', i.e. if the farmer is made to account for it in some way (e.g. by taxation or by bearing higher costs because the land is zoned for conservation) then the relevant 'optimum' moves to L_N. Note that L_N is less than L_P, so that internalising the externality involves less land conversion and hence more biodiversity conservation. The distance L_N - L_P is a measure of the *local market failure* (LMF).

The same procedure can be used to account for the global externality, the value of the losses borne by people outside the nation that owns the forest. The distance L_G - L_N is a measure of *global market failure* or *global appropriation failure* (GMF).

The analysis also provides us with a rule for the 'proper pricing' of land. It is given by:

$$P_L = MC_i + MEC_i + MEC_g$$

Note that this optimal amount of land conversion is not zero - some deforestation still takes place. This result of the economic analysis is often regarded by environmentalists as unsatisfactory. Indeed, if one adopts the 'moral' standpoint discussed (in conclusion) it will appear to be 'immoral' to allow *any* deforestation to take place. This illustrates a further contrast between the economic and the moral standpoint: the latter tends to focus on the costs of deforestation *only*. It ignores the benefits of deforestation, i.e. the gains to be obtained by the farmer in question. The economic approach quite explicitly compares these benefits with the costs. The potential effect of population growth and economic growth can also be demonstrated in Figure 2. It can be illustrated by shifting the MPB curve to the right over time. If the costs of further land conversion do not change (which they might as the 'frontier' gets further and further from established urban centres), then such shifts will make it more worthwhile to convert land.

Clearly, then, if there is to be a policy on biodiversity conservation it has to focus on the main fundamental causes of loss identified in this section. The information requirement stands to reason in the sense that priorities should be set for directing investment. Local market failure may be addressed by local measures such as the zoning of land to forbid, say, forest burning. Global market failure which will need to be addressed by 'creating' markets in global value and ensuring that compensation for forgoing the development option is paid to the landowner. Finally intervention failure which will need to be addressed by showing governments that there are gains to be made by avoiding expensive policies of subsidisation of forest clearance. In sum all these measures will make the conservation versus development cost-benefit trade-off more transparent. A question remains about fundamental forces driving niche competition (ie population growth) and the development process in general.

1.6 Conservation Versus Development.

The basic trade-off between biodiversity conservation and alternative land use may be summarised as:

$$B(CON) - C(CON) > B(DEV) - C(DEV)$$

or

$$B(CON) - C(CON) - [B(DEV) - C(DEV)] > 0$$

where

B(CON)	=	the benefits of sustainable use of the forest;
B(DEV)	=	the benefits of traditional development of the land for, say, agriculture or forestry or industry;
C(CON)	=	the costs of the sustainable use option;
C(DEV)	=	the costs of the development option; and the inequality indicates the requirement for conservation to be preferred.

In fuller form:

$$\int_0^T B_t(CON) - C_t(CON) - [B_t(DEV) - C_t(DEV)] \, . \, e^{(\rho - r)t} > 0$$

where:

T is time horizon;

ρ is rate of relative price appreciation;

r is the discount rate.

Depending on the complexity of the problem, it may be straightforward to factor any intertemporal dimension of the trade-off using discounting. However, the concern here is more with augmenting this decision to take account of the social costs and benefits associated with biodiversity conservation.

Beyond the issue of market development for investment in biodiversity, the more fundamental development question may be asked about the nature of the process which pits conservation versus development. In Summer 1992 in Rio de Janeiro the world's nations agreed a global Convention on Biological Diversity. It aims to protect the world's biological resources from further erosion or, at least, to slow that rate of erosion down. Yet the rate of erosion of biodiversity is increasing and there is no reason to expect one of the fundamental reasons - population growth - to decrease. Apart from the evolutionary evidence on periodic and background extinctions and the theoretical explanation provided above, what then is the underlying trend relating biodiversity and economic growth? Understandably all the uncertainties highlighted here obscure the picture that is undoubtedly emerging at several levels of biological organization. At best it is possible to paint a partial picture such as the case with environmental Kuznets curves[6] (see for example Grossman and Krueger 1995). These curves relate a growth indicator such as per capita GDP (cross-section or time series) to a particular environmental pollutant to describe critical turning points in the development process. In other words, at what point do countries apparently see the value of investing? From the biological literature alternative relationships between anthropogenic activity and threatened species have been determined using Path Analysis (Kerr and Currie 1994). Antle and Heidebrink (1995) demonstrate the trade-off in terms of the more traditional Production Possibility Frontier and alternative income expansion paths implied by potential views on the income elasticity of demand for aspects of environmental quality. Their point is much the same as that raised in the Kuznets curve literature, relating to the empirical determination of environment-growth turning points. Arrow *et al* (1995) consider the implications of an excessive emphasis on growth thresholds which can be inferred from this literature. They stress the importance of the content of growth (in terms of property rights and humanity's ability to gel with, rather than destroy ecosystem dynamics), rather than growth *per se*. Although implicit in their treatment of carrying capacity, the same authors avoid the ultimate doomsday scenario of growth prescriptions swamped by population pressure. The latter remains as a rather depressing bottom line and the humanity's ultimate fate in the grand scheme of evolutionary time. More immediately, the review by Antle and Heidlebrink does provide another view of the development trade-off when factors with public good characteristics are available to be used at a suboptimal rate. In other words, their view of the conservation development choice again raises the issue of market failure and valuation to which it is now appropriate to turn.

1.7 Valuation

Notwithstanding the negative implications of the doomsday scenario, the immediate story emerging form all this is that valuation of biodiversity can be a powerful method for allocating resources. But valuation matters because biodiversity loss matters. It matters because there are definite economic consequences of its loss.

The potential effects of accelerated extinction and depletion of the genetic base may be discerned over varying time horizons. In the long term, processes of natural selection and evolution may be dependent on a diminished resource base, simply because fewer species are being born. The implications of species depletion for the integrity of many vital ecosystems are far from clear. The

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[6]As yet the nearest such relationship is that between deforestation and GDP

possible existence of depletion thresholds, associated system collapse, and huge discontinuities in related social cost functions, are potentially the worst outcome in any reasonable human time horizon. Such scenarios are indicative of the links between ecosystem integrity and economic well-being. Similarly the impoverishment of biological resources in many countries might also be regarded as an antecedent to a decline in community or cultural diversity, indices of which are provided in diet, medicine, language and social structure (Harmon 1992).

It is possible to speculate at length about the effects of losing biodiversity and there is ample evidence on the effects of crop yield variability resulting from genetic concentration in agriculture (Hazell 1989). More immediately, we regret extinction because of the irreversibility of the act. In many ways we are diminished and welfare change occurs. Even without these more intangible effects, and whatever view is taken in the extinction debate, conservation raises some genuine economic choices. Primary among these is the issue of whether biodiversity is a waste of space. For example, area requirements for long term conservation of viable populations at average species population densities, suggest minimum requirements as follows: 33,000 ha for primates (Norse et al 1986); 10,000 ha for hornbills (Medway and Wells (1976); 250,000 ha for large mammals like Elephants (Olivier 1978). Thus the conservation of both flora and fauna requires large extents of (typically) forest amounting to approximately several hundred thousand hectares (250,000 - 400,000 ha) as the minimum size for reserves if tropical and sub tropical forests are to safeguard most species. An economic basis for conservation (particularly in high opportunity cost developing world) is paramount.

A Classification of Valuation Procedures

Non market valuation has developed as an application of the neoclassical theory of welfare change measurement and seems ideal for dealing with the problems just outlined. In the case of resource availability, the goal is to estimate the amount of money that would just compensate the demanders of that resource. There is a huge and growing literature on the origins development and current status of methods (Freeman 1993; Bjornstad and Kahn 1996) and specifically in relation to biological resources (Pearce and Moran 1994; Jakobsson and Dragun 1996). The aim here is to introduce the main methods (and associated variants) and to justify the choice of a particular approach as the basis of further research. Some reference to alternatives to the neoclassical paradigm are reserved until the concluding chapter.

There are basically two broad approaches to valuation, each comprising a number of techniques. The approaches are the Direct and Indirect approaches. The Direct approach looks at techniques which attempt to elicit preferences directly by the use of survey and experimental techniques, such as the Contingent Valuation and Contingent Ranking methods. People are asked directly to state or reveal their strength of preference for a proposed change. In contrast, Indirect approaches are those techniques which seek to elicit preferences from actual, observed market based information. Preferences for the environmental good are revealed indirectly, when an individual purchases a marketed good with which the environmental good is related to in some way. These techniques included the Hedonic Price and Wage techniques, the Travel Cost method, Avertive Behaviour and Conventional Market approaches. They are all Indirect because they do not rely on people's direct answers to questions about how much they would be willing to pay (or accept) for an environmental quality change.

The Direct Valuation Approach

In the direct approach, an attempt is made to elicit preferences by either experiments or questionnaires. The use of surrogate markets in many fields such as marketing and psychology has evolved into the range of hypothetical market methods comprising contingent valuation (Mitchell and Carson 1989) and other stated preference methods like contingent ranking and conjoint analysis (Adamowicz 1994, Louviere 1994). These are all variants on asking people directly to state or reveal 'what they are willing to pay (WTP) for some change in provision of a good or service or to prevent a change' and/or 'what they are willing to accept (WTA) to forego a change or tolerate the change. In ranking exercises this information is inferred from the choices made between various bundles of goods including levels of the environmental good. In all cases a contingent market encompasses the good itself, the institutional context in which it would be provided, and the way it would be financed. The situation the respondent is asked to value is hypothetical and respondents are assumed to behave in an identical way to that in a real market. Structured questions and various forms of 'bidding game' can be devised, involving 'yes/no' answers to questions regarding maximum willingness to pay. Econometric techniques are then used on the survey results to find the mean bid values of willingness to pay.

The main advantage of CVM is that in theory it measures precisely what the analyst wants to know - the individual's strength of preferences for the proposed change - and can be used not only for non-marketed goods and services, but market goods as well. If people were able to understand clearly the change in environmental quality being offered, and can answer truthfully, this direct approach would be ideal. However the central problem with the approach is whether the intentions people indicate ex-ante (before the change) will accurately describe their behaviour ex-post (after the change) when people face no penalty or cost associated with a discrepancy between the two. This is known as 'Strategic Bias' and occurs if there is a 'free rider' problem. Contingent valuation methods are all handicapped by a number of other biases related to the nature of the questionnaire, the interview and elicitation format and the estimation and ultimate use of mean or median willingness to pay data. Equally there is the important issue of the validity of the methodology and responses vis a vis economic and psychological theory. Most importantly given the level of uncertainty associated with many aspects of the diversity issue, the question of cognition is central to a decision about the use of valuation methods. In the case of stated preferences the extent of what can be valued is - at the limit - constrained by what the investigator understands to be the issue. For brevity the reader is referred to the comprehensive treatment of bias and validity issues the growing literature on hypothetical methods, in particular the earliest definitive guide to the method (Mitchell and Carson 1989).

The Indirect Valuation Approach

Indirect approaches are those techniques which seek to elicit preferences from actual, observed market based information. Preferences for the environmental good are revealed indirectly, when an individual purchases a marketed good with which the environmental good is related to in some way. The techniques included Hedonic Price and Wage techniques, the Travel Cost method, Avertive Behaviour and the Dose-Response method. They look at what people actually do pay. In the eyes of many economists this behavioural trail is infinitely more reliable than any hypothetical responses. A behavioural trail is also left by the Replacement Cost technique which is often evoked as a surrogate valuation method. However, it does not share the same theoretical basis as the previous methods

The Indirect group of techniques can be divided into two categories. These are: surrogate market approaches and conventional market approaches. Surrogate market techniques involve looking at markets for private goods and services which are related to the environmental commodities of concern. The goods or services bought and sold in these surrogate markets will often have as complements (or attributes) and substitutes the environmental commodities in question. Individuals reveal their preferences for both the private marketed good and the environmental good when purchasing the private good.

The main methods are surrogate market approaches include Hedonic techniques and Household

Production Function techniques. Expenditures on commodities that are *substitutes* or *complements* for the environmental characteristic are used to value changes in that environmental characteristic. The latter includes the travel cost method which has been widely employed to value biological resource (see chapter five). Hedonic methods are increasingly used in areas such as air and water quality. Garrod and Willis (1992) have employed the technique to evaluate the effects of forest characteristics on proximate property prices. The Hedonic Method (HPM) is in fact similar to the Household Production Function approach since both make a complementarity assumption. With HPM an attempt is made to estimate an *implicit price* for environmental attributes by looking at real markets in which those characteristics are effectively traded. Thus, 'clean air' and 'peace and quiet' are effectively traded in the property market since purchasers of houses and land do consider these environmental dimensions as characteristics of property. The attribute 'risk' is traded in the labour market. Similarly high risk jobs may well have 'risk premia' in the wages to compensate for the risk. The land or property market and the wage differential method are the two most widely used HPM.

Conventional market approaches when output is measurable

There are a number of alternatives to the methods outlined above, using market prices for the environmental service that is affected, or, if market prices are not an accurate guide to scarcity, then they may be adjusted by shadow pricing. Where environmental damage or improvement shows up in changes in the quantity or price of marketed inputs or outputs, the value of the change can be measured by changes in the total 'consumers plus producers surplus'. If the changes are small the monetary measure can be approximated by *market values*. Two approaches doing this are the dose-response technique and the replacement cost approach. The first aims to establish a relationship between environmental damage (Response) and some cause of the damage such as pollution (Dose), such that a given level of pollution is associated with a change in output which is then valued at market, revealed/inferred, or shadow prices. The replacement cost techniques looks at the cost of replacing or restoring a damaged asset to its original state and uses this cost as a measure of the benefit of restoration. The approach is widely used because it is easy to find estimates of such costs. Interest in replacement and mitigation as an alternative to resource compensation is growing in many countries. This is partly due to the complexity and controversy surrounding valuation methods discussed here. The approach is correct where it is possible to argue that the remedial work must take place because of some other constraint such as a water quality standard. Under such a situation the costs of achieving that standard are a proxy for the benefits of reaching the standard, since society can be assumed as having sanctioned the cost by setting the standard. However, if the remedial cost is a measure of damage then the cost-benefit ratio of undertaking the remedial work will always be unitary. That is to say remedial costs are being used to measure remedial benefits. To say that the remedial work must be done implies that benefits exceed costs. Costs are then a lower bound of the true value of benefits. If, to pursue the water quality example, the standard has clearly been set without thought for costs, then using replacement costs as a measure of minimum benefits could be misleading.

Another situation where the replacement cost approach is valid would be where there is an overall constraint not to let environmental quality decline (sometimes called a sustainability constraint or a minimum standard). In these circumstances replacement costs might be allowable as a first approximation of benefits or damage. The so-called shadow project approach relies on such constraints. It argues that the cost of any project designed to restore an environment because of a sustainability constraint is then a minimum valuation of the damage done.

Information on replacement costs can be obtained from direct observation of actual spending on restoring damaged assets or from professional estimates of what it costs to restore the asset. It is assumed that the asset can be fully restored back to its original state. However some damage may not be fully perceived, or may arise in the long term, or may not be fully restorable. Benefits will therefore be underestimated. Another problem here is that restoration of damaged assets may have secondary benefits in addition to the benefits of restoration such that replacement costs will underestimate total benefits.

Opportunity costs are the final approach in which no direct attempt is made to value benefits. Instead, the benefits of the activity causing environmental deterioration - say, a housing development - are estimated in order to set a benchmark for what the environmental benefits *would have to be* for the development *not* to be worthwhile. Clearly, this is not a valuation technique but, properly handled, it can be a powerful approach to a form of judgmental valuation. It is used here to indicate the kinds of economic returns that must be secured by biodiversity use if such land uses are to be economically preferred to the alternative land use.

1.8 Choice of Valuation Technique

A growing body of case studies is indicative of the appeal of valuation techniques as additional tools to guide the conservation of biological resources (Pearce and Moran 1994). The choice of a particular techniques is motivated by a specific objective. In this case the aim is the conservation of biological resources and preferably the conservation of biological diversity *per se*. Some methods are undoubtedly superior in serving both objectives. The case made here is that despite potential biases and validity problems, CV offers greatest flexibility through direct enquiry.

Interest in CVM has increased over the last decade or so because, firstly, the method provides the only available means for valuing Non-use values - the values obtained from Indirect techniques are not aimed at capturing Non-use values. Secondly, estimates obtained from well designed properly executed surveys appear to be as good as estimates obtained from other methods; Thirdly, the design, analysis and interpretation of surveys has improved greatly as scientific sampling theory, benefit estimation theory, computerised data management and public opinion polling has improved. Relative to other methods, the appropriate data can be generated easily (perhaps too easily). Subject to cognitive limitations, the investigator can identify and attempt to value any facet of biological diversity, which is not the case with the two main alternative valuation methods. In the case of the Travel Cost method, several key assumptions about travel motives are necessary. Basically the central assumption is that visit costs can be taken as an indication of recreational value. However, if individuals have changed their place of residency so as to be close to a site then the price of a trip becomes endogenous and the central assumption is violated. Similarly time costs and the issue of meandering visits require arbitrary assumptions assigning value to the good of interest. Similarly, using the hedonic pricing method, the extent of inference between the values of lower aspects of diversity and complement goods are limited. In both cases issues relating to functional form introduce arbitrary assumptions that can affect the valuation of resources (Willis and Garrod 1991; Garrod and Willis 1992). In the travel cost case the regression function relating trips to cost may be non linear and unbounded. In HPM the form of the hedonic price function is also crucial. But CV is not immune to such assumptions and these are discussed in chapters three and four. Clearly though the ability to construct a market and control the hypothetical change is an immense advantage relative to the use of revealed data. This advantage is the main justification for subsequent use of the method, and it is therefore appropriate to summarise the essence of applying CV.

There are three basic parts to most CV survey instruments:

First, a hypothetical description (scenario) of the terms under which the good or service is to be offered is presented to the respondent. This will include information on when the service will be available, how the respondent will be expected to pay for it, how much others will be expected to pay, what institutions will be responsible for delivery of the service, the quality and reliability of the service.

Second, the respondent is asked questions to determine how much he would value a good or service if confronted with the opportunity to obtain it under the specified terms and conditions. These questions take the form of asking how much an individual is WTP or WTA for some change in provision. Depending on the preferred elicitation format, econometric models are then used to infer a WTP for the change. An aggregate welfare measure can be calculated by multiplying a favoured measure of response central tendency (mean or median) over a relevant population of users.

Finally, response validity is tested by relating WTP responses to respondent socioeconomic and demographic characteristics. Confirmation of *a priori* expectations of the relationship between WTP income, age and other variables, being a good indication of meaningful responses.

1.9 Conclusion

This chapter has established the nature of the issue, explains why it matters and proposes a short term solution to the problem of biodiversity loss. The biodiversity problem can clearly be accommodated to a considerable extent within a cost-benefit framework. The questions to address are what do we value about biodiversity and what categories of value are met? Can we systematically value biodiversity and how far does this help set priorities for conservation?

Available methods can help demonstrate the economic values of biological resources in the contexts where the values are often not reflected in market processes. Explaining why, despite those economic values, biodiversity continues to be threatened involves finding ways to capture or realise economic value. As a whole, this thesis attempts to address how far have we come and where do we need to go in this endeavour. It takes an unapologetically utilitarian view of conservation value

In the grand scheme of global change, we are clearly involved in a holding operation. Many uncertainties need to be addressed. However, evoking the arguments of option and quasi-option value, economic valuation can be regarded as the device to buy time to allow the more informed choices which will have to be made in future. We argue that addressing the economic causes of biodiversity loss is extremely important if the world really does want to slow down the erosion of its biological resources. Much of the biodiversity that needs saving resides in the developing world. Since biodiversity conservation is frequently not, and understandably so, a priority for the developing world, the resources needed for conservation must come from the North, while the political commitment must come from the South and North alike. However we would like the world to be, the brute fact is that only policies which offer mutual self-interested gains to North and South alike stand a chance of succeeding. In the longer term we may hope for changes of attitudes and priorities in the world generally, especially as incomes rise in the South. But relying on such changes to bring about conservation is foolhardy and counterproductive. That is why the economic approach matters. It does emphasise mutual economic gain as the foundation for the solution to the biodiversity problem.

The following chapters are organised as follows. Chapter two on diversity theory, examines the meaning of diversity and value in a biological context. The chapter also establishes the interface with economic developments which *inter alia* show how national and global conservation priorities can be set in theory and practice. On the basis of current understanding there may be a basic irreducibility in the concept that for the foreseeable future creates a data problem for establishing a consistent cost-benefit approach for dealing specifically with diversity. Accordingly, chapter three is unapologetic in resigning any further attempt to be precise about diversity (rather than biological resources), and addresses the theoretical developments behind the proposed use of CV to value biological resources. Chapters four and five put theory into action in applications to wetland valuation and game parks respectively. Chapter six draws a line under the use of valuation and adopts a middle ground between economics and biology to address the issue of setting global priorities. In a sense the issue amounts to the adoption of second best in both fields to move towards a global optimum level of conservation investment. Finally chapter seven concludes the thesis with a brief review of policy alternatives and brief reflections on future avenues of research in a rapidly moving field.

Chapter Two
Biodiversity theory

2.1 Introduction

As is apparent from the overview in Chapter One, the terms 'biodiversity' and 'biological resources' are often used interchangeably. Thus references to the valuation of biodiversity typically refer to specific tangible resources such as forests or animal species which, by virtue of their familiarity are amenable to the use of established environmental valuation methods. Working at this level it is possible to develop a compelling cost-benefit case for conservation. However a question arises as to the extent to which this conservation case is consistent with any biological conservation criteria.

From a biological perspective, it can be argued that the objective function in conservation decisions is somewhat vague. Focusing on the total economic value (TEV) taxonomy is not necessarily the same as measuring the value of diversity *per se* which is arguably the most important biological objective. The theoretical and empirical cases for diversity preservation have received limited attention outside the fields of ecology and plant breeding. The example of the impact of genetic concentration on agricultural yield variability is frequently evoked as the clearest example of the perils of diversity loss. Thus, staving off the onslaught of diseases and famine seems to be a good reason to value diversity in some species regardless of the fact that most people instinctively seem to prefer a heterogeneous environment. But the nature of preferences for a more diverse world cannot be taken for granted and it is far from clear whether below-species (e.g. genetic) diversity actually has economic value. In the first instance our ignorance of the extent of biologically 'significant' differences constrains our ability to form preferences over anything other than those we find physically endearing. These preferences can provide economic values but arguably they are not the only values which ought to be the basis of quantitative (e.g. cost-benefit) decisions. As it is, conservation decisions made on alternative qualitative grounds reflect the conflicts inherent in divergent value objectives. Most recently these differences have been evident in the discussion of conservation priorities (McNeely 1996). Only if by some fluke were biological and economic criteria to coincide would consensus be conceivable. Alternatively it may be possible to dictate conservation of large units which are surrogates of the biological wealth they undoubtedly contain. The science of such a strategy is still unproven.

We would like conservation decisions to be determined by economic criteria and, if diversity does have some option/existence welfare significance[1] then this should be factored into conservation decisions. Analysis of conservation programmes has shown that charismatic fauna inevitably get the lion's share of conservation spending (Metrick and Weitzman 1994). This economic fact means that species charisma rather than say, their genetic complement is valued most highly. Like the problem of relying on species richness data (i.e ranking areas by the number of species they contain), this is most certainly a default decision and it seems reasonable to investigate the extent to which diversity *might* be made commensurate with cost-benefit decisions *were* it the case that difference actually matters. Research into the nature of diversity and its economic significance is prior to the application of conventional valuation methods and the purpose of this chapter is to see the extent of any quantifiable link between diversity and value to go beyond current (qualitative) preservation methods. This issue is relatively unexplored[2] but it seems reasonable to evaluate its immediacy for conservation policy before opting for less precise valuation methods. This chapter therefore presents a critical review of diversity theory as it currently stands and is organized as follows. First, an appropriate level of diversity is characterised as a basis for selecting a realistic level of analysis. This scale needs to be tractable and yet consistent with an established biological framework for measuring diversity between species. The area in question is the field of taxonomy which includes the study of biosystematics and phylogenetics (see UNEP 1995, pp.28). Second, the chapter evaluates a formalised economic approach using taxonomic structures. As will become clear, the different methods of formalising 'difference' relationships do not facilitate economic analysis. The section looks at models suggested for measuring the diversity of competing sets of species and, importantly, for measuring the loss of diversity resulting from an extinction. Given a watertight diversity measure - which has yet to be established - both these issues can be cast in a notional cost-benefit framework which is maintained as the relevant decision framework.

Having outlined the measurement of difference, the following section speculates about the utilitarian motives which link diversity theory to monetary valuation. Given the uncertain basis of the existence rationale, the implicit assumption is that biodiversity has to be related to a use motive. Thus, using the justification of (future) use-related option value, the analysis shows that distance - that is, a

[1]The important point being that apart from option and existence values, no other direct and indirect economic values can be demonstrably attributed to this facet as a means to providing an obvious rationale for conservation.

[2]Exceptions being the work of Weitzman (1992) and Solow *et al* (1993) which motivated the line of enquiry in this chapter.

cardinal measure of pairwise genetic or other character difference between species - is only more desirable if it increases the likelihood of finding a useful product such as a cure for a disease. This issue is closely related to familiar arguments for the maintenance of crop genetic diversity in agriculture which have received considerable theoretical (but less empirical) attention (Pimental *et al* 1996). Because of the potential intractibilities of dealing with the returns to an almost infinite pool of character[3] 'distances' or differences, consideration of use-related option value requires some probabilistic link between the preferred diversity measure and value. The ideal model will also account for the behaviour of agents involved in biodiversity prospecting.

The final section serves as a reality check from both a scientific and economic perspective. Despite the best scientific endeavours to map life on earth at all levels, it seems unreasonable to imagine that the models outlined here will prove immediately practical for arresting current rates of decline in biodiversity. While the discussion opens up what appears to be an unexplored interface between economics and biology, the immediacy of the problem points to working with what is known about diversity, and the establishment of surrogate approaches. As it happens, a basic indicator of species richness may be both a scientifically acceptable unit of account, and amenable to economic considerations (via a species-area relationships established in population biology). This information facilitates ground-level decisions about area designation for *in situ* conservation. The section attempts to describe the importance of well-established programming methods in area selection, and their role in the important topic of conservation priorities which will be discussed further in chapter six.

The ultimate objective of this chapter is to indicate the potential for alternative approaches to current *ad hoc* and opportunistic policy approaches to conservation. It is apparent however that the successful resolution of many biodiversity problems will require either a polymath or much closer collaboration between several disciplines, including mathematics biology and economics and operations research.

2.2 Valuing diversity

It is true that no unifying theory of diversity quantification exists to make diversity decisions commensurate with cost-benefit criteria. The complex nature of biodiversity may serve as a reasonable if not complex metaphor for refining a general theory, and there is a considerable body of research in statistics and information theory dealing with diversity measures. Attempts to formalise a diversity measure in other fields include those for linguistic diversity (Lieberon 1969), industrial concentration

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[3]Where characters may be any appropriate defining trait from genes to morphological differences.

(Horowitz 1970), income inequality (Theil 1967), and possibly architectural diversity. Speculative links to the concept of entropy via information theory have been made by Weitzman (1992). Information theory is also the basis of the Shannon-Weiner index of diversity which is used in ecology (see Krebs 1994). This is a method for comparing the diversity of areas on the basis of patterns of presence or absence of different species. As such, the measure is not strictly different to species richness and misses a fundamental aspect of difference embodied in evolutionary structure.

Table 1 Diversity scales

Advantage: precision as a measure of character diversity	A Scale of surrogacy for character diversity	Advantage: ease of measurement
low	(ecosystems)	high
⇓	landscapes	⇑
⇓	land classes	⇑
⇓	species assemblages	⇑
⇓	higher taxa	⇑
⇓	species	⇑
high	(characters)	low

Source: Williams and Humphries (1996)

As chapter one showed, it is possible to define diversity measures at various levels of the biological organization. In one sense, since biodiversity *is* the irreducible complexity of all life, then there can be no single objective definition, only measures appropriate for restricted purposes (Norton 1994). There is inevitably a good deal of subjectivity in any approach to the issue, and by extension, potential for conflict in the decisions related to prioritization on any particular basis. However, to clarify these scales, Table 1 describes the relationship between scale and cost of actually conducting analysis. Thus focusing on sub-species genetic traits will arguably produce the most precise difference measures but will inevitably imply a high research cost if all species are considered. In contrast, there are relatively few ecosystem and landscape types to be categorized and differentiated. On the other hand, the problem is that the science of relying on these as surrogates to capture lower diversity remains imprecise.

One view is that measurement might be dictated by qualities which give value. The basis of diversity value was mentioned in the introduction and the problem here is essentially that the maximisation of option value could be consistent with diversity measures at a number of scales. Option value is basically about preserving flexibility in terms of present and future potential direct and indirect uses[4]. The interpretation of uses may be reflected in an eventual market value or more widely related to a system-wide potential to adapt to future change (Reid, 1994) and resilience to shocks and potential discontinuities. Although these use interpretations appear drastically different, there is no scientific basis to focus on diversity measures at either end of the hierarchy (in table 1) as arguments can be made for both ends. Thus focusing on surrogates (from table 1) is advantageous from both a cost perspective, and because higher levels of analysis may integrate more of the functional processes which may be nearer the elusive ' glue' which holds everything together (Walker 1992, Reid *et al* 1993). However, the extent to which this is possible is still uncertain and it is also not clear how a consistent diversity theory can be developed at the habitat level[5].

At the opposite extreme, much more of the fundamental genetic constitution of organisms is being revealed at an ever more rapid rate. In the following analysis it will be helpful to think of individual or combinations of genes as the ultimate character units under discussion. The choice of this character level is arbitrary, and there is absolutely no consensus on what the basis of measuring diversity should be. Clearly the interpretation of option value based on specific character representation goes to the heart of what exactly is the nature of diversity-related value from conservation. Attempts to maximise option value will dictate different approaches to conservation of the correspondingly 'optimal' portfolio of character diversity. The two preceding views of option value correspond to divergent avenues of research which will be discussed. Precision in any empirical application is tempered by the inability to directly enumerate all possible characters and/or all possible character combinations. Maximising diversity value via a surrogate or predictive (character) pattern methods is an alternative. As will be seen these may also be the basis for constructing a notional index for ranking biodiversity sets.

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[4] Fisher and Hanemann (1985) offer a now fairly standard interpretation of option value as "the present benefit of holding open the possibility that a future discovery will make useful a species that we currently think useless".

[5]The diversity theory outlined here deals ostensibly with the inter-species diversity (eg genetic character) complement. On reflection, one might conceive of ecosystems as surrogates for species within which the character complement might be measured simply by the number of different species.

Defining characters

As the basis of measuring difference between organisms there are many characters to choose from. Reasonable candidates are genes (differences in patten of several genes or the structure of one specific gene for members of the same family), phenotypic/morphological traits (i.e what species look like), or functional roles. The use of genetic distance measures need not be highly correlated with phenotypic characteristics, and vice versa. The preference for concentrating at more refined levels means that the focus of interest is on character sets below the species level. There is inevitable uncertainty about how and which characters will turn out to be of value. Furthermore, the emphasis on a particular character type immediately weights those organisms which add more of that attribute. The implications of this weighting problem will be discussed below, but there are several facts to bear in mind when considering the development of any diversity theory. First, the immense diversity of life on earth at all its levels (known and unknown), means that the true measurement of diversity is basically impossible. Identification is on-going and any measure is (by definition) partial, if not on occasions completely speculative. Second, as will be seen, such uncertainties hinder the establishment of a consensus objective for measuring diversity. Third, focusing on the lowest level of analysis (i.e character types of table 1) moves further away from economic valuation methods. Morphological characters, for example, are appreciably closer to the popular perception of biodiversity than unexpressed genetic traits. It may be possible for individuals to express preferences for relatively diverse states of the world; how meaningful such an exercise would be at the genetic level is in doubt.

Valuation and Taxonomy

The complexity of diversity is given some order by the discipline of systematics - a branch of taxonomy which involves the study of the diversity of organisms and any and all relationships among them (UNEP 1995 pp.28). The focus on common relationships between species or their particular character traits (homologies) that define taxonomic structure can be extended to the study of the numerous differences in organisms (constituent chemicals, morphological structure and behavioural traits). This analysis gives rise to estimates of phylogeny (lineage trees) which interpret characters as sharing a single evolutionary origin through common ancestry, and by extension, denotes difference (i.e characters not shared) as a branching process. The phylogeny of a set of species thus reflects the evolutionary diversity of that set. Each species differs, and is valued, according to the amount of evolutionary information that is unique to it, compared with parts shared with other species in the set.

The construction of phylogenetic tree structures which provide the basis of a diversity measure, is about quantifying the extent of total mismatch of a set of species. These relationships can be based on different levels of information. The most information intensive measures construct trees in which branch lengths have a cardinal interpretation (e.g. Weitzman 1992 see below), followed by trees where the branch lengths do not have a distance interpretation. In the latter case it may be sufficient to infer diversity of a set of related species from the structure of branching points on the appropriate tree. Alternatively some hypothetical distance measure can be derived from a model which attempts to describe how the various branches grow and can be predicted to evolve in future.

In all cases the construction task is controversial. In the first place, even though it helps to think in terms of genes, such structures can in theory be based on any character unit. Where exact genealogical information is available, this can be used to determine the number of differences between pairs of species in a set of interest. Thus, as Crozier and Kusmierski (1994) show, it is possible to build up an exact branch length tree of the dissimilarity between species in the bower bird family (*Ptilonorhynchidae*) on the basis of one portion of DNA sequence from one single gene[6] of each species member of the family. This is a risky strategy however, as the pattern encoded in this one bit of sampled DNA may not be the truest reflection of the difference between family members. In other words, the representation that appears from this information is at best hypothetical as a result of using one many sampling strategies to get the material on which to base difference measures itself. Clearly there are statistical complications. Furthermore, the construction of species family trees requires more than one piece of tissue which only gives a snapshot of how things are currently related. The inference of how they continue to change requires the use of models which predict future character changes on the basis of those inferred to have given rise to current species[7]. An alternative way to view this is as a model of the distribution of features among species either on the basis of tree branch length or branching points. The use of branch length to scale character accumulation essentially gets away from the potentially perverse results arising from limited interpretation of genealogical divergence based only on tree nodes or branching points. In particular it has been shown that this form of measure is potentially insufficient to distinguish between sets of species which can be described by similar tree structures but which obviously encode different amounts of genetic heritage (Solow *et al* 1993; Humphries *et al* 1995). Nevertheless, the consensus on these models is far from unanimous (Williams and Humphries 1996) and an additional complicating factor in deriving diversity measures.

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[6]Some of the statistical complexities of this and similar structures are discussed in Cummings *et al* (1995a).

[7]The use of these models is in contrast to the aforementioned example of an exact tree constructed by Crozier which simply charts currently existing genetic differences.

The purpose here is to abstract from the relative merits of competing and somewhat controversial methodologies for constructing tree relationships (Williams and Humphries 1996). Recalling the hypothetical nature of these constructs, it remains to show how they are a basis for the assessment of diversity-related option value. Related strands of analysis based on similar pattern models are described in dual contributions by Weitzman (1992) and Vane-Wright *et al* (1991) respectively.

2.3 Phylogeny, conservation and area selection: what are we valuing?

The relevance of phylogenetic tree pattern in area selection has been shown by Vane-Wright *et al* (1991). Essentially, area selection or prioritization will be conducted according to a specific definition of value inherent in a phylogeny. If a species appears to have evolved along a path with relatively few nodes (branching points), then it seems reasonable to suspect that its tip of the tree should be assigned a weight to reflect the likelihood of it containing more uniquely evolved elements than a species at the end of a path with several nodes indicating shared lineage. This preference for uniqueness weights branches of a tree and (ranks species) in proportion to the *total number* of features contributed to a protected set. This is just one value criterion and there are numerous alternatives based around valuing specific *combinations* of characters (i.e basal with highly evolved species etc). It is precisely this subjectivity which precipitates the need to scrutinise the objective of conservation value (e.g resilience).

From figure 1, taken from Williams and Humphries (1994), the first element of the so-called 'agony of choice' is made clear by supposing that an appropriate weighting system would dictate which 3 of 8 species described by a hypothetical phylogeny might be prioritised for conservation - perhaps within a subset of protected areas which can just be accommodated within an existing budget. The diagram shows five alternative taxonomic diversity criteria which essentially vary in the weight assigned to species variously rooted in the particular tree. Equally high scoring choices are bracketed by the dotted line. Groups which are essential to represent under the particular criteria are shown in black (dots). Groups that are alternatives and those of low priority are shaded and white respectively.

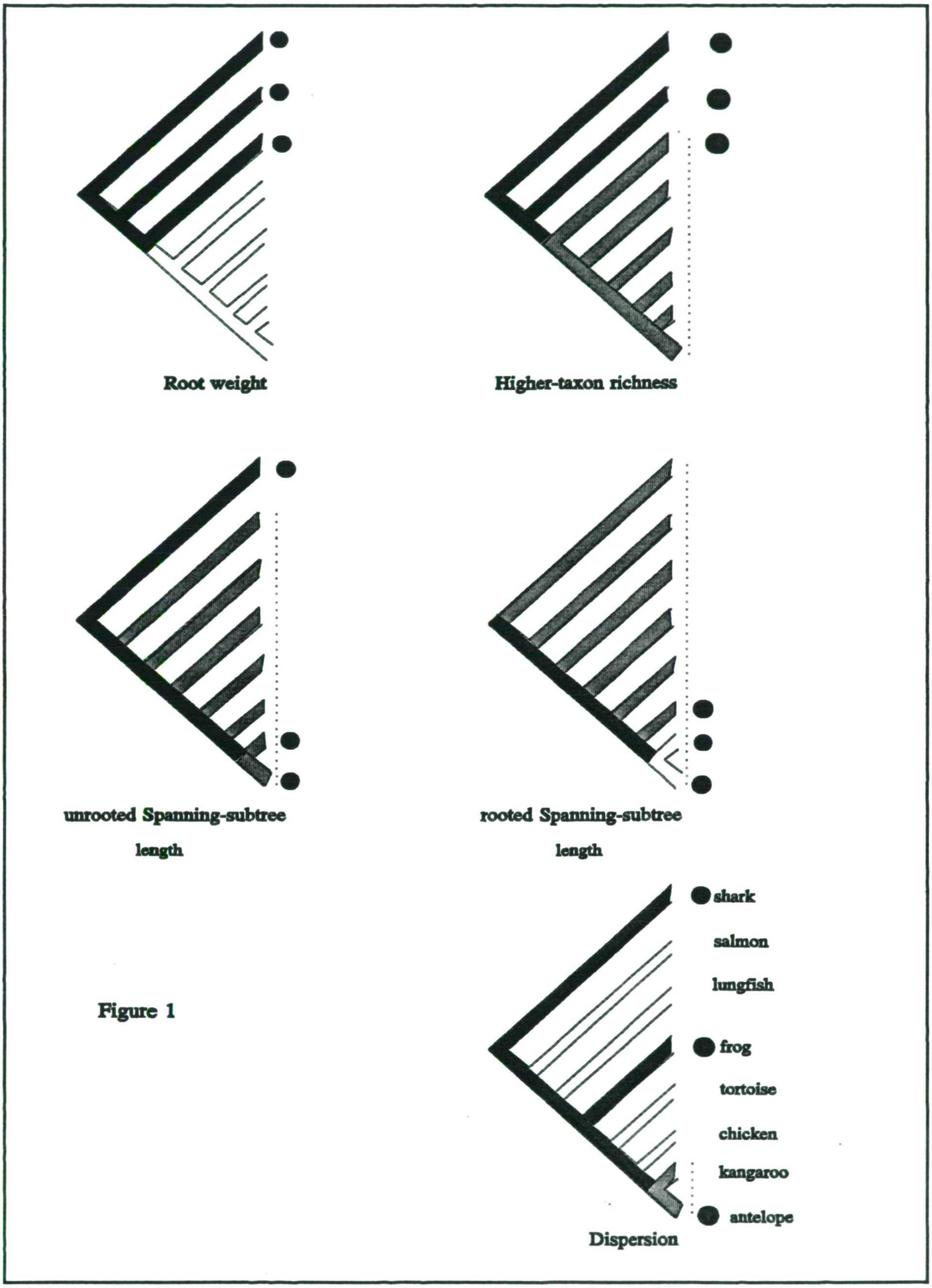


Figure 1

Species conservation criteria covering combinations of total tree length (e.g root weight) or the spread of likely different characters among species (e.g dispersion)

The intuition here is the choice of species which best represents the information encoded in the branches, and more importantly, given real world conservation choices, how to match a specific objective with actual species conservation. On the last point, Williams *et al* show that higher taxon diversity and dispersion (regularity) select for character richness (total number) and character combinations respectively. The desirability of any choice depends on what the objective is. But the higher taxon choice approximates a species richness criteria and might be favoured if the aim is, say, prospecting for pharmaceuticals. Character combinations on the other hand, may be more important than mere numbers. Maximising richness of *different suites* of characters may be more important for ecosystem integrity.

Figure 1 is a simplified example of what can be termed the weighting problem. Given a consensus objective on which parts of the tree to represent, any conservation decision can then be assessed according to how well it meets this objective. The criteria have been combined into numerous other measures developed to summarise the diversity of taxonomic trees. The search for appropriate criteria for representing diversity is on-going. Those suggested so far combine information on branch length and some variant of the relationship defining uniqueness as roughly inversely proportional to the number of internal nodes on an evolutionary path to the tip of a tree thus:

Faith (1992) proposed a measure of phylogenetic diversity which equates diversity of a subset with the tree branch lengths connecting members which draws explicitly on the predictive nature of phylogenetic models. In proposing this measure, he also suggests a complementarity measure which assesses the marginal increment to diversity from adding a third species to a pair. This is a simplification of a generalised but statistically difficult approach later developed by Weitzman (see below).

Crozier (1992) presents a specifically genetic diversity measure (in response to his own criticism of the Faith (1992) measure as not specifically using information on genetic differentiation between taxa). The value of preserving a group of species is measured in terms of the probability that it contains more than one allele which can be related to branch length.

Williams and Humphries (1994) offer a cladistic dispersion measure combining the length of a tree covered by a subset of species and information on the number of nodes between pairs of taxa.

Faith and Walker (1994) have drawn on a family of p-median measures derived in Operations

Research. The measure has desirable properties for locating objects on networks which are similar to metric trees. Basically the aim is to maximise character combinations by maximising the intersection of a set of discs laid over the tree whose radii will define a unique set of character combinations.

Witting and Loeschcke (1995) added a probabilistic slant to phylogenetics by combining weighted node information with hypothetical extinction risk information for the minimization of the expected loss of phylogenetic diversity in conservation decisions. This approach was previously suggested by Weitzman (1992) as a basis for a cost-benefit assessment of extinction.

Solow et al (1993), independently show a basic case where internal node counts are not sufficiently discerning to separate competing subsets.

From an economic perspective, the obvious problem is that all these indicators are value-laden. At this point it is worth reiterating there are three essential assumptions necessary to get this far, and therefore underlying any subsequent economic theory based on phylogeny: first, a currency of diversity; second an evolutionary model of branch length (where exact data is not available); third an appropriate weighting criterion. These assumptions all imply something about value which can be contested. Williams and Humphries (1996) make much the same observation in indicating the contradiction inherent in alternative interpretations of basic phylogenetic information. Ideally, ignorance of how an organism may prove to be of value in the future means there can be no justification for attempting to weight differently the units of diversity value. A possible alternative is the development and use of probabilistic approaches guided by distance information (see below). Weighting introduces subjectivity into what constitutes the basis of either existence or option value. Furthermore, just as models of genetic character distribution cannot guarantee phenotypic diversity, there is no guarantee that weighting to maximise one value category (eg option value) will be consistent with another (eg existence or use). If the interest is in characters expressed in the phenotype of an organism, unexpressed difference (say in DNA) may be of little interest. Yet a valid argument can be made for weighting at any level. Again, an important caveat is that different forms of character diversity are distributed in different ways such that focusing on one may not guarantee another. An excessive focus on genetic characteristics might imply a world where diversity is maximised in lesser plants and animals.

Another related point on the species level relevant to this discussion has been made by Rojas (1992).

She indicates that confused approaches in systematics and evolutionary biology do not lead to the basic interpretation of a species being circumscribed by any consistent definition. In other words, the definition of a species in one taxonomic group (eg on the basis of morphological discontinuity or interbreeding capability), may mean little in the context of another group. This has been neatly summed up by the quote that a species is defined as "what a competent taxonomist says it is," and has little to do with evolutionary potential. From an economic perspective, an alarming implication is simply that the number of existing species will depend on how species are defined (biological species, cladistic species or evolutionary species). Moreover, this identification is relevant to the decision whether or not to use the predictions of phylogenies rather than species richness.

Vane-Wright *et al* relate the weighting problem to the ultimate decision on area selection. Figure 2 shows a structure corresponding to a particular phylogenetic pattern demonstrating the character history of several taxonomic groups. Abstracting from the specific weighting criterion (see Vane-Wright *et al*), 'importance' weights 'W' are assumed given for terminal taxa A through E. Three of the five taxa occur in three areas R1-R3 and, according to the weighting system, row T gives the total aggregate scores for the occurrence of the taxa. For each of the three regions, row P1 gives the percentage diversity score, indicating that R3 is the top priority region. Row P2 gives the percentage diversity scores for the remaining two regions with respect to taxa complementarity - the concept which describes the use of cost-effectiveness to select the area adding the greatest *incremental* species difference to an existing set. In this case having chosen R3 it is possible to see that R1 is the second priority to achieve 100% taxa coverage, rather than 89% coverage had R2 been selected. This simplified example raises important issues related to priority setting. First, that *species richness* might be a less than perfect criterion for selecting priorities. Instead the use of complementarity highlights the need to avoid double counting in character representation, or that the marginal value of a site is the contribution of species of species represented in the site not represented elsewhere. Areas R1-R3 are equally species rich, but the sequential choice matters if a budget only covers the choice of two areas. Second is the related question of cost. The simplifying assumption made here is that R1-R3 are three identical areas that may be acquired at equal cost. However it is feasible that cost differentials might make the most biologically desirable sequence in this example too expensive to implement as an area selection programme. An issue that needs to be simultaneously addressed, therefore is how cost considerations can be integrated into this form of analysis. Methods for dealing with competing objectives encountered in conservation problems may be addressed using standard mathematical programming.

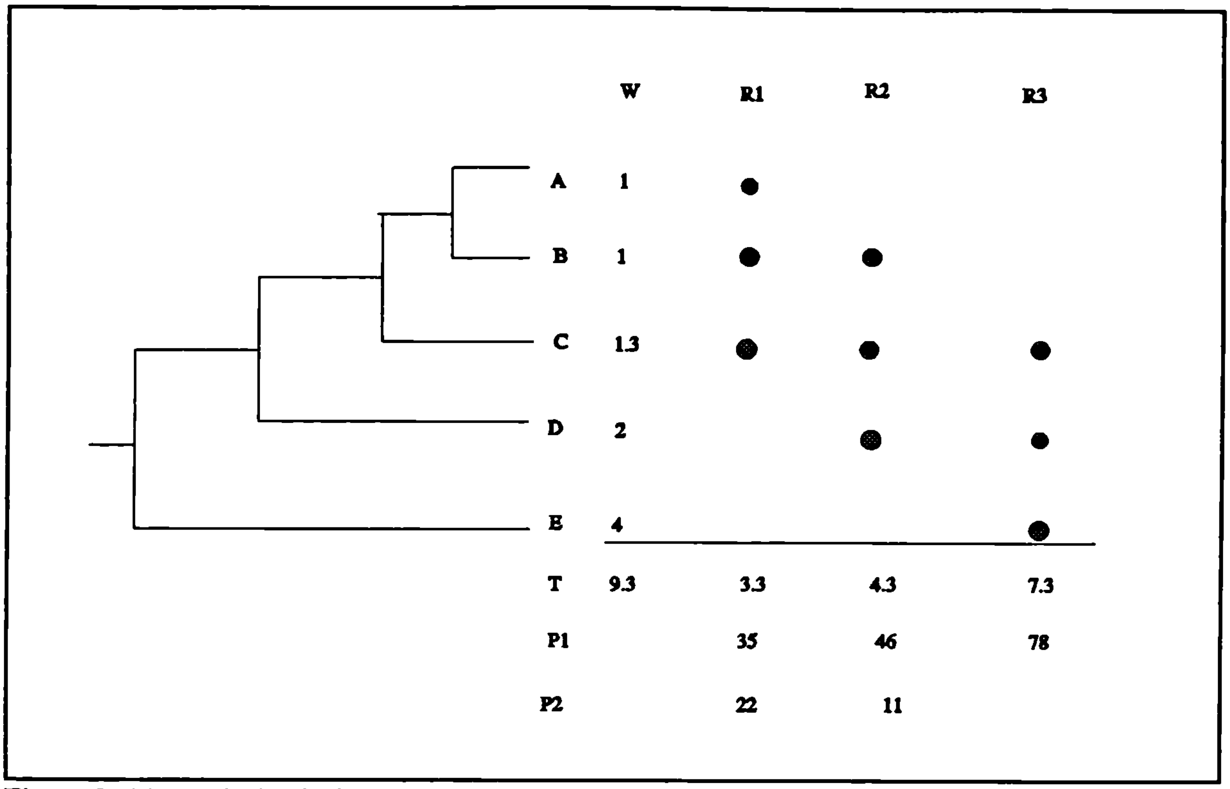


Figure 2. Theoretical priority area analysis (adapted from Vane-Wright *et al* (1991))

The above form of analysis would seem to offer the beginnings of a calculus of option value-related diversity, and gives rise to practical developments on a number of fronts. Most immediately, the selection problem outlined is the basis of recent developments in algorithm-based methods for area selection. At this stage however it is difficult not to agree with Harper and Hawksworth (1994), who question how sufficient phylogenetic data could be generated in the near future in order to compare the diversity of whole countries. They imply that traditional bottom-up classification obscures a bigger picture approach to conservation served by focusing on several geographical scales. The development and use of surrogate information is a recognized alternative approach. Surrogacy can be applied at various scales. The predictive models used in phylogeny themselves forms of are surrogate methods because all characters cannot be counted. As previously mentioned, the use of species richness information may be sufficient for character richness or diversity and it is interesting to address the economic criteria relevant to conservation decisions on this basis.

2.4 Towards an economic theory of diversity

Diversity measurement implicit in phylogeny serves as an appropriate (if complex) metaphor for a general theory on quantifying diversity. Part of this complexity emerges from the necessary assumptions in tree construction relating to the evolutionary process of how characters emerge. This section will examine the basis for constructing an economic theory of diversity from similar information. This framework can be used to place the extinction issue in a theoretical cost-benefit framework. To do this, it is necessary to express phylogenetic information in terms of a diversity function which is maximised subject to given survival probabilities and a constraint. The numeraire of value implicit in this approach is pairwise distance measured in terms of summed dissimilarities in some arbitrary character.

Developing such a model is the task first undertaken by Weitzman (1992). Defining a set of internally consistent conditions to describe the diversity of any summary matrix of pairwise distances, Weitzman arrives at a structure which is dual to a phylogenetic pattern model. The hypothetical distances in question arise from an evolutionary model which recent biological advances show to be unwarranted. Accordingly Weitzman's method is at best only a lower bound on the diversity of a set. The approach is illuminating in terms of the economic conclusions, as well as again highlighting the sheer complexity of the problem.

The bead model

The tree-like structure derived by Weitzman represents an evolutionary branching process ultimately giving rise to sub-set entities S (species), acquiring and discarding beads (the unit of constitution), as they move away from a common ancestry. The beads are analogous to characters in the phylogenetic model above. The analogy of a set of species evolving by descent is set out in figure 3, which shows 6 species as twig tips evolving from common ancestor. Thinking of each these species as an individual, each consisting of a constant number of beads, evolution *ideally* proceeds by a process of simultaneous accumulation and discarding of beads per unit of time. New species occur at a bifurcation when one entity becomes two, with each new individual henceforth accumulating and shedding unique beads. The difference between 2 species is defined as the distance back to a common ancestor corresponding to the time elapsed over which beads have been independently accumulated. This constant process of character change represents the important property of an *ultrametric* tree.

More formally, between any pair of elements ij, $\in$ S, a dissimilarity or distance (d) measure based on the respective accumulation of beads is such that:

$$d(i,j) \geq 0;$$

$$d(i,i) = 0;$$

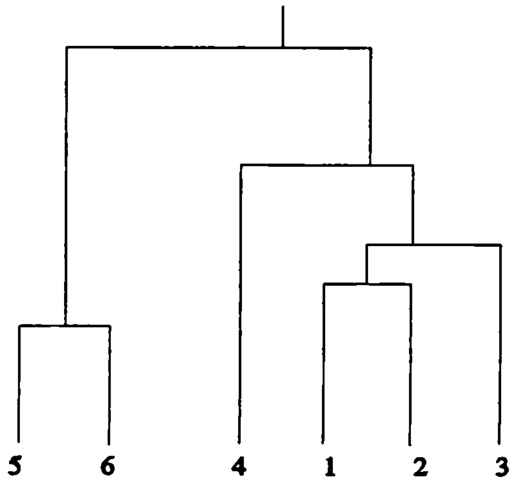


Figure 3 An hypothetical taxonomic tree for six related species.

$$d(i,j)=d(j,i), \quad \forall\, i,j \in S$$

For a collection of species an interpretation of diversity of the set S then refers to the total number of unique beads which arise in the given evolutionary process. Under an ideal model this also corresponds to the length of the vertical branches in figure 3.

Given a group of species S, we may typically be interested in the extinction of one member, or the point to set relationship[8]. By extension, the loss of diversity resulting from the extinction of a species is the loss of the branch length connecting that species to its nearest ancestor. Note that all of this is predicated on just one model of tree evolution which gives rise to ultrametric distances. This distinction leads to the key flaw in Weitzman's approach.

To formalise further the requirements for a diversity measure, certain axioms are required (Weitzman 1992, Solow *et al* 1993). In general two basic *monotonicity* axioms should apply to the set. First, the diversity function V(S) is monotonic in species. If species j is added to sub-set Q of S then:

$$V(Q \cup j) \geq V(Q) + d(j,Q) \quad \forall Q\ ,\forall_j \notin Q$$

where d(j,Q) represents the distance j to its nearest relative in Q. Alternatively, using an evolutionary metaphor, V might equally represent the length of the process giving rise to Q. We are interested in the number of additional, as yet unaccounted for, characters arising in the additional evolutionary steps given by d(j,Q), that is, adding j to subset of S of Q. Second, monotonicity in distances basically conveys the idea that Q will be more diverse the greater is d(j,Q).

To skip a few steps to the general result, it can be shown that the desired diversity function V(S) is inductively defined to be the solution to the recursion:

$$V(S) = \max_{i \in S} \{V(S\backslash i) + d(i, S\backslash i)\}$$

where (S\i) denotes the set S excluding element i. To get to this result, Weitzman makes a number of assumptions which tighten the monotonicity condition to yield the above as a unique dynamic programming equation. Specifically, viewing the monotonicity condition as a large set of constraints that must hold for all Q and all j (given the initialising condition), simply define diversity as the

[8]An initialising condition for this discussion is a convenient normalisation such that:
$V(i) = d_0 = 0$, for all i

minimum V(S) to satisfy both monotonicity and the recursion. In other words, for S, the induction process will construct V(S) (a rooted genealogical tree whose twig tips represent existing species) as the most parsimonious feasible reconstruction of S or the minimal number of character state changes required to account for its evolution. That is:

$$V(S) = \text{minimum } V$$
$$\text{s.t. } V \geq V(S \backslash i) + d(i, S \backslash i) \ \forall i \in S$$

The way to think about this over all set members is in terms of the increments of the diversity function $\{V(S \backslash i)\}$ being built up progressively for all i belonging to S. More rigorously, saying that diversity of set S, V(S), is the most parsimonious reconstruction of S, is equivalent to saying that V(S) minimises the length of the evolutionary process giving rise to S, subject to the monotonicity constraint. The solution to this is the above recursion. In the recursion process that satisfies these conditions, Weitzman proves a pivotal role is played by two nearest neighbours in S, for example, the pair (g(S),h(S)), one of which is always 'cast out' of the set to form the first tree increment mentioned above. In successive iterations over S, these two elements are identified and the distance d(g(S),h(S)), between them measured. Given these two elements the recursion becomes :

$$V(S) = d(g(S),h(S)) + \max \{V(S \backslash g(S)), V(S \backslash h(S))\}$$

Now, if g and h are not identical, only one of the pair will uniquely satisfy the rhs maximum. If this turns out to be g, then we know that $V(S) = d(g(S),h(S)) + V(S \backslash g(S))$, of which d(g(S),h(S)) is known. For the second right hand side term $V(S \backslash g(S))$, the same procedure can be reapplied, but to a set of dimension n-1, where n is the dimension of the original set S. Suppose the next two nearest neighbours in set $(S \backslash g(S)$ of dimension n-1, are $g'(S \backslash g(S)$ and $h'(S \backslash g(S)$. If g' uniquely satisfies the maximum in the recursion formula, then in the second stage is $V(S \backslash g(S) = d(g'(S \backslash g(S)),h'(S \backslash g(S))) + V(S \backslash (g \cup g'))$ where the unknown is again the second term $V(S \backslash (g \cup g'))$ to which the recursion is applied, and so on.

The geometric (rooted tree) interpretation of the programming procedure is shown in figure 4, with tips representing existing species. At each iteration there is a pair clustering of the two nearest elements in the set, and a pair-group clustering given by the diversity between the furthest pair and the rest of the set. For each nearest neighbour, consider the bit that is left out each time as constituting an off-shoot the size of the distance (or mismatch) between the nearest neighbours of S at that particular iteration. This is attached to the 'representative' branch of the remaining neighbour, itself still linked to the group cluster. Because this distance is added to a running sum V and the next iteration is n-1, it is possible to see how the tree length is constructed to be V(S), leading back to an ancestral root.

The diversity between 3 species can demonstrate the above. In figure 5, the distances a,b and c between 3 species is shown such that $a < b < c$. The corresponding evolutionary tree is constructed as follows. Pair 1 and 2 are the closest relatives - separated by evolutionary period(s) a - and are clustered together with the remaining element 3. According to the recursion we now want to choose the maximum distance between 3 and the pair 1,2 which is given by c. Therefore the total branch length of the evolutionary tree is $a+c$.

The main advantage of this summation of the diversity of a set of species is the ease by which the diversity of a species lost from a set can be associated with the length of an evolutionary branch. Moreover, this definition produces a number of desirable properties of the measure itself, its basic duality to taxonomic principles and, not least, its economic significance which will be made clearer below. The first of these is evident from comparison with the biological literature discussed earlier and is not discussed further (see Weitzman 1992 p390-393). The important taxonomic statement which requires attention, is that ultrametric distances reduce diversity theory to perfect taxonomy theory. Basically this relates back to the model assumption about how characters are accumulated along branches. The relaxation of this assumption is important, as ultrametric pairwise distances imply that the rate of evolution (the rate at which new features arise) is constant over all branches. It seems reasonable to investigate the implications of alternative models of evolution; in particular, the case where the rate of character divergence along a branch has nothing to do with its relative length.

In the preceding discussion the extent of the inequality in the minimisation of V is of considerable importance and essentially the difference between the perfect world of ultrametric distances (producing an exact equality) and the generalised condition which provides a somewhat imperfect lower bound on the diversity of the set[9]. It turns out that the diversity of a set can be measured exactly by tree length (and full equality in monotonicity can only hold consistently) for the unrealistic assumption of ultrametric distances. The precision of this diversity measure depends crucially on the assumption of ultrametric distances between species in the tree. In other words, branch lengths calibrated by constant rates of character accumulation. Such an assumption may not be an accurate reflection of

[9]In other words, set diversity will be reduced by the extent of duplication implied by the inequality.

the evolution of a set of species and the number of additional features may not be related to the distance to a nearest neighbour. More specifically, Faith (1994b) shows that it is possible to construct a plausible evolutionary tree pattern under this alternative assumption such that the Weitzman algorithm can be shown to double count parts of the resulting hypothetical structure. This is the imperfect 'lower bound diversity measure mentioned previously. In other words, with such an evolutionary model, the diversity measure is not so conveniently built up from the diversity of subsets and the branch length (distances) of species to the subsets implied by the use of V(S) above. Furthermore, the use of Weitzman's method may actually construct the wrong taxonomic tree when exact information on a distance to character relationship is available.

A convenient exposition of some of these problems has been provided by Faith (1994b). In figure 6a, the unequal branch lengths suggest divergent rates of evolution and character representation in the 3 species A-C. Distances $a>b>c$ are all distances from the respective species to the point p, and distance A to B is $a+b$. Applying Weitzman's method to such a structure yields:

$$\begin{aligned} V(S) = k + \max \{&b+c + \min (a+b, a+c),\\ &a+c + \min (a+b, b+c),\\ &a+b + \min (a+c, b+c)\}. \end{aligned}$$

of which - counting up the dashed characters in the diagram - the last element max (12+ min(10,5) = 17) is the largest. This suggests that this measure of diversity does not in fact correspond to tree length, while the fact that $V(S) = k + a+2b+c$ shows that the intermediate length is counted twice. Similarly figure 6b shows a tree in which the initial distances reflect differences in the number of features and the length of branches reflect the number of changes in features. The most parsimonious tree accounts for each new feature having arisen once along the tree topology shown. Given the corresponding differences between pairs of species, Weitzman's method returns the tree in figure 6c below, showing that the ultrametricity assumption is a poor basis for the inference of phylogenetic pattern.

Weitzman (1995) recognises this drawback and presents the method as a lower bound generalisation of the simple model, allowing for the likelihood of character duplication when quantifying the

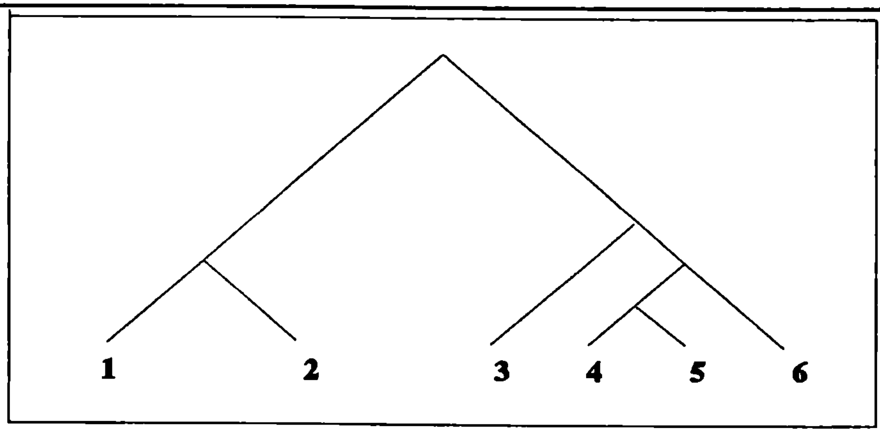


Figure 4

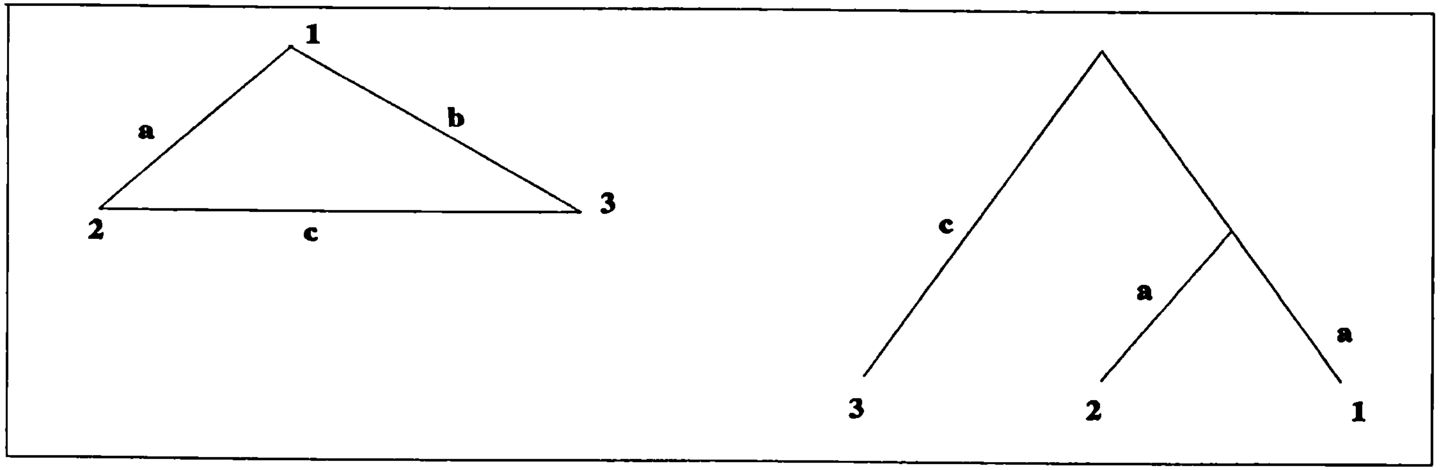


Figure 5

The geometric interpretation of Weitzman's recursion with pairwise clustering of the nearest neighbours and maximised distance to the remaining element.

diversity of a group of species under the assumption of ultrametric distances[10]. Basically, this bound suggests that Weitzman's diversity measure is not so compelling for use as an index of diversity. By extension, Faith (1994) also concludes that a measure of phylogenetic complementarity is the only way to measure the distance from a species to a set, and that this measure can only be calculated using pairwise distances to exactly calculate the complement of character differences by adding the new species. Clearly if the number of species in a set is large, the calculation is somewhat complicated as it would have to be carried out recursively. An algorithm for a large number of species would be highly cumbersome.

Even if Weitzman's measure is technically limited, the important contribution is the necessary conditions for a diversity measure inherent in the approach. A remaining challenge is therefore to integrate these desirable properties with a more explicitly utilitarian rationale for measuring diversity. Prior to this, it seems reasonable to outline the basic CBA decision framework for which the approach was intended.

Diversity theory in a cost-benefit framework

An ideal diversity function might maximise the present value of expected diversity given some existing notion of the probabilities of extinction of the individual components of S. The function would be the sum of the deterministic diversity function of various collections of species weighted by the existence probabilities of the various collections. It is helpful to think that these probabilities can be influenced by conservation spending patterns, for in reality the problem posed is the maximisation of the present value of expected diversity subject to a budget constraint. Weitzman (1995) considers one example of such a function based on the existence of some idealised tree like figure 3, again under the ultrametric case. A hypothetical conservation decision involves the fate of the two most closely related species, 5 and 6. The extinction of one of these species incurs the smallest diversity loss, but the extinction of both will incur the loss of a whole evolutionary line. With the aim of maximising expected diversity, the optimal strategy might be to concentrate relatively few resources on saving 5 if 6 is safe, or it may be to concentrate large amount of resources on 5 if 6 is in high danger.

[10] This problem has been recast in a couple of helpful metaphors about the cost of making diversity or the amount of redundancy encountered in making a hierarchical search (eg through a hypothetical library that has evolved according to a tree for a specific character string) (Weitzman 1995).

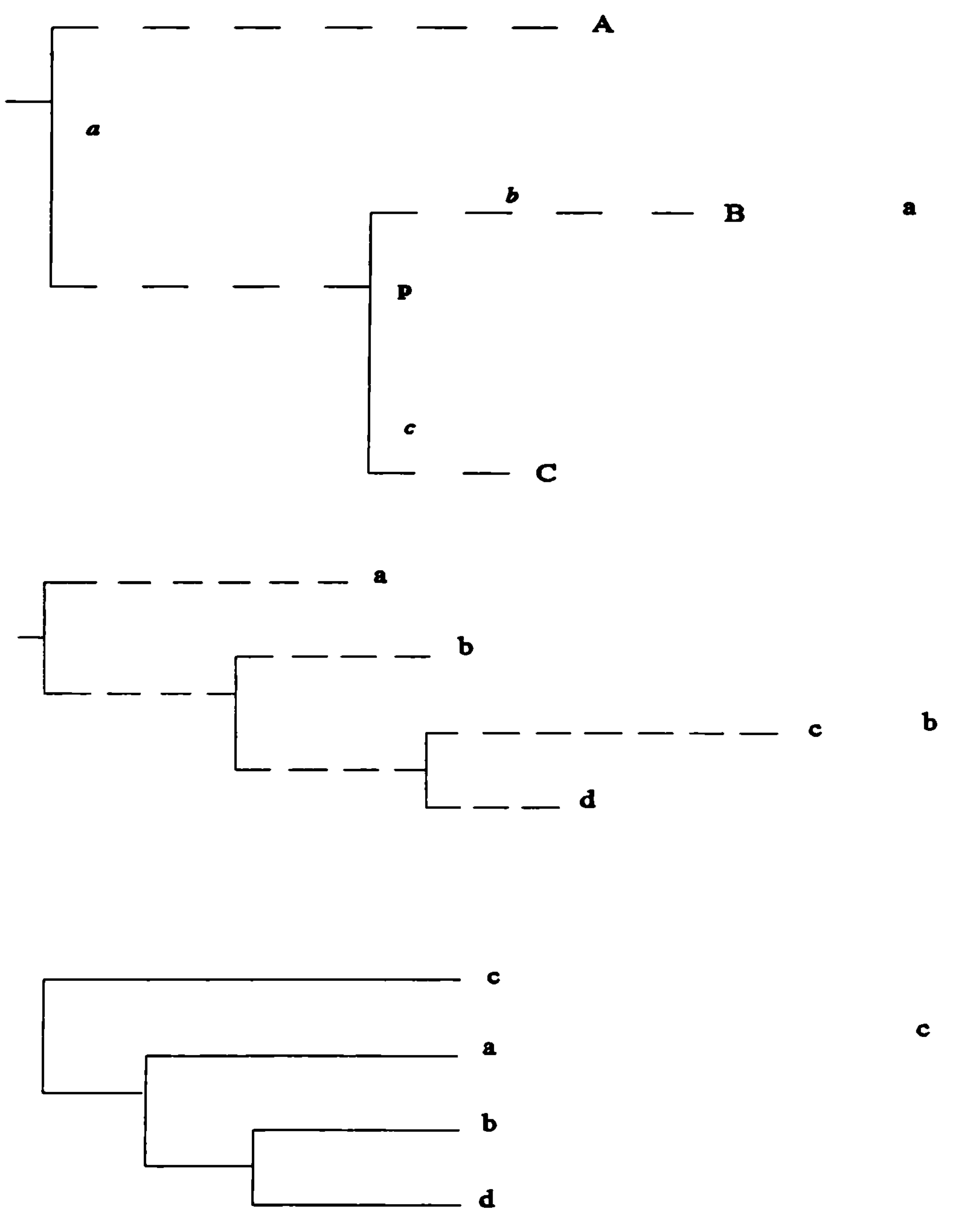


Figure 6

Contradictory results from Weitzman's algorithm. For the metric tree 6a the recursion reveals double counting and a diversity measure which is lower than the actual number of character changes (dashes) which have occurred (i.e at best a lower bound diversity measure). For 6b the recursion will erroneously imply the form 6c.

Suppose survival probabilities[11] of 5: $p5 = .98$;
and for 6: $p6 = .02$

The question is which of the 3 alternatives maximises highest expected diversity?

	p5	p6
1. status quo	.98	.02
2. endangered species 6 more endangered	.99	.01
3. endangered species less endangered	.97	.03

Conservationists might be expected to favour option 3, making the relatively endangered species safer. But this minimises expected diversity, whereas diversity is in fact maximised by option 2.

Under 2, the probability that both survive $= .99 * .01 = .0099$;
under 3 $= .97 * .03 = .0291$;
thereby favouring option 3.
But the probability of extinction of both is:
under 2 $.01 * .99 = .0099$
under 3 $.03 * .97 = .0291$

This smaller probability of both being extinct thus favours 2 over 3, in other words making the safe species safer, at the expense of making the endangered more endangered.

To further formalise the decision framework, consider S as a collection of n species denoted by $i = 1....n$. The independent probability of i surviving is x_i. Each column n vector $X \equiv (x_i)$ of survival probabilities defines an expected diversity function:
$U(X) \equiv E_x(V)$.
The objective function might be:
$\phi(X) = BX + U(X)$
where B is a vector of net benefit coefficients associated to saving species (direct, indirect etc).
Suppose the cost of conserving i with probability $x_i = cx_i$.
Given a n row cost vector of cost coefficients $c_i \equiv C$, then the basic problem is to maximise $\phi = (X)$,

[11]We assume a one to one trade-off in survival probabilities, whereas there is every chance that two closely relate species may not be independent.

subject to $CX \leq A$ (a preservation budget), given $0 < x_i < 1$; $i = 1....n$.

For the ultrametric distances, an efficient solution to such a problem will use the "nearest neighbour" algorithm described earlier. The solution proceeds by eliminating the least valuable species until the budget constraint is met. If at some iteration the subset of species Q exists with a probability of 1, while S\Q exist with a probability of zero, and Σ cixi $> A$ (for all i in Q), the procedure selects the least desirable species in Q, j(Q) satisfying:

$$\frac{bj+d(j,Qj)}{Cj} = \min_{i \in Q} \left(\frac{bi+d(i,Qi)}{ci} \right)$$

Thus the probability x_j of the species adding least direct benefit plus distance (diversity) to Q (measured as the distance from it to its nearest ancestor in Q), is brought down from 1 towards zero, until the constraint is satisfied or the species is extinct, whichever occurs first. If need be, the same procedure can be worked on the remaining set Q\j and so on, on the understanding that, at each stage, the remaining matrix of distances may be changed by the absence of a previous member. It is possible to think of numerous additions to this basic cost-benefit exercise. For example, it might be reasonable to relax the unrealistic view of survival probabilities, perhaps to reflect the possibility of joint probabilities, and so on. At present however, there is limited information about which species are present or absent in given locations, much less whether species would be expected to survive in the same locations under current or hypothetical conditions. However, as Weitzman points out, the approach goes some way to reflecting the global significance of local decision making.

At this point it seems appropriate to summarise the main advantages and disadvantages of Weitzman's approach which has been highly influential on subsequent economic enquiry. The main advantage is the evident link between taxonomy and, albeit in a limited way, to the question of valuation. As it stands, Weitzman proposes a cost-effectiveness criterion and it remains to be seen how a value is associated with the numerator in the exercise. Depending on the value attached to distance and the reliability of the tree model, the information provided by phylogenetic structures is akin to an index number or preference ordering suitable for guiding cost-benefit decisions. The approach has many disadvantages. Basically there is still no reason to value diversity. Without the attachment to a revealed value one is left to speculate about several motives including bequest, option and existence. Moreover while there is no specific reason to hone in on a unit of interest, the biological unit of interest remains arbitrary. Wilson (1992, pp. 73-74) states that genes are the ultimate currency, and as Crozier and Kusmierski show, phylogenetic relationships at the most fundamental levels can guide economic decisions. However, as Weitzman's approach demonstrates, the use of predictive models can be potentially misleading and the data requirements are formidable. Consider the informational requirements implied by the recursion formula, which might then translate into a related land use decision (eg designation or otherwise). Recall from chapter one that at current rates of classification, there is some certainty that S will be unknown. By extension a comprehensive distance matrix which characterise S[12] will also not exist. In such circumstances, some degree of set surrogacy may be the most practical approach, and it turns out that surrogacy may in fact have a compelling scientific rationale.

2.5 Linking diversity and value

How much would a comprehensive measure of diversity actually tell us about the value of diversity? A diversity measure would certainly be practical were it acting as an index of the value motives one supposes to be attached to diversity *per se*. But the location of value in unexpressed characters creates obvious problems in validating any associated WTP. The basic economic question arising from the foray into the phylogenetic literature arises in deciding how to tie distance information to a specific quantifiable economic motive.

From a biological perspective (e.g. Faith 1994ab), the kind of diversity measure discussed here is more closely associated with option value. As previously mentioned, two specific (albeit arbitrary) interpretations of option value are the potential direct use view and the indirect use view, the latter associated with system resilience. Cost-benefit decision theory can already accommodate adjustments for dealing with the issues raised by asymmetric development and option value (Beltratti *et al* 1993, Perrings and Pearce 1994), and it is questionable whether the information provided by distance measures does anything other than complicate the analysis. This is particularly true if one is able to consider distance measures at infinitesimally small scales and end up with competing but equally diverse sets at several levels of the biological hierarchy.

It is worth noting therefore, that the strict association of V(S) with any value category is highly circumscribed by: a) basic preference uncertainty which prevents any clear association of distance *per se* with any form of use or non use value; but also, b) the uncertainties in defining cladistic relationships which lead to a subjective interpretation about the location of option value and resulting conservation choices. Moreover to build on the information available in phylogenetics it is necessary

[12]This species problem (ie the number in existence) is a general umbrella term in applied statistics for the use of sampling theory to estimate the number of unseen classes from a finite but large population (see Bunge and Fitzpatrick 1993, and Bunge *et al* 1995). For a review of other estimation methods see Stork (1993).

to be somewhat arbitrary in the treatment of all the very real motives for valuing diversity in order to focus on how any of them might be quantified.

Use-oriented option value

Empirical estimates of option value are strikingly absent. Of the two broad descriptions of option motive; relating pure diversity to potential use value at least allows a reasonable probabilistic assessment to motivate potential choices of subsets of species. In other words, a set may be characterised as more diverse by virtue of a greater probability that it contains some desirable property such as a cure for a disease. In theory, this probability can be calibrated using appropriate distance information described above.

Such an approach improves on a considerable number of attempts to quantify the returns to habitat conservation for pharmeceutical value (Pearce and Puroshothaman 1992; Mendelsohn and Balick, 1995) or non timber forest products (Peters *et al.*, 1989). In short, the assumptions underlying such studies have been somewhat unrealistic, leading to the conclusion that these values cannot be regarded as the main economic argument for conservation (Moran and Pearce, 1997).

Brown and Goldstein (1984) appear to have offered the first probability-based assessment of the returns to diversity in a theoretical model of the marginal value of a wild seed variety. The value of a specific variety can be related to its substitution possibilities by a commercial counterpart (in the event of damage from pathogenic agents), and time to failure of a specific character in the same counterpart. The former is ultimately linked to the total stock of species with the cost of actually holding a larger stock offsetting the benefits. The data requirements for the model are extremely large. For example, the model has to assign a probability of failure to each useful characteristic in a set for the counterpart. Furthermore the model puts a premium on richness rather than diversity and does not use distance data.

Polasky *et al* (1993) and Solow and Polasky (1994) do attempt to use such information. A species in any set may contain a specific cure for a disease, the properties of which are that having more than several species provide the cure is no better than having one species provide it. A model to compare competing sets may be constructed and calibrated using any available distance information relating member species.

Assume T to be the total set of species under consideration, $T = (s_1,..,s_n)$, and S the conserved subset we assume to be successfully influenced by a conservation strategy. S will be of value if it contains a cure which might be the case with a probability P(S). Option value is then the product of this probability and the value of the cure C, or in terms of a conditional probability = CP(S,T)P(T), where P(S,T) is the conditional probability that S contains a cure given that T contains a cure. The elements of T are assumed to be characterised by a matrix of metric distances, and the desirable cure is assumed to be enclosed in a sphere arbitrarily centred within radius R distance of at least one element of S. The volume of this region for fixed R=r is V(r;S) and, assuming a similar region V(r;T), the conditional probability that S contains a cure given that T contains a cure is p(r;S,T) = V(r;S)/V(r;T).

For given assumptions about relative dissimilarity of the sets, it is possible to picture the conditional probability as the relative size of two spheres dependent on the size of r (see Polasky *et al* 1993). For instance, if r is small enough to encompass only the immediate environment of each element (ie species are independent in the sense that any one may be the unique repository of the desired characteristic), the conditional probability is simply the ratio of the number of species in S to T. Alternatively, it is feasible to have a configuration of r and distances in the 2 sets that imply a conditional probability of 1.

With known r, it is possible to rank subsets S_1 and S_2 by comparing their resulting conditional probabilities (the relative volume from the combination of set distances and r).

Where R is unknown and no subset automatically dominates in terms of being more distant in euclidean space or simply having many more species but returning the same score, it is necessary to specify a distribution for R. This provides a distribution which can be scaled by the matrix of distance of each competing set and a given value for r to enable subset comparison using:

$$p(S,T) = \int P(r;S,T)\, f(r)\, dr$$

where $f(r)$ is the probability density function.

Applying this method to a matrix of distances between glucosinates generated by principal coordinate analysis, Polasky *et al* (1993) separate competing sets S_1 and S_2 by calculating P(S,T) for $f(r) = b\ exp(-br)$, choosing two alternative values for b. The approach shows that lower values give higher weight to values of r, weighing dissimilarity more heavily than cardinality (the number of species), or high values of b give lower weight to dissimilarity in favour of cardinality.

The measure P(S,T) can be shown to be consistent with the basic criteria of monotonicity in species and twinning, but not monotonicity in distance (see Solow and Polasky 1994). The approach relates a specific view of option value to a measure of diversity. In terms of providing a unique diversity measure, a comparison with Weitzman's measure reveals that these can return conflicting rankings. The basic reason for this is that P(R,S,T) returns a measure dependent on a the total species configuration plus a form for f(r), which in combination weigh similarity against cardinality. Weitzman's approach, focuses only on the nearest neighbour distances, and can be unaffected by changes in set configuration which change diversity but leave the nearest neighbours unaltered. The advantage of this measure of diversity is that it attempts to account for aspects of substitutability which may be as relevant (as distance) in terms of utility. The significance of this point will be made clearer below. On the other hand, a distinct disadvantage (in moving away from cardinality) is that f(r) needs to be specified. Whatever specification is chosen is arbitrary and most likely unwarranted in biological terms. In particular, Faith (1994a) has commented on the arbitrary nature of the linear response of features to the space rather than some unimodal or sphere like response which is equally feasible.

Solow and Polasky (1994) retain much the same model structure for an extension of this work, which provides an interesting statistical bound on the probability of any event of a cure from a set of species $S = (s_1,..,s_n)$. The event B_i that s_i is a cure is given by:

$$B(S) = \bigcup_{i=1}^{n} B_i$$

The expected benefit of S is the value of a cure times the probability $p(S) = Pr(B(S))$. Thus *assuming the value of the cure to be fixed*, p(S) is a basis for comparing sets of species.
Without any further information on the set, it is reasonable to assume $Pr(B_i)$ = unknown p, i = 1,2,.....,n, and further that:

$$Pr(B_i | B_j) = p + (1-p)\, f(d_{ij})$$

where f is a known function satisfying conditions:

$$f(0) = 1, \;\; f(\infty) = 0, \;\; f' \leq 0.$$

The intuitive sense that links this approach to the previous one, is that the conditional probability of B_i given B_j declines from 1 to p as d_{ij} increases from 0 to ∞. As in the previous model, it is possible to define a function for f(d) (eg $f(d) = \exp(-\theta d)$, $\theta > 0$).

The intersection of the events B_i with B_j will be given by:

$$Pr(B_i | B_j) = p^2 + p(1-p) f(d_{ij})$$

In general, it is not possible to find p(S) from the univariate and bivariate marginal probabilities. However it is possible to state a lower bound on p(s) using a result due to Gallot (1966) to approximate the lower bound probability of a union of events A_i (which is not the same as p(S) itself).

$$Pr\left(\bigcup_{i=1}^{n} A_i\right)$$

The expression for the bound is not reproduced here, but again depends on the statement of a matrix of elements $f(d_{ij})$ satisfying the above conditions which ensures that the bound is an increasing function of diversity. For example, if $f(d_{ij}) = 0$ for all $i \neq j$, that is, all species are unrelated, the diversity measure becomes equal to the number of species and, conversely, for perfectly related species the same measure approaches 1. As in the previous model, the main disadvantage of this approach is that it assumes knowledge of the function f. Solow and Polasky suggest that the best method for dealing with this may be just to assume that the function has a simple parametric form and to view the resulting diversity measure as a family of measures indexed by the variable parameter. Alternatively, they conjecture that 'sufficient information may be available to approximate this function reasonably well'. However, considering the difficulties involved in generating distance information, the problem of the form of f is apparently an additional, albeit theoretically elegant, complication. Furthermore, these models seem equally tied to ultrametric distances.

Characterising bioprospecting

A further, somewhat unrealistic, implication of these models is that they do not reflect bioprospecting behaviour[13], the means by which pharmaceutical value is realised. The implication of the sphere model described above is that members of the subset of species within this neighbourhood are

[13]An excellent overview of the bioprospecting industry is ten Kate (1995).

characterised as perfect substitutes. This may sometimes be an accurate biological approximation, but translating this into the expected value interpretation of option value is likely to be an inaccurate generalisation, and certainly does not reflect the observed behaviour of companies active in bioprospecting. In reality, what is observed is that bioprospecting may involve investigation of both close and distant relatives of species[14]. Furthermore, it is reasonable to suppose that processing cost, effectiveness, and other qualities will be differentiating factors impinging on the value of an ultimate cure, which should be reflected in any realistic model.

Polasky and Solow (1995) go some way to addressing these realities in proposing possible modifications to basic probabilistic models of the value of a set of species. Their model explicitly addresses a range independence/substitute cases which might more accurately characterise the neighbourhood of a useful species feature. Coincidentally, a model proposed by Simpson *et al* (1996) focused on the probabilistic basis of previously mentioned pharmaceutical studies, showing that while they make important contributions to an understanding of the industry, they fall down on their (ex post) treatment of potential redundancy in sampled compounds. This argument is developed in the specific case of the value of a marginal species (and by extension the incentives for the conservation of the marginal hectare), and a process to describe the behaviour of prospecting agents. In essence, in the search for a particular characteristic, a species either has it (is a "hit") or it does not. Once one species has been found to have this characteristic there is no value in finding it in other species (all hits are perfect substitutes). The search through many species will encounter redundant resources which are not scarce and therefore should not (as is the case of cruder studies) be part of the equation when grossing up the value of prime habitats. This redundancy simply increases the search and eats into the net revenue of any ultimate 'hit' which as a result has a low marginal value. Basically when there are many species, the marginal value of any one has to be low. By contrast, if the hit probability is high, then some other species will have the characteristic. Either way according to the authors the marginal value is low, thus reiterating the need for alternative conservation arguments. This assessment is in stark contrast to back of the envelope estimates, multiplying the probability with which a randomly sampled organism contains *some* commercially valuable chemical compound (whether unique to that organism or not), by the expected value of a commercial product.

In relating revenue to 'hit' potential, the model of Simpson *et al* reflects redundancy by assuming a binomial hit terminates the search process. In other words, species are considered to be independent

[14]An example of this is the screening of relatives of the pacific yew tree *taxus brevifolia*, which was the original source of the anti-cancer agent taxol (see Day and Frisvold 1993).

rather than sharing any commercially valuable traits. As Polasky and Solow show, it is possible to think of hits as imperfect substitutes. They suppose instead that a hit allows a draw from a distribution that determines the value of that particular hit. Having multiple hits is valuable because there is some chance that a more valuable hit will be found. In this model the marginal value of a species need not decline to insignificance nearly as quickly as in the limiting case suggested by Simpson *et al*. Polasky and Solow (1995) essentially generalise this case for different substitutability and independence assumptions, using both the hit probability structure and value parts of the equation[15].

For the case of a perfect substitution, 'single hit' model, where different species may provide the same benefit, value will be product of value V and an expression that the collection S of m species will contain at least one species providing the benefit. Thus the expression

$$V(S) = V(1 - (1 - p)^{m})$$

does not explicitly alter the assumption of value being subject to attenuating factors such as effectiveness or the search process costs.

It is possible to devise a modification to the above model which associates the draws on a useful species to a specific distribution of values. In this 'multiple hits' model where species in the set are not exactly perfect substitutes, but do provide some value, the expected value can be determined by the useful species with the highest value. An extension is that species may be independent rather than substitutes. This can be reflected by a binomial probability function for a species being useful related to a distribution of values and so on. Basically the result is that the expected value is not straightforward once models attempt to characterise possible independence and substitutability in usefulness.

Given the almost limitless supply of compounds derived from the all species the complexities involved in these models become more apparent. Any information on distances is relevant for inferring degrees of substitutability of species and the relative value of collections. For example, if species are dependent (as characterised some criteria inferred their distance matrix which implies shared traits), then the assumption of a binomial distribution may not appropriately characterise the probability of

[15]That is, whether a cure is unique to one member of the collection, or common to several but in varying degrees of quality and cost.

finding a useful species. Solow and Polasky (*op cit*) show, in a set, the variance of a distribution for any alternative to this probability assumption can be made a function of the distances. This link between the expected value of a collection of species and substitutability indicates an immediate economic reason to focus on the biological relatedness or 'distance' between species as a potential indicator of diversity value. The interesting trade off raised by such models can be investigated for any arbitrary assumption about the hit structure. In the absence of specific model information, using a normal approximation they show that when the expected number of useful species (the variance of which is a function of distances or the extent of dependence) increases, the marginal increase in the expected value of the collection declines. The rate decline is dependent on the distribution for this model but can in some circumstances suggest that the expected value of a collection will be smaller under dependence than under an assumption of independence. This is apparently analogous to a suggestion by Rothschild and Stiglitz (1970) on risk, and suggests *ceteris paribus* that more diverse (ie independent) species are desirable than less diverse collections.

2.6 Applied diversity theory

A natural use of the aforementioned probability models is their extension to decision making via species-area curve relationships of the type mentioned in chapter one. Several caveats are necessary for these models to reflect more accurately conservation problems. One is the relationship between set size (species richness) and distance, which is assumed independent. This is typically not the case for comparing assemblages across different environments (eg marine environments have fewer species which are considered more diverse). Also the screening process assumed in the dependence model is somewhat deterministic in ignoring the likelihood of prior information about species. So despite the greater conditional probability of getting adjacent hits, there is no fixed tendency to screen close rather than at distance when other factors such as cost are not binding. In short there is a considerable gap between theory and practice. Specifically to the extent that there are any consistent criteria for setting areas aside for conservation purposes, they are likely to be a good deal less refined than the models highlighted to this point.

An apparent handicap from the preceding sections is that it is impossible to measure all character differences directly therefore the reliance on specific models for making any reasonable assessment of set character diversity and option value. Yet the taxonomic diversity measures are formulated for maximising diversity value within a small taxonomic group of organisms for which estimates of genealogy and geographical distribution exist. As mentioned, such exacting data requirements mean that heuristic surrogate approaches such as species richness are necessary.

2.7 Surrogacy in diversity measurement

Table 1 showed taxonomic character differences as lying within a well-defined hierarchy of more indirect measures of overall diversity and corresponding cost of use. As described, the use of various models to predict character distributions is already a recognition of that all characters of all organisms cannot be counted directly. From then on, according to Williams and Humphries (1996), the scale of indirect measurement is a matter of degree, with the appropriate question relating to how well any surrogate predicts whichever character is thought to be the ultimate repository of option value.

Williams and Humphries (1996) comment on an emerging consensus on the use of species richness (or its surrogate higher taxon richness), as a reasonable surrogate for character richness. The conditions under which this is unlikely to be the case are where 'taxonomic clumping' occurs (ie species rich areas that are character poor). Assuming data on a species exists in a geographic location, the problem is the maximisation of species richness in a set of reserves which will be the feasible set meeting some exiting budget. This problem is close to maximal coverage problems in Operations Research to which linear and integer programming methods can be applied. Such problems are flexible in coping with many of the realities of ground-level conservation, and an increasing number of applications are apparent in the conservation biology literature (see Csuti *et al* 1994, Pressey *et al* 1993).

The set selection problem

Most of the current discussion is naturally motivated by a belief that current *ad hoc* or opportunistic conservation efforts are inadequate or in some sense sub-optimal. The resulting financial and biological opportunity costs have given greater impetus to the assessment of the cost-effectiveness of conservation decisions, and the issue of biodiversity priorities (see Pressey and Tully (1994), Prendergast *et al* (1993), Metrick and Weitzman (1994).

Optimality may have as many definitions as the term biodiversity itself, yet there are several guiding criteria once consensus is reached on the appropriate surrogate. *Complementarity*, as previously described, implies a selection method taking account of what has already been saved in a network. *Flexibility* - implies that there may be more than a unique optimal set satisfying the objective function, with frequently occurring elements suggesting the *irreplacibility* of particular vital members that should be part of any representative network. With appropriate data, these criteria can be addressed in regular programming problems.

In the simple example presented above, Vane-Wright *et al* demonstrate the basic idea of complementarity in reserve section. This method employs a greedy (richness-based) algorithm starting with the site containing most species and sequentially selects further sites adding the most additional species.

Table 2. Site X species data matrix producing a counter example to the greedy algorithm. The algorithm chooses sites 1, 2 and 3, in this sequence, as priority sites, while it is obvious that only sites 2 and 3 required to preserve all species

	Species							
	1	2	3	4	5	6	7	8
Site 1	0	0	1	1	1	1	1	0
Site 2	1	1	1	1	0	0	0	0
Site 3	0	0	0	0	1	1	1	1
Site 4	1	1	1	0	0	0	0	0
Site 5	0	0	0	0	0	1	1	1

Underhill (1994) highlights a potential inefficiency with this procedure with reference to a similar hypothetical maximal coverage problem such as that in table 2. The table shows that no species is endemic to a single site and that no sites are automatically included in the reserve system. Basically the greedy algorithm will begin by selecting the most species-rich site, namely site 1. On the basis of complementarity, eliminating the already conserved species, site 2 is then the best of the rest followed by site 3 or 5. The greedy algorithm selects three sites for the reserve system. However, it is apparent that the coverage problem could be solved by only 2 sites (2 and 3). The basic problem is that the greedy algorithm will not allow site 1 to be subsequently dropped once it has been included in an earlier iteration.

Underhill (1994) disputes the wisdom of this interpretation of complementarity[16] when an efficient solution with full regard to previous selections can be attained using readily available linear programming methods. Given a matrix of sites times species the set coverage problem to minimize the number of sites can be redefined as follows:

[16]The interpretation is in fact that used in the Natural History Museum of London's heuristic selection algorithm, WORLDMAP.

$$Min \sum_{j \in J} x_j \qquad (1)$$

subject to

$$\sum_{j \in N_i} x_j \geq 1, \quad \forall\, i \in I \qquad (2)$$
$$x_j = (0,1) \quad \forall\, j \in J \qquad (3)$$

the index J = (j=1,..,n) are the candidate reserves available for selection while I = (i = 1,..,m) denotes the index set of the population of species to be covered. N_i is a subset of J representing the reserves which contain species i of the population. The variable x_j is a binary site variable for each site under consideration (=1 if site j is selected, =0 otherwise).
and x_i the species constraint from members of a set I = (i = 1,...,m) each member of the set to be represented at least once thus giving m constraints. The third constraint indicates that the variables must be binary and that there are n such variables. Appendix 1 applies this to a data set of large mammals in Kenyan parks.

This species coverage problem is dual to the primal problem of maximising the number of species represented where there is some limit on the number of reserves that may be chosen. Formally this is expressed:

$$Max \sum_{i \in I} y_i \qquad (4)$$

subject to

$$\sum_{j \in N_i} x_j \geq y_i \quad \forall i \in I \qquad (5)$$
$$\sum_{j \in J} x_j \leq k \qquad (6)$$
$$y_i = (0,1) \quad \forall i \in I \qquad (7)$$
$$x_j = (0,1) \quad \forall j \in J \qquad (8)$$

Sets I,J,N_i and variables x_j are as per the primal problem. This maximal coverage problem places a restriction on the number of sites that can be selected by using constraint (6) to limited the number of selected sites to k. This implies that not all species may be protected, so that the constraint used in (2) has to be modified. Rather than the requirement that each species be conserved on at least one site, the variable y_i is now binary (1,0) to indicate whether or not the species is present in a selected site. In the objective function (4) y_i will be one except where $x_j = 0$ for all j in N. In other words, constraint 5 will ensures that a species will not be counted as preserved if none of its sites are selected.

In solving the basis problem the criteria of flexibility and irreplacibility can be investigated by iterative solutions worked under alternative appropriate constraints. Several such alternatives will be mentioned below.

The more usual restriction than limiting the number of parks in the maximal coverage problem, is that the procedure will have to satisfy a budget constraint. In the maximal coverage problem this might be related to the purchase of land or simply some form of easement or covenant (see Buist *et al* 1995). The appropriate modification will be to replace the parks constraint K with:

$$\sum_{j \in J} c_j x_j \quad \leq B$$

where c_j represents the cost of site j and B the total budget.

The set coverage problem restricted every species to a single representation according to constraint (2). If multiple representation is preferred then it may be feasible to set the constraint to conserve the species at $p > 1$ sites. This might be a prudent course of action if one site is considered to be particularly threatened, or if specific information from population biology dictates some minimum distribution. For a feasible solution the species has to occur at p or more sites. From the species site matrix, define an occurrence variable to sum the number of sites in which the particular species occurs. The appropriate constraint in (2) would the be greater than or equal to the minimum of p and this occurrence total. In the maximal coverage problem, a similar restriction can also reflect the fact that some sites may already be protected and that the question is simply how to supplement this existing system in an optimal way. To do this, any of the appropriate x_j in (3) are set to 1, and the problem solved accordingly. In keeping with the realities of conservation it is possible to speculate about numerous other potentially binding constraints on problems (eg contiguity of sites, the availability for protection etc).

The greedy algorithm proposed by Vane-Wright *et al* (1991) and the branch and bound algorithm commonly used for integer programming are only two possible algorithms. Alternatives include various forms of rarity based algorithms (Csuti *et al* (1994) which scores a species by the inverse number of sites in which it occurs. Using data on the spatial distribution of species in Oregon, Csuti *et al* (1996) assess the relative performance of 19 algorithms in terms of criteria such as the number of areas selected, iterative procedure and tie-breaking, time required etc. The finding is that although the branch and bound procedure appears most efficient in both number of sites accumulated for species represented rarity, heuristic algorithms do not do badly.

Perhaps the best advantage of LP methods is that they are well known in other fields and generally available (Williams 1994a). For computationally intensive large data sets solution times may be a consideration using regular Integer programming algorithms (see Pressey *et al* 1995). Problems are 'NP-complete,' which means their difficulty increases roughly exponentially with the number of constraints. The simplest way to make sure a problem converges in reasonable time is to simplify the problem. For example, if reserve site 2 contains all the species of reserve site 1, it can be eliminated from consideration by adding a restriction to the budget constraint to instruct it that 2 costs more than 1 thereby reducing the number of x_j. Another simplification in the maximal coverage problem above is to relax the integer (1,0) constraint on y_i, since optimization forces them to be 0 or 1 given binary x_j. Doing this reduces the number of binary variables from $m+n$ to n.

The question of relative utility of alternative selection algorithms depends not only on the mathematics, but also on the practicalities of conservation planning. Therefore, as suggested by Pressey *et al* (1994) a preferable procedure might involve some interactive method between a stepwise and fully automated algorithmic solution (eg updating new information about sites that have already been selected). This may also be the best way of dealing with intertemporal vulnerability which is difficult to incorporate in the problem. In conclusion, it seems best to regard set methods are complementary to other various other tools employed in gap-analysis (Ingram and Williams 1993) such as Geographical Information Systems.

2.8 Conclusion

This chapter has dealt with diversity, and examined how it may be valued, and associated methods to integrate something akin to option value into decisions on conservation versus development. The topic is introduced prior to the use of valuation methods to demonstrate the extent of any interface between economic valuation and a framework which seems amenable to the provision of an unambiguous diversity measure. In short this interface is still limited but there are good reasons for further pursuing this line of enquiry.

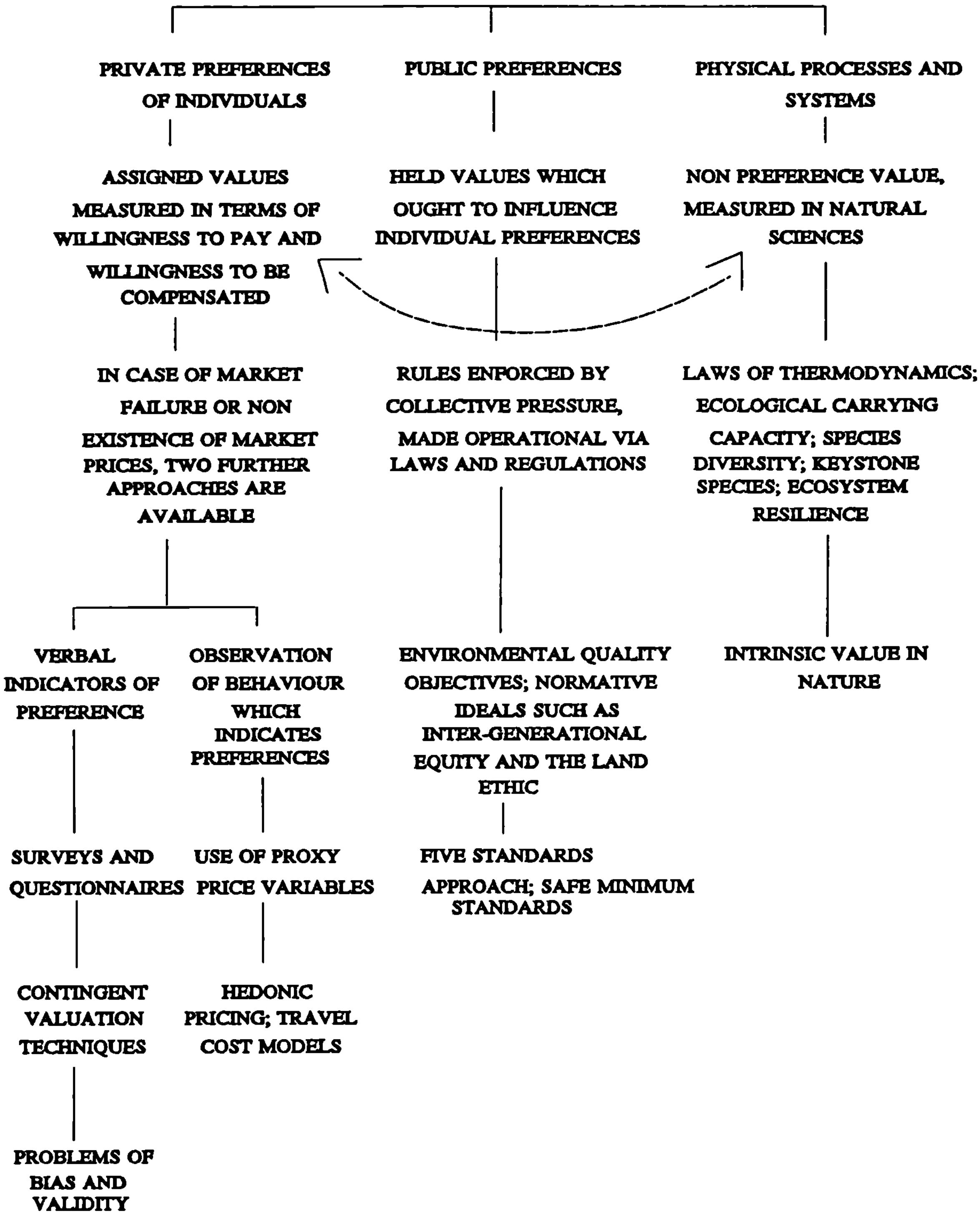
First, in the near hysteria which has characterised the extinction debate, failure to address some fundamental questions about the unit of value may mean that conservation activity is economically sub-optimal. Second, since economists are frequently admonished for being value obsessed, it is important to establish links between wider concepts of the 'sustainability' debate and the maintained economic paradigm. This is possible using distance theory which as shown in figure 7, establishes a bridge between economic and scientific values.

Third, from a practical perspective, disciplines such as molecular biology, taxonomy, conservation biology and ecology, often cannot interface with economic methods. Yet, the same disciplines are steadily generating a wealth of almost limitless quantitative information as species are identified and classified, and it seems inconceivable that none of this information is amenable to economic analysis.

Looking below charismatic species, there is uncertainty about the motives underlying the preference for more diversity, and much of the discussion about the biological location of option value is somewhat prior to the issue of how to assign monetary values. There is scope for further research to establish an optimal approach to dealing with the undoubted benefits accruing from biological diversity. Although much more information is ideally required for identification etc, the taxonomic basis of analysis seems well established and, in theory, amenable to economic analysis. The different approaches to measuring diversity are useful because they highlight different objectives or rely on different levels of information. As seems clear from this chapter, some measures are information-intensive.

Some immediate potential lines of enquiry include: a) untangling the motives behind preferences for diversity; b) the integration of exact and pattern distance models into economic valuation, and c) development of operational research programming methods for setting priorities at various levels,

Figure 7 Environmental values



Source: Adapted from Pearce and Turner (1990) including opportunity cost analysis.

The main conclusion is that from an economic perspective, diversity as a feature of commodities including biological entities, is a relatively unexplored concept. This seems to be due to difficulties involved in reaching consensus on which level to locate value[17], particularly option value. Although a powerful emotive argument, option value remains an appeal in the face of ignorance. In practice far more effort is required to integrate the concept into conservation decisions. On the other hand the choice of what to preserve cannot be decided by measuring diversity alone. There are many other elements important to such decisions about which we are equally ill-informed. In fact, it turns out we do not even have consensus on how many species may exist, nor even for those that are known, which can and cannot be regarded as species in their own right (Rojas 1992).

If diversity is a normal good, we prefer more and our welfare is undoubtedly increased indirectly by knowing that a wider portfolio exists, or directly by the option of one day having the use of an as yet unknown a biological resource. But there is no clear satiation point in such reasoning and while diversity conservation implies high opportunity costs, it seems reasonable to investigate any potential to set reasonable bounds on option value. This chapter has set out to see the extent to which a true measure of diversity is theoretically commensurate with costs and benefits or, more precisely, what difficulties are specific to conservation policy. Taxonomic measures can be based on a wealth of biological data and provide a framework in which the implications of competing definitions of option value can be assessed. The issue of the appropriate level of option value goes to the heart of the source of value in biodiversity. How is the appropriate unit measured and maximised and how consistent are the resulting priorities with other values such as existence or (albeit probabilistic sense) use values, which are always the most tangible bases for assessing the returns to conservation.

There are other complications in using phylogenetic pattern models related to the basic assumptions for constructing phylogenetic trees and the accuracy of phylogenetic models themselves for predicting character representation. This chapter does not suggest these models as a definitive approach. Indeed, in practical and scientifically correct terms, the appropriate strategy may be to locate value at the species level (abstracting from however these are defined), and using surrogates species. This approach has a lot of appeal for biological and economic assessment. The most compelling reason is the data limitations. An acceptance of the need for surrogates effectively makes the economic

[17] This problem is neatly summed up in Plato's belief that species are only sharply defined in the mind of God.

assessment much clearer. In the first place, area selection problems are more tractable. Furthermore, there is a clearer approach to addressing both use and non use value apart from option value. For example, the use of indicator species permits assessment by contingent valuation methods and other stated preference methods. Also it become far easier to implement alternative decision approaches not involving precision in defining difference. An example is the use of Safe Minimum Standards (Hohl and Tisdell 1993), possibly motivated by the need to take a precautionary approach to development activities likely to conflict with conservation decisions. Safe minimum standards have implicitly motivated many conservation programmes (eg the IUCN 1994) can be motivated by population biology. However, safe minimum standards are not totally immune to many of the uncertainties and inefficiencies of *ad hoc* methods. For example, it is basically impossible to make a species safe with a probability of one. In recent times, the precautionary approach has become quite fashionable in policy questions characterised by uncertainty irreversibility (Myers 1993b). In the specific argument about weighting characters in this chapter, precaution can be evoked to argue both in favour and against weighting. On a global scale, precaution can often be viewed as a peculiarly rich country approach to uncertainty. With one eye on the political ramifications, the need to lock up tracts of virgin rain forest may be much less appealing avenue for developing countries.

Where does all this leave the economic assessment of biodiversity? Commenting on the inevitability of cardinality in environmental decision-making Brown notes that few controversial decisions made on quantitative grounds in the past are routinely made with qualitative arguments today, Brown (1996 pp19). In other words, an increased demand for quantification in biodiversity decisions seems inevitable. The challenges in this area are intriguing and one can envisage two sets of developments to this end. First the inevitable systematics agenda to establish an inventory of the earth's biodiversity. This project is a global initiative and is under way (see Claridge 1995). Very much like the human genome project, it will surely provide the basic data to expand the Weitzman-type distance theory analysis. Second, is the refinement of economic valuation methods, or more precisely the provision of information. Essentially this chapter has attempted to clarify views of what we are actually valuing, which for the time being is somewhat cruder than diversity itself. An important point is that the two potential strands of analysis are not totally unrelated.

Given the immediacy of biodiversity loss, the most feasible option would be to be to concentrate on strengthening the economic case based on surrogate species and ecosystem diversity. This may guarantee (albeit inadvertently), the maintenance of diversity. Concentrating on surrogates such as species richness also leaves the field very much open to the valuation methods to assess other aspects of value. In the light of this finding, the following three chapters therefore evaluate the role of the contingent valuation method for informing conservation decisions

Appendix 1

The Set Coverage Problem.

The set coverage in this example draws on the census data of large mammals in sampled zones belonging to Kenyan parks and reserves Table A1, GOK (1995). Supposing the sample to be representative of the population of each park, the basic problem as presented in the chapter, is the minimisation of area while covering each species at least once. Many alternative software packages are available to deal with this type of relatively simple problem and the example used the linear and integer programming options in the Solver routine of Microsoft Excel 5.

Although not immediately obvious the optimal solution is actually only two parks (MMA and TRA). By careful inspection of implied coverage, this can be deduced without the algorithm which is slowed down by the number of integer constraints.

As previously discussed, there is clearly a lot of potential to analyze competing 'accumulation curves' under alternative assumptions. For example, the conservation of the charismatic big five for tourist purposes. Ideally, any cost information would facilitate the cost-effectiveness of alternative conservation objectives.

Table A1
BIODIVERSITY ANALYSIS IN
PROTECTED AREAS OF KENYA
KREMU DATA - PROTECTED AREA & 20KM
BUFFER ZONE AROUND

MAIN DATA MATRIX FROM KREMU REPORT

NAME	BOG	BND	KMK	LOS	MRL	MRS	NAK	STN	SBF	SHL	SIB	TRA	TSV	AMB	MMA
%SAMPLE	5.9	5.78	5.87	5.53	5.63	5.53	10.13	5.25	5.63	5.55	5.53	5.56	5.87	5.72	10.13
SMPLD AREA KM2	134.9	276	677.3	381.6	76.2	363.6	92.1	321.4	207	133.6	261.2	348.3	503	161	517.7
ZONAL AREA KM2	2286	4775	11538	6900	1353	6575	909	6121	3676	2407	4723	6264	8568	2814	5110
EL	0	0	16	40	10	44	0	2	57	15	0	0	815	20	135
GF	0	0	115	78	0	27	8	0	13	0	0	98	252	94	168
ZB	17	0	2	0	194	0	45	0	17	0	0	91	1140	921	613
ZG	0	0	16	2	2	15	0	0	56	0	36	3	0	0	0
TG	0	0	0	0	42	0	207	0	4	0	0	0	0	14	4256
GG	0	0	42	136	5	121	117	23	62	0	86	70	508	425	391
KG	0	0	0	0	0	0	0	0	0	0	0	0	648	0	207
IM	0	0	11	0	36	0	203	0	11	0	0	1	434	36	4454
WL	0	0	0	0	0	0	0	0	0	0	0	0	58	472	616
TP	0	426	0	0	0	0	0	0	0	0	321	87	0	0	995
HH	0	0	0	0	0	0	0	0	0	0	0	48	0	0	0
BF	0	18	0	0	0	34	30	0	22	0	0	133	746	72	317
ED	0	0	0	0	17	0	14	0	0	0	0	16	556	233	75
OS	0	5	36	19	1	32	0	0	38	0	18	23	87	64	15
OX	0	0	19	48	2	21	0	2	16	0	105	42	199	11	0
LK	0	0	8	1	0	1	0	7	0	0	4	19	142	7	0
GN	0	0	45	24	5	12	0	3	13	0	49	18	78	2	0
WB	0	24	9	0	0	0	183	3	0	0	0	43	23	3	51

Chapter 3

Contingent valuation and biodiversity: Theory and methodological issues

3.1 Introduction

The previous chapter provided an appreciation of the biological resources - diversity distinction and the difficulties encountered in trying to factor diversity into cost-benefit methods. Chapter 1 advanced CV as a flexible valuation method for the measurement of total value and one can speculate that CV responses include many different motives including a preference for intrinsic value. CV offers a practical, albeit imprecise, approach to conservation. This chapter resumes the discussion of CV with a critical review of some theoretical and methodological issues raised in applying the method. The chapter is divided into three sections. The principal focus will be on the methodological approaches for the derivation of an exact welfare measure from survey data - an area of growing debate in the valuation literature. In particular, considerable attention will be devoted to the use of, and problems associated with, surplus elicitation using the discrete choice (DC[1]) question format, which has recently dominated the contingent valuation literature. The method is generally thought to be more incentive compatible than the open-ended alternative which requires respondents to state freely their willingness to pay for or against an environmental change. Yet, having been introduced from biological assay and other established areas of economics, the specific empirical problems involved in analysing yes/no responses in the CV context suggest this advantage may be more than offset by the statistical inaccuracies resulting from the use of inappropriate models.

Irrespective of the proffered format, the implicit assumption in CV practice is that preferences exist over changes to the subject good, and that these -or their verbal equivalents - can be characterised by the standard apparatus of demand theory to establish some orders of magnitude for the resulting change in components of total value. In this regard, a great deal of scrutiny has been focused on the characterisation of existence value and its place in benefit-cost analysis or in damage litigation. Recall from chapter 1 that the measurement of existence values was presented as the main advantage of the method. This chapter makes no further attempt to specify the precise role of CV beyond the claim to elicit a total value which may include an existence value component which may or may not vary with according to the spatial proximity of the respondent and the subject good. From a resource allocation point of view, this total value is still advantageous even if the distinction between user and non user values cannot be easily made. Yet side-stepping the deconstruction of total value does not facilitate the task of conducting CV to gain unbiased estimates of welfare change. The consistency

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[1]Otherwise known as a dichotomous choice, referendum or 'take it or leave it' format.

of stated - albeit hypothetical - preferences with demand theory is of considerable interest for assessing the advantages of the most flexible of non-market valuation methods. However, it is the hypothetical nature of CV which complicates the undertaking, opening up many categories of goods which are the objects of values which may potentially drive a wedge between the fictional self-interested Hicksian consumer and observed hypothetical choice[2]. The motives for observed choices are therefore worthy of attention, and by extension some consideration of which goods causing the greatest deviations and apparently extreme responses.

As a precursor to dealing with the responses, and in order to motivate the utility theoretic foundation of the CV responses using the DC format, the first section presents an account of the established theoretical underpinnings of the method. In so doing, the section provides clues as to the potential problems involved in valuing complex goods such as biological resources, and some ways to modify the CV approach to avoid the occurrence of, or account for 'unreasonable' responses.

Notwithstanding such 'fixes' as are possible within current CV practice, and in the light of the values known to be held with regard to biodiversity, the final section attempts to illuminate the causes of persistent forms of extreme response observed in applied work which cast doubt on the ability of the CV to contribute anything meaningful to the allocative process. The essential question is whether preferences can reasonably be expected to exist over complex goods, or what it takes to lead respondents to a position of *informed* stable preferences that can be accommodated by CV. If it turns out that some of these 'informed' preference structures invalidate the CV method, can this be accommodated and explained by economic theory and more importantly, are there alternative approaches to setting limits to the costs and benefits of biodiversity conservation? By stretching the interpretation of demand theory, extreme responses can in some circumstances be shown to be theoretically valid, although this conclusion does not always facilitate policy decisions. There is much at stake in the great CV debate, and the observation of theoretical "blind spots" is somewhat damaging. Instead of articulating the apparent limits of CV or theory, many proponents have questioned the findings of studies, maintaining that extreme anomalies are usually artefacts of the survey design. Regardless of whether or not the method is universally adopted, the debate is set to continue. The long-term result may well be an improved theory of consumer choice. From an immediate policy perspective some short term consensus would be welcome.

[2] Sugden (1996) points out that inconsistencies are by no means the preserve of a hypothetical/ real dichotomy in that real consumers do not exactly fit the Hicksian template either.

This would be a convenient point to pass some judgement on CV were it not that some of the apparent fundamental elements of the debate are so polarised. In the first instance, and related to the measurement of existence value, it is claimed that the absence of any objective method of measurement means that it is essentially impossible to refute any claim about their magnitude. In the absence of any validatory alternative, the existence value results of a CV are essentially a leap of faith. Given this absence of independent data needed to validate predictions of existence value, one might concur with the view of Blamey and Common (1993) regarding the commonly adopted methodological stance of Friedman (1953)[3], as being untenable in the case of CV. This charge is however dependent on whether some alternative validatory model (e.g of charitable giving) should necessary accurately reflect existence motives.

A further caveat arises in discussion of the appropriate model for comparing CV. The assumption of the market model is clearly too rigorous for Mitchell and Carson (1989) who note that CV of pure public goods may be more appropriately judged as improved referenda (p296). Similarly Carson (1995a), notes that it is somewhat disingenuous to subject stated preferences to rigour that has not been applied to other data such as household expenditure surveys. Taken together, such statements cause considerable confusion over what CV actually does. On one hand the CV seems to have been mistakenly sold as an existence value elicitation device. Yet the appropriate yardstick for assessing responses remains uncertain and the links with theory suspect. In such circumstances one might legitimately question the understanding underlying multi-billion dollar damage liability cases. For example, is a ruling in favour of acknowledging existence values as compensable entities made on the basis of an understanding of the consistency with theoretical prediction or something less rigourous?

A final point to make is that any alternative theoretical model emerging as a result of the CV debate is unlikely to be the sole preserve of economics. Several plausible models of choice are particularly illuminating when economics appears deadlocked. These issues will be alluded to where necessary, but this chapter attempts to ring-fence discussion to several specific topics. It is hoped that this discussion will provide an ideal background for the consideration of some of these issues in two CV exercises presented in subsequent chapters.

As a conclusion it will be seen that in both the statistical analysis and the choice of subject goods, a considerable degree of judgement is necessary on the part of the researcher. As was evident from the

[3] Which asserts that direct testing of assumptions is unnecessary so long as predictions based upon them are confirmed.

battery of evidence for and against the method wielded during the Exxon Valdez debate[4], it is precisely this subjectivity in approach which may be the achilles heel in the promotion of a water-tight and consistent approach in economic assessment.

3.2 A review of welfare measurement.

A review of basic welfare measurement is motivated by the need to give CV responses some context grounded in economic theory. Specifically, it is important to observe elicited responses as consistent with demand theory as this provides a basis for assessing the theoretical validity of the method and a justification for using CV in resource allocation. Economic theory also provides a reference point for understanding the problems that emerge in actual CV exercises. These include the disparity between willingness to pay (WTP) and willingness to accept (WTA) compensation, the difficulties that may be encountered using the theoretically correct elicitation method to value some categories of good, and why CV may simply not be an appropriate method for valuing some goods.

As will become clear, the use of duality theory from applied demand analysis offers a convenient framework for viewing welfare measures, and for considering the random utility structure underlying dichotomous responses (DC). A drawback of the a neoclassical approach and one worth recalling at the outset, is that it is largely deterministic in its treatment of the unknown. In particular, hypothetical behaviour has its own theoretical constructs which may or may not be consistent with economic rationality[5]. The contribution of cognitive psychology in this area is noteworthy (Schkade and Payne 1994 Fischoff 1994 Hutchinson *et al* 1995). The purpose however is not an extensive comparison of views on cognition and response motives beyond the TEV framework. Where necessary the import of appropriate fields to explain anomalous findings is alluded to.

The economic value of an environmental good equals the utility change caused by a quantity change of that good (actual or its hypothetical representation). Consideration of actual or hypothetical quantity or quality changes follows similar lines to the more commonly considered effects of price change (see Freeman 1994). Given some strict theoretical assumptions utility changes can be translated into monetary equivalents and evaluated as discrete changes in consumers surplus. These

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[4] An excellent introduction to CV can be found in a special issue of Choices (second quarter 1993). A balanced view of recent developments may be found in papers by Portney (1994), Hanemann (1994a), and Diamond and Hausman (1994). Hausman (1993) presents a contrary view.

[5] In the context of CV Schkade and Payne (1994), for example, reflect on the possibility of a plurality of respondent response models. Given these individual differences one can question the validity of aggregated responses.

assumptions basically pertain to the existence of "well-behaved" utility mapping[6] of indifference curves for the good or quality change of interest. Abstracting from the actual content of individuals' preferences[7] these are said to provide sufficient information for consistent ordinal ranking of states of the world.

Assuming these conditions hold, it is possible to specify/maximise forms of the utility function subject to a budget constraint and derive Marshallian demand curves of the form:

$$x_i = x_i\,(P,M)$$

where P is the vector of prices and M is income.

In specific circumstances these provide an observable clue for deriving underlying preferences and therefore utility functions from market data on individuals' responses to changes in prices and income. These conditions are provided by integrability theory which requires the Slutsky matrix of substitution terms be symmetric and negative semi-definite (Deaton and Muellbauer 1988). If this is the case, it is possible to specify a system of differential equations which can be integrated to recover the expenditure function and thereby derive surplus measures from the difference in two expenditure functions (see below). The process of integration is not however straightforward.

Marshallian consumer surplus CS is bounded by the marshallian demand curve, the observable analogue of utility which is commonly not observed (see below discussion). The problem with welfare measurements based on a marshallian approximation, derives from the necessary assumption of constant marginal utility of income as one moves along the curve representing the bundled income and substitution effects. This assumption is likely to be violated in the real world, leading to an inexact measure of utility due to the income effect of the price change. To obtain an accurate measure of the value to consumers of the change in resource allocation due to the relative price change, it is necessary to compensate for income effects. Integrating along constant utility indifference curves,

[6] The axioms of consumer choice are completeness, transitivity, convexity and non satiation (Deaton and Muellbauer 1980). Carson (1995a p6) reviews the literature on the relaxation of these axioms.

[7] There has been an on-going debate about the definition of allowable motives for economic satisfaction. Kahneman and Knetsch (1992) have dismissed CV responses as the purchase of moral satisfaction, which they say, is not an economic motive. Samuelson (1993) cited in Carson (1995b) rejects the notion that some motives are not legitimate determinants of economic behaviour.

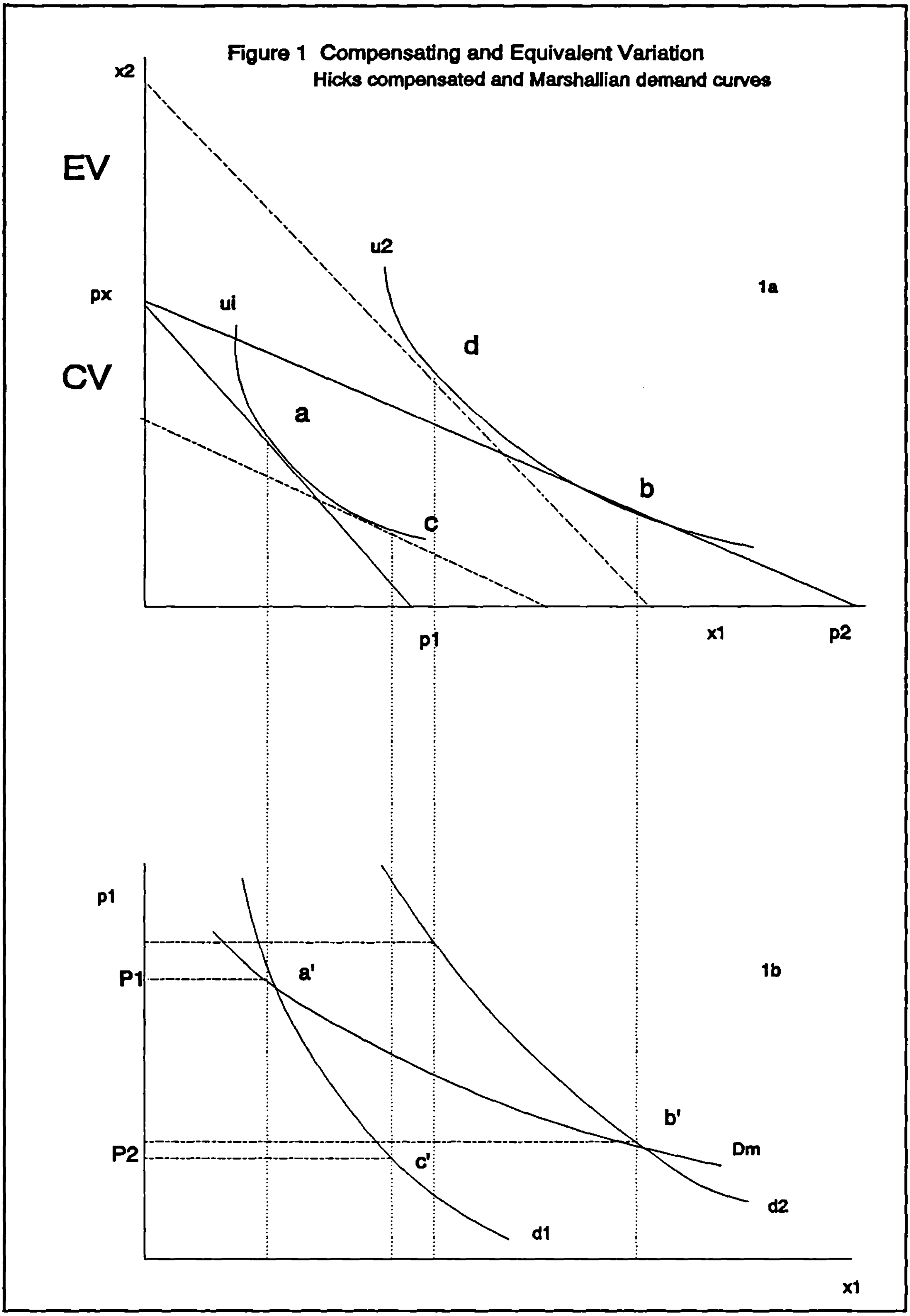


Figure 1 Compensating and Equivalent Variation
Hicks compensated and Marshallian demand curves
x2
EV
px
CV
u2
ui
1a
d
a
b
c
p1
x1
p2
p1
P1
a'
1b
b'
P2
c'
Dm
d2
d1
x1

Hicksian compensated welfare measures bound welfare change resulting only from the substitution effect of relative price changes.

There are basically five measures of consumer surplus, three of which are shown in figure 1 for a fall in price p1 - p2. Given an initial point 'a' on the indifference curve u1 between good x and the composite good numeraire x2, a price fall scenario of figure 1a, shows two forms of compensated demand curves which may be derived according to the reference point for regarding the change.

The compensating variation CV asks what change in income may be subtracted from the individual after the price change and leave him indifferent between his initial situation and the new price set. The CV takes the initial level of utility as the point of reference and can also be interpreted as the maximum amount an individual would be willing to pay (WTP) for the opportunity to consume at the new price set. Conversely, for a price increase the CV measure is the willingness to accept to be indifferent to the price change. The Equivalent Variation (EV) of a price fall measures the monetary equivalent of the utility gain of the price fall and in this context can be viewed as the WTA for foregoing a price decrease (which they had a right to). In the case of a price increase, the EV is a willingness to pay measure to consume at the original price set and utility.

Focusing on the price decrease, figure 1b shows the Hicksian compensated demand curves for the two welfare measures and the intermediate Marshallian curve. The CV for example is approximated by the area to the left of the Hicks-compensated demand curve d1 between the relevant price lines, that is area (p_1a'c'p_2).

For a normal good with an income price elasticity greater than zero, it can be seen that both compensated demand curves are less price elastic than the ordinary demand curve.
Evaluated thus, for a price fall, EV>CS>CV. In other words the WTA>CS>WTP, since WTA is evaluated at the higher reference utility level to which the respondent has an assumed right.

Theoretically, the exact welfare measures of the change can be viewed in terms of utility maximisation or the dual expenditure minimisation decisions. In terms of the indirect utility function[8] the CV is the solution to:

[8] Derived by substituting the ordinary demand equations from the primal constrained utility maximising problem into the original utility function. The dual expenditure minimisation problem which reveals Hicksian demand functions which substituted into the expression for total expenditure give the expenditure function, (Deaton and Muellbauer 1980).

$$v(p1, M) = v(p2, M - CV) = u1$$

In other words the WTP to consume at the lower price level.
On the other hand, the EV can be shown to be the WTA to forgo the price fall:

$$v(p1, M + EV) = v(p2, M) = u2$$

In terms of the expenditure function:

$$CV = e(p2, px, u2) - e(p2, px, u1)$$

$$EV = e(p1, px, u2) - e(p2, px, u2)$$

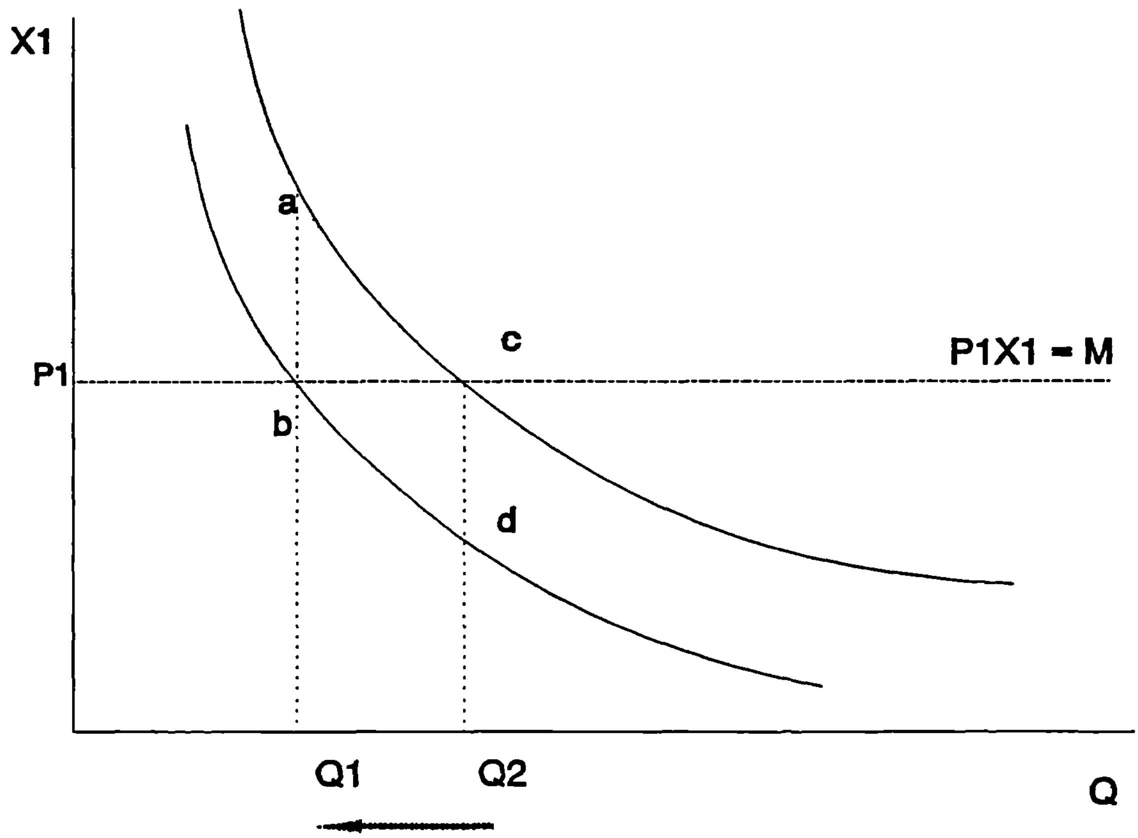
From figure 1b it can be seen that Marshallian surplus lies between the EV and CV, and may seem a suitable approximation of either to be used as an index of utility change for use in compensation tests. Such an approximation would be particularly advantageous from the point of view of empirical work and it is of some interest to know the conditions when an approximation is appropriate. In fact, the measures can only be the same if the marginal utility of income is constant and equal to 1. In other words, all indifference curves are assumed to be vertically parallel.

(Willig 1976) has shown that the difference between the three measures depends on the income elasticity of demand for the good in question and consumer surplus as a percentage of income, and that within certain bounds an approximation of exact welfare measures may be appropriate particularly for small price changes. Other authors have cautioned against reliance on these bounds however, since they are developed for changes in Marshallian surplus resulting from the change in price of market goods. In the case of environmental goods the approximation error may not be straightforward (Bockstael and McConnell 1980). Furthermore, other methods for recovering or at least approximating exact measures do exist (see Bergstrom 1990 or Freeman 1994 for a review). If the integrability conditions are not met by the demand curve used in Willig's approximation, then this is not generated by utility maximisation and other methods may be more appropriate.

The analysis in terms of prices assumes the individual is free to adjust quantities consumed in response to changes in relative prices and income. In the context of CVM of environmental goods,

Figure 2

Compensating and Equivalent Surplus measures, quantity constrained case



Compensating and Equivalent surplus measures.

For a quantity or quality reduction CS = a-b and ES = c-d a more plausible assumption is of individual consumption being restricted to an invariable quantity bundle where optimizing adjustments in consumption are not permitted. Compensating and Equivalent Surplus measure welfare changes attributable to restricted quantity changes. Such a case is shown in figure 2, which shows utility curves for the consumption of environmental good Q, and some market good X which can be regarded as a composite market commodity. Qo represents the initial quantity of Q and the horizontal budget line px=M shows how income is exhausted on consumption of xi given p1, and assuming that the environmental good is unpriced[9]. To approximate welfare scenarios of CVM studies in subsequent chapters, assume the case of a quantity or quality decline from the initial situation Q2 to Q1. We can show the measures of compensating and equivalent surplus as follows. Assuming the initial environmental quantity, the ES c-d measures the income equivalent of a quality decline from Q2 to Q1. In other words, a move onto an inferior utility curve U1. The ES in this case an be interpreted as a willingness to pay measure and it is clear that this might be the appropriate question in a CV scenario which asks respondents to consider the maintenance of an amenity in its current condition. By contrast, the compensating surplus a-b assumes the quantity fall has occurred or is inevitable, and is the amount of money required to make respondents indifferent to the fall by returning them to initial utility at the inferior quantity. Note that as drawn, the magnitudes of WTP and WTA depend on the convexity of the indifference map. In other words for different assumptions regarding the marginal rate of substitution, differences between them may be more pronounced and in some circumstances extreme for a case of lexicographic preferences implying a higher or infinite WTA (see below).

3.3 The choice of a welfare measure in CVM

For many environmental changes, there is a lack of consensus over the reference utility level, i.e. in terms of the quantity reduction in figure 2, does the respondent have a right to be on U2 or does he have to pay to be there. As Hoevanagel (1994) points out, CVM practitioners are in the fortunate position of being able to choose between welfare measures. In doing so several theoretical and methodological considerations provide guidance : i) The disparity between WTP and WTA; ii) The implied property rights; iii) The uniqueness of the measure used; iv) The consistency of the measure used in terms of being a reliable index of utility. These are key issues in survey design and are briefly be discussed in turn. However, it is worthwhile noting that it is at this stage of translating the theoretical construct into an elicitation device that CV is inevitably invaded by a raft of cognitive issues which, though not beyond the domain of neoclassical economics.can only be given cursory attention.

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[9] Freeman (1993) considers the interesting and relevant case if welfare change Q is priced.

The WTP WTA disparity

As drawn in figure for the normal good case WTA > WTP simply because the former is measured using the compensated demand curve at the (post price fall) higher level of utility. It has been shown that a Marshallian approximation of either of the Hicksian measures may lead to some margin of error which will be accentuated for particular configurations of the income effects and magnitude of consumer surplus. This leads to a necessary choice between WTP and WTA as the appropriate surplus measure

Despite a theoretical extension analogous to the Willig results to quantity changes in unpriced environmental goods, which appear to place both formats on an equal theoretical footing (Randall and Stoll 1980), the continued observation of a divergence in hypothetical and real studies (see Kahneman *et al* 1990 for a review of studies), places CV researchers in a dilemma over the need to match format to the implied property right. Commenting on the persistent empirical evidence however, Mitchell and Carson (1989) emphasise the combination of factors contributing to the disparity. Of most importance are: i) the rejection of the willingness to pay format (see below); ii) the cautious consumer hypothesis (Hoehn and Randall 1987); iii) loss aversion, or the endowment effect formalised in prospect theory and reference-dependent preferences (Kahneman and Tversky 1979); iv) moral responsibility in purchase and sale decisions over certain goods (Boyce *et al* 1993; Peterson *et al* 1995); v) reinterpretations of economic theory to cover income and substitution effects (Hanemann 1991).

In the context of endangered species all of these factors may be significant. Since the last of these remains closest to the standard neoclassical framework it is reviewed first. Issues iii and iv are reviewed in a later section. Hanemann (1991) showed that the Willig-type bounds for the quantity constrained case derived by Randall and Stoll are in fact a function of the ratio of two parameters, namely the income elasticity of the environmental good and the elasticity of substitution between it and all other goods. If this is true, then it turns out to be easy to speculate about values of either, which validate a WTP/WTA disparity for private or public goods[10]. Intuitively it might be expected that irreplaceables such as precious species or human life would be associated with very low elasticities of substitution for any given ratio of willingness to pay to income. On the other hand, for a common private good about which nobody should feel too emotive, the converse would be true.

[10]In the latter case speculation is difficult as elasticity estimates for environmental goods are difficult to come by. Crude estimates of income elasticity from CVM data have been attempted by Kristrom and Riera (1996). Flores and Carson (1995) reinterpret the income elasticity of WTP as a virtual price elasticity and show its relationship with the income elasticity of demand.

That is, a moderate or high elasticity of substitution and a low income elasticity.

Going beyond the property right issue, it is difficult to rationalise Hanemann's results, although they do provide an elegant salvation for theory. In essence, the analysis proves that the format of question does matter from a theoretical point of view, and gives some foundation to dozens of empirical studies that had emphasised the disparity for the types of good Hanemann referred to. The reinterpreted role of income and substitution effects provides an insight into where CV researchers might if possible, need to reinterpret the property right to avoid the WTA format (see Mitchell and Carson 1989 pp 38). At the limit reinterpretation circumvents the problem of excessive bids simply because WTP should be bounded by the respondents' income, whereas willingness to accept could be an infinite amount. The only problem with the above is that the WTP/WTA disparity appears to extend to goods not typically characterised by the income and substitution effects described by Hanemann eg. mugs and pens (Kahneman *et al* 1991). While these conditions provide one highly plausible theoretical basis for the disparity, it seems reasonable to seek further possible explanations. It turns out that in the species context some of the motives enumerated above (such as iii), may explain extreme responses.

The implied property right

There is a legitimate sense in which the owner of a resource (or an individual who perceives whole or partial ownership rights), should be asked his willingness to sell (WTA compensation (an ES for foregoing a quality increase to which the respondent has a right or CS for accepting a deterioration) for surrendering all or a part thereof.

A property right allocation is inherent in the selected question format but protest responses[11], outlying or identifiable protest bids demonstrate the difficulty in adhering to the theoretical welfare measure consistent with that right. Coursey *et al* (1987) have shown that it is possible to reduce the disparity in repeated trials, but typically the question becomes one of deciding whether a more reliable WTP question can be used as a surrogate for WTA without provoking protests.

The value generated by a particular format will depend on the acceptability of the property right. In order for the respondent to fathom the hypothetical property right created by CV they must imagine: 1) that the transfer of the property right could feasibly occur; 2) how the transfer of property rights effects their individual utility; and 3) how that utility translates into some unit of currency. The

[11]In the form of uncorroborated zero bids, exaggerated sums or high rates of item non-response.

outcome of the last of these decisions is embedded in the response the investigator will actually observe, although all three will be the outcome of cognitive processes which are difficult to predict for specific goods and respondents.

Although many environmental resources may be collectively held, respondents do not always recognize their property right. If they do, they do not regard it as inalienable in the face of environmental change or the need to make trade-offs for individual or group betterment. This is a somewhat sweeping hypothesis and it is clearly possible to think of contrary cases where a property right is doggedly defended[12]. If this is generally the case however, one might suggest that there are in fact very few public goods for which a willingness to pay variant (ES to prevent a quality decline in figure 2) might be unsuitable to replace a WTA (CS for the change taking place)[13]. Mitchell and Carson (1989) imply as much in distinguishing between the strength of the property right over private and public goods when the differentiating factors are whether a good is used or not, and, in the case of a public good, whether it is privately or collectively held. In the case of a public good such as air quality however, the situation with respect to the property right is somewhat different. An individual consumer of the good does not have a generally acknowledged right to transfer it, even in return for a cash payment. Moreover if there is a cost to providing the good, it is borne by all consumers through a combination of higher prices, taxes and fees. The appropriate analogy is not a payment for an ordinary private good, but payment of a maintenance fee in a communal housing scheme for maintenance of surrounding amenities. More amenity can be had for the payment of a larger fee and vice versa. The important point is that a reduction in the quantity of the collective good is still accompanied by a payment from those enjoying the good and not to them.

In practice the perspective of any individual respondent cannot be compartmentalised so conveniently for the purpose of deciding on question format. Nor is it straightforward to delineate cases of goods where redefinition of the property right would be without problems. Other issues such as the prevailing institutional and cultural context and scientific awareness may distort perceptions of ownership of public goods[14]. For the purposes of conducting CV, this suggest that consistency with theory is less important than judgement on the part of the survey designer. On the whole, there

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[12]Protests against road construction in many countries would seem to be the clearest recent example of individuals recognising and claiming their rights.

[13] Species survival might be an exception to both.

[14] For example an attachment to pollution-free air, pesticide-free food or BSE-free beef have not always been claimed.

appears to be little or no evidence that reinterpretation of the property right ever invalidated a survey exercise and indeed the prominence of prevailing property right to resources in most CV studies can be judged by the fact that the issue hardly ever rates a mention in many published studies.

Consistency of the measure used

Although the Marshallian demand curve is often used to approximate the theoretically exact Hicksian alternatives, it cannot (except in exceptional circumstances) be used as an index of utility change or in any compensation test. Between compensating and equivalent variation or surplus, it has been found that only the equivalent variation (surplus) can consistently rank monetary equivalents of welfare change (see Freeman 1993 pp59 for graphical exposition). Basically this is because it evaluates all utility changes from an initial position using the same set of prices whereas the compensating variation measures the monetary requirements for constant utility when prices change. The result is that it is possible to arrive at a non-unique welfare measure for the change. The relevance of this distinction for exact welfare measurement (and therefore for CV question format), has been raised in the literature by Morey (1984). Essentially, the possibility for deriving a non-unique money metric using to a welfare change when these are approximated (in the case of a quantity constrained good), by compensating surplus measures implies that Hicksian equivalence measures be used instead. In other words, from figure 2 above, the CVM format for a quantity increase should concentrate on asking the WTA (ES) for foregoing a proposed increase or WTP (ES) for avoiding a quantity or quality decrease. In practice the distinction between equivalent and compensating measures may not matter a great deal, since the alleged non uniqueness would not be casually observed in a CV survey. Indeed, Mitchell and Carson (1989 p25) suggest the conditions to be remote in practice, stressing the lack of intuitive appeal in asking equivalence measures[15] and the lack of precision in CVM to discern ranking problems. Notwithstanding these findings, equivalent surplus is the preferred format of the surveys in following chapters.

The uniqueness of the measure used.

The uniqueness discussion does not immediately guide a welfare choice but is more important for understanding how multiple price or quantity changes should be evaluated in theory.

The uniqueness property pertains to the resulting consumer surplus measure in the context of a path adjustment constituted by a single or simultaneous multiple price or quantity changes. As demonstrated by Johannson (1987), the line integral for multiple price changes can be path

[15] This is an odd view and presumably because of the implied property right

independent providing certain conditions for the relevant cross price elasticities are observed. A sufficient condition for this to hold is that the marginal utility of income is constant with respect to those parameters which are changed. For a single quantity change, both surplus measures are unique. In other words money measures of welfare are unaffected by the sequence in which several environmental goods are executed within one project, although the marginal value of one quantity change must be evaluated subject to all previously considered changes.

Willingness to Pay or Willingness to Accept?

Taken together the foregoing do not obviate the need for the CV researcher to investigate specific local conditions for deciding on an appropriate question format. Probably the most compelling rationale is provided by existing property rights (actual or perceived) over the good in question. However, for many environmental goods the relevant sample frame are likely to be sufficiently disinterested not to consider their rights vis a vis the subject good. In other words, if one thinks one can get away with a particular incentive compatible format without invoking protest responses related to the right to ask WTP/ WTA questions then this is probably the right format. Some reasons for apparently invalid responses irrespective of question format are reviewed in a later section.

Having considered the theoretical construct underlying CV questions, the following sections move to a post-survey stage of data analysis and design criteria for exact welfare measurement.

3.4 Data analysis in discrete choice contingent valuation

Abstracting from the nature of use, the ultimate aim of a CV exercise is the calculation of a valid mean or median welfare measure to be aggregated over a relevant affected population for use in benefit cost analysis or natural resource damage assessment. In both cases, optimal resource allocation depends on the derivation of a measure which is theoretically valid, consistent, and can be shown to be resistant to number of arbitrary assumptions such as sample size and treatment of extreme responses. Consistency is particularly important since CV- derived measures are to be subject to increasing scrutiny in legal fora. Yet, the lack of consistency found in a review of CV studies compounds existing scepticism of the approach. Convergence of methodology is therefore of some importance. This section reviews the necessary assumptions made to derive Hicksian welfare measures from CV data.

3.5 The open-ended question format

As the name suggests open ended responses are a freely stated valuation in response to the type of question asking:

"what is the maximum amount of money you/your household would be willing to pay to per year have/avoid this change[16]?

Following from the fact (or perhaps article of faith), that the investigator assumes respondent preferences over the good in question to exist with certainty, open-ended responses should immediately give the investigator the information he is looking for. The fact that the utility function is not defined for some goods, seems to demonstrate discontinuities or reversals, is the crux of many difficulties with CV and will not be a concern until a later section. Of more immediate concern is the so-called "cognitive load" involved in the transition to an alien market environment, composed of (in the case of a non use survey), an unfamiliar good, hypothetical outcome and the absence of any ball-park figure or value cue[17]. Taken together these elements are said to reduce the incentive for respondents to appraise their preferences leading to random responses, extreme bids or zeros or else to engage strategic overstatement as a way of in free riding or simply as a "costless way to make a point" Arrow *et al* (1993).

Deriving the mean or median from open-ended data is generally straightforward. Judgement is however required over the presence of outlying responses which may need to be trimmed from the data set, thereby opening up the possibility of inconsistency with the Hicks-Kaldor criterion, if the value is genuinely held. Mitchell and Carson (op cit.) present trimming procedures for a robust estimator of an expected mean. In most studies data treatment is typically ad hoc and dependent only on internal consistency tests such as ratio of WTP amount to deposable income.

In determining the theoretical validity of responses, modelling open-ended data is also problematic in presence of outliers. An OLS regression of bid amounts on explanatory variables, assumes that errors are identically and independently normally distributed. Where this is not the case, parameter estimators and associated variance are no longer asymptotically efficient and F and t tests no longer valid.

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[16]The respondent will previously receive a description of the good, a change scenario and will have been reminded of other constraints on his response. The questionnaire will also suggest an appropriate payment vehicle.

[17]Payment cards are frequently used to circumvent this problem, although as shown by Cameron and Huppert (1989), resulting responses should be analyses as censored rather than continuous data.

Where outliers cause OLS properties to break down, robust non linear methods may be available (see Judge *et al* p.887). One method used in CV is a Box Cox transformation of WTP bids (Box and Cox 1964) to normalise the errors and reduce the influence of large bids (see Schulze *et al* 1990, Grosclaude and Soguel 1994). The approach is valid provided the true error term is really normal after controlling for systematic influences on WTP. Kanninen (1995) expresses some caution over the use of resistant fitting techniques which can substantially bias maximum likelihood parameter estimates.

At the end of the day the rigour of such massaging depends on how much weight the investigator wishes to put on the results of an open-ended survey. Though the method has much to recommend it in terms of transparency, the foregoing criticisms have made it unfashionable. Small open-ended surveys are typically carried out as pretests to determine the range of the DC bid vector, and to provide pointers on how the bids should be distributed. Moreover, response patterns can be indicative of wider problems such as poor good definition. Although there is no reason to suppose that WTP responses should be asymptotically normal, (assuming heterogeneous response categories in the sample), the observation of a bimodal distribution suggests polarisation of bids based on conflicting information sets about the good. McClelland *et al* (1992) have shown how this variance can be collapsed using full information surveys.

3.6 Discrete choice data

In recent years, the dichotomous choice format has become the method of choice in most CV applications and focus of rapid empirical development. Analysis using the DC in CVM questionnaires originates with Bishop and Heberlein's (1979) original goose hunting experiment, which has evolved into a referendum variant, offering respondents the opportunity of saying (or voting) yes/no to the following type of question:

> would you be willing to pay $X? Or, if there were a vote tomorrow on a programme that would cost you/your household $Y would you vote yes or no?[18].

(amounts X and Y pre-specified and systematically allocated to sub-samples of respondents).

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[18]There is a subtle difference in these questions related to whether the survey should ask a respondent's willingness to pay, or ask him to consider the price of the good or programme. Arguably, willingness to pay involves increased introspective scrutiny of preferences, while a vote may evoke some consideration of the common good.

Responses provide qualitative data censoring the respondent's true WTP within bounds and can be modelled using a variety of methods developed in bioassay, product reliability and labour economics. It is also possible to model responses within the utility-theoretic framework described earlier.

The popularity of the DC approach has grown because of its perceived incentive compatibility, Hoehn and Randall (1987). Such choices mimic everyday purchase decisions and this fact seems to have been the strongest argument leading to the endorsement of the referendum variant for natural resource damage assessment by the NOAA panel[19]. The market-like scenario reduces the cognitive-load, and has been shown to increase response rates (Hanemann and Kriström 1994). From an administrative point of view, the format is also conducive to use in mail surveys, which are the least costly survey alternative.

The downside of the approach - and essentially the main part of this chapter - is that relative to a set of open-ended responses, the information provided in a DC data set is very sparse, requiring a considerable amount of inference on the part of the investigator when modelling the set of binary responses and arriving at the expected WTP. More recently, the apparent incentive compatibility of the DC format has been re-examined with a view to determining the extent of any starting point bias. Farmer and Randall (1995) offer other cognitive models of starting price behaviour that might explain a consistently observed disparity between open-ended and closed ended formats. Meanwhile Fisher (1994 p.8) notes "the case for closed-ended CV responses being free from strategic bias, has not been made either in theory or by empirical findings". Cummings *et al* (1995) question the incentive compatibility advantage as a result of real versus hypothetical trials in an experimental setting. An empirical comparison of the difference in response formats is deferred until the following chapter.

Modelling framework

Within the neoclassical framework there are basically two approaches for analysing DC responses. These are Hanemann's utility difference approach (Hanemann 1984), and Cameron's alternative expenditure or variation function method (Cameron 1988). McConnell (1990) has linked both and shown the conditions under which the approaches are dual to each other. This section concentrates on the former.

[19] Set up to pronounce on the validity and role of CV in the wake of the Exxon Valdez controversy, this panel included two nobel laureate economists. Note that the use of CV in natural resource damage assessment remains a peculiarly US practice.

Hanemann's approach provides a convenient link between existing qualitative response models developed mainly in biometrics, and the random utility models of choice used extensively in other fields of economics (see McFadden 1976 for a review of the origins of the latter). The approach begins with an assumption about a respondent's indirect utility function $U(J,Y;S)$[20] where S represents a vector of socioeconomic characteristics, Y is income. In response to a referendum/closed ended question, $J = 1$ if the respondent is WTP the specified sum \$A to achieve or avoid a proposed quality change, or $J = 0$, if the respondent declines to pay \$A for the same.

There is an observable or systematic element v(.), and a random element 'ϵ' (accounting for unobservable taste parameters), that influences utility and therefore responses to CVM questions. The assumption is that the individual respondent knows his own utility function with certainty, but that it will contain some components which are unobservable to the investigator (Hanemann 1984).

A 'yes' response to the DC question "are you willing to pay \$A?" reveals that:

$$v(1,Y\text{-}A;S)+\epsilon_1 > v(0,Y;S)+\epsilon_0$$

or

$$v(1,Y\text{-}A;S) - v(0,Y;S) > \epsilon_0 - \epsilon_1$$

In other words, the random WTP probability depends on a utility difference part (ΔV) and a stochastic error component represented by F^n, where $n = \epsilon_0 - \epsilon_1$. The latter is typically assumed to be logistically distributed, the form which arises from the difference between two independent and identically distributed random weibull variables (Domencich and McFadden 1975). Using the cumulative form of this distribution yields a common logit model wherein the probability of an event taking place (in this case WTP response), is monotonically linked to the selected utility difference known as the index function.

$$P_y = (1+e^{-\Delta V})^{-1} \qquad (1)$$

Alternatively $P_y = F^n(\Delta V)$ and $P_n = 1 - P_y$, can be modelled parametrically using any one of a number of distributions including the normal distribution giving a probit model.

An equivalent approach is to consider the cumulative distribution of the random WTP variable itself,

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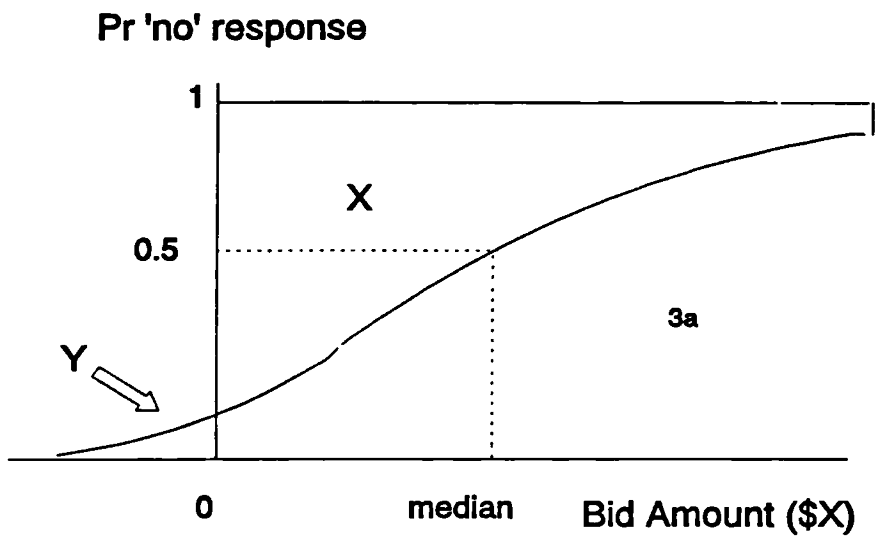
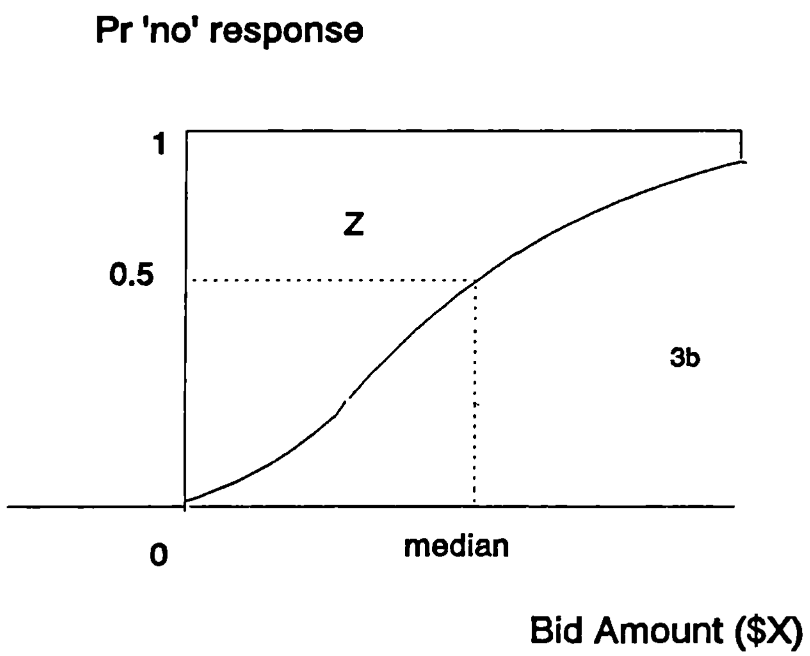
[20]Assuming market prices to be constant allows us to omit a price variable. Assume also that standard requirements of monotonicity, quasi convexity and homogeneity in the relevant arguments are met.

Figure 3



c.d.f. when WTP can take positive (X) plus negative (Y) values



c.d.f. when WTP is restricted to take positive values (area Z) only written G_{WTP} (\$A), which traces the probability that WTP $\leq$ A. Therefore 1 - G_{WTP} (\$A) gives the probability that A < WTP , and the respondent will accept the suggested price \$A (Kristrom 1990), thus:

$$P_y = F^n (\Delta V (A)) = 1 - G_{WTP} (A)$$

which indicates that the fitting of a binary response model such as (1) above can be interpreted as estimating the parameters of the distribution function G_{WTP} (A). Figure 3 provides an approximate graphical example of such a function, and illustrates the mean and the median welfare measures discussed below.

The Cameron approach

Cameron (1988) offers an alternative approach to modelling DC responses by focusing directly on the distribution of willingness to pay rather than starting with a specific form of the utility function. Her approach offers the attraction of sidestepping the functional form issue (for the index part of the model - see below), and in this respect the theoretical root of the approach is less clear. The approach is essentially dual to Hanemann's, with the difference being that the stochastic component is added to the cost rather than the utility function. McConnell (1990) elaborates further on when the approaches are exactly dual to eachother. Basically, this boils down to the treatment of income effects in the deterministic and stochastic parts of the respective approaches.

Cameron focuses on the cost function. Let c(s) be the individual's true compensating surplus with s denoting a vector of socioeconomic characteristics. This surplus can be considered comprising an observable part c(s)' and a stochastic part ϵ, thus $c(s) = c(s)' + \epsilon$. A respondent compares his own WTP to the asking price A, and is willing to pay A, if $c(s) \geq A$, or if $c(s)' + \epsilon \geq A$, so that the probability of acceptance can be given by:

$$Pr_y = Pr(\epsilon \geq c(s)' - A) = G_\epsilon (c(s)' - A)$$

where G_ϵ is the cumulative density function of ϵ. In the simplest case when the valuation function is linear, $c(s)' = s\gamma + \epsilon$, where γ is a parameter vector and ϵ is normally distributed $\epsilon \sim N(0, \sigma^2)$ then:

$$Pr_y = 1 - \Phi(\alpha A + s\beta)$$

where $\alpha = 1/\sigma$ and $\beta = -\gamma/\sigma$, and the parameters in the vector γ can be recovered from the maximum likelihood estimates of alpha and beta (Cameron and James (1987). Cameron claims that her approach allows the identification of the scale of the censored WTP variable σ
and also enables the calculation of elasticities of WTP with respect to exogenous variables. The downside of the approach, is that there is some uncertainty regarding the merits of being able to avoid specifying a utility function. Specifically this issue relates to the need to maintain consistency with demand theory. Because of it's apparent theoretical consistency, further analysis concentrates on Hanemann's approach.

In the *parametric* framework two model choices are apparent. First, the approximation of the error term in the modelling framework, which may be logistic, but may equally employ alternative distributions such as the normal (probit), lognormal or weibull (weibit) forms. The choice is a matter of judgement, and is related to the specification of the second choice element ΔV, the form of the utility difference and the combination of the two which best fits the data. A later section deals with the issue of the appropriate functional form for the index (utility difference) function, while discussion of fit statistics is left until chapter four.

3.7 Defining welfare measures

The acceptance (rejection) of a bid amount by respondent 't' in a DC format only allows the investigator to determine that the respondent's true willingness to pay is above or below the offer amount. Treating the respondent's true willingness to pay as a (for now generically distributed) random variable, it has been shown that the expected value of this random variable can be expressed in continuous form as:

$$E(WTP) = \int_{-\infty}^{\infty} b f(b)\, db = \int_{0}^{\infty} [1 - F(b)]\, db - \int_{-\infty}^{0} F(b)\, db$$

where F(b) is the cumulative density function representing the probability of a 'no' response and d(b), the probability density function (see figure 3). Since most subjects of a CV study are assumed to give positive utility, negative WTP is generally ruled out. In other words, the continuous form of the random variable is typically restricted to non-negative values (figure 3b).

$$E(WTP) = \int_0^\infty [1-F(b)]\, db$$

The expectation described by this function assumes that F(b) has a lower limit at zero (ie nobody will say no at WTP \$0) and an upper limit =1, as bid amounts tend to infinity (that is there is some bid amount high enough to induce a certain negative response). Graphically this implies a sigmoid function and suggests that the range of bids offered to individual respondents should be selected to insure that the extremes of the integral are 'banged down'.

The integrals are typically solved using Simpson's rule, or approximated by some trapezoidal equivalent (see Loomis 1988). Using the parameters of the appropriate functional form, estimated in (1), Hanemann (1984, 1989) defines mathematically equivalent formulas for the mean and median WTP. Assuming responses to be distributed logistically and using a commonly employed utility difference, table 1 provides the appropriate formulas using the parameters alpha and beta for a commonly used univariate linear model such as:

$$P_y = \frac{1}{1+e^{-\alpha+\beta A}}$$

McFadden (1994) provides equivalent parametric formulas for alternative distributional assumptions such as the Gamma and Weibull which offer the advantage of nesting more common forms.

Estimation by maximum likelihood provides parameters for the chosen model that maximise the likelihood of observing the responses that were actually observed. Using the non-linear logit command in canned routines such as LIMDEP involves regressing the log of odds ratio $\ln(\pi_i/1-\pi_i)$ on A where π_i is the proportion of yes answers (Kristrom 1990a). The method estimates parameters maximising the (log) likelihood function with respect to the model parameters, that is:

$$\ln L = \sum_{i=1}^{n} y_i \ln P_i + (1-y_i)\ln(1-P_i)$$

where P_i is the probability of the i^{th} individual responding 'yes' and is in the parametric approach such as a logit model, also a function of the distributional parameters.

For the model which is linear in income (see below), it can be shown that integration is unnecessary to obtain the expected willingness to pay (see Kristrom 1990 for a proof). Where the model is estimated with additional covariates, the conditional mean/median formula using alpha over beta for a model with a linear functional form, can be calculated using the grand coefficient alpha which is composed of the constant plus the coefficients of the other variables multiplied by the mean value of the appropriate variable. Beta is the coefficient on the bid amount variable.

3.8 Truncation

The expected values for the random WTP variable defined in equation 2 assumes integration to infinity. However, extreme right tail of the integral may contain an undesirable proportion of positive predicted probabilities for values beyond either the selected bid range or the highest amount stated in an open ended sample. Hanemann (1984, 1989) provides a convincing argument for integrating over the whole range of the function, namely that a positive response to the highest bid level, may not represent the maximum willingness to pay. In the context of a single DC question (precluding a further statement of a higher WTP), a downward bias can be removed by including the bid amounts predicted by the model that some respondents would, if they had the chance, hypothetically accept.

However the occurrence of extreme predicted WTP observations might be accentuated by model misspecification error[21], producing unexpectedly inflated mean values. In this case, the first thing to do is to alter the distributional assumption of the model to check the sensitivity of the mean to different predictions of the tail values. More commonly, (and given a preference for conservative estimates), in such cases many practitioners have truncated F(b) at a level deemed to produce 'meaningful' results. Justification for truncation and the preferred values are rarely articulated, but tend to be at the maximum offered bid level, typically defined as the highest reasonable bid observed in open-ended pretesting (eg Bishop and Heberlein 1979, Bowker and Stoll 1988; Garrod and Willis 1995). Alternatively, some percentile of the total distribution may be an appropriate point of

[21]The other factor influencing these observations, is of course the amounts that respondents are actually asked to consider, the bid vector.

Table 1. Welfare measures

Utility difference ΔV	Median WTP	Mean WTP	Mean WTP (positive predicted values only)
(1) $\alpha - \beta A$	$\frac{\alpha}{\beta}$	$\frac{\alpha}{\beta}$	$\frac{\log(1+e^{\alpha})}{\beta}$
(2) $\alpha + \beta \log(1 - A/Y)$	$Y[1-e^{-\frac{\alpha}{\beta}}]$	$Y[1-e^{\frac{\alpha}{\beta}}\frac{\pi}{\beta \sin\frac{\pi}{\beta}}]$	no solution
(3) $\alpha_0 - \alpha_1 \log A$	$e^{\frac{\alpha_0}{\alpha_1}}$	$\frac{e^{\frac{\alpha_0}{\alpha_1}}\pi}{\alpha_1 \sin\frac{\pi}{\alpha_1}}$	$\frac{e^{\frac{\alpha_0}{\alpha_1}}\pi}{\alpha_1 \sin\frac{\pi}{\alpha_1}}$

Notes: F^n the random WTP variable assumed distributed logistically. Utility difference forms (1) and (2) given by Hanemann (1984). Model (3) introduced by Bishop and Heberlein (1979). Alpha and beta are model parameter estimates. Last column provides WTP estimates excluding negative part of the integral. Note that with the log transformation of bid (A), $-1 < \beta < 0$ the mean of the distribution is undefined or infinite. Even with β less than -1 the right hand tail may be given disproportionate weight.

truncation (Hanemann 1989, Moran 1994), in which case the expectation can be defined as:

$$E(WTP) = X_{max} - \int_0^{X_{max}} P(X)\,dX$$

Where X^{max} is either the highest bid in the open-ended survey or some percentile of the distribution.

Truncation is common practice where for whatever reason the investigator prefers to use the mean rather than the median (see below), or when it can be shown that extreme value predictions outside the range of observed responses from a pretest cause radical changes in E(WTP). Unexpected acceptance of excessively high bids, and the influence of outlying observations are a concern in the use DC contingent valuation studies, and it is also apparent that the DC format merely disguises rather than solve the problem of outliers which are so problematic with the open-ended format. However the outlier problem does highlight the importance of one element that is under the investigator's control, namely, the bid vector (see below).

Strictly speaking the truncated form is not a correct statement of expected willingness to pay. If the range of integration is not carried out to infinity and there is good reason to truncate, the cdf should be normalised accordingly (so that the expectation of the truncated random variable = 1) (see Boyle *et al* 1988).

Normalization using a constant $K = 1/F(X_{max})$ redefines the expectation of the truncated random variable to be:

$$E(Z) = \int_0^{X_{max}} [1 - F(z)/F(X_{max})]\,dz$$

Figure 4a shows that this reduces the area of integration (and the expected value) relative to the that implied by the non-normalised truncation, when $F(X)_{max}$ does not actually approach one. In other words, the normalised distribution yields a lower expected value.

This normalization corrects a potential bias in the area of integration which can only be assumed to be negligible if $1/F(X_{max})$ actually approaches one. It has been suggested that normalisation may help to narrow the observed disparity between welfare estimates derived from open and closed-ended welfare measures (Leon 1995).

The assumption that $F(X_{max})$ approaches unity seems to be common in most examples of truncation.

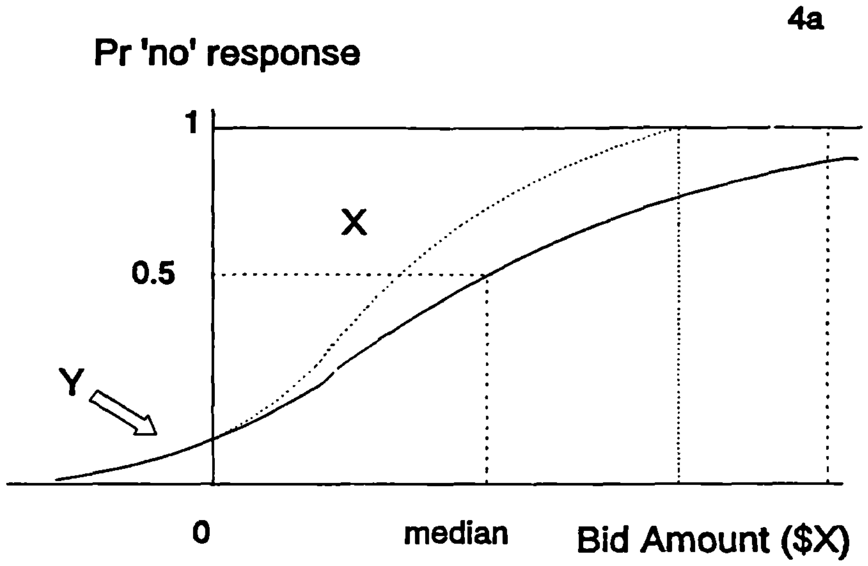
Garrod and Willis (1995) for example, justify truncation by reference to the 'observed distribution of the [pretest?] data' which tells them that the maximum willingness to pay is around £1000. Discrete choice observations beyond this amount presumably add nothing to the mass of the pdf and therefore no bias is entailed. What most studies do not seem to clarify, is the need for normalisation suggested by a high proportion of acceptances at the point of truncation or higher bid amounts. Ignoring normalisation and arbitrary truncation to avoid high acceptance rates for the highest bid level results in the underestimation of E(WTP), which is essentially due to the disparity of the assumed distribution of WTP implied by the selected bid range, and the actual sample willingness to pay responses. Ignorance of the shape of the underlying distribution of the latter, therefore raises the importance of accurate bid selection (Bowker and Stoll 1988, Cooper and Loomis 1992, Duffield and Patterson 1991). In the case of fat tails, the distribution of bids is too close to the centre of the mass of the actual willingness to pay distribution. Conversely, setting too many bids at extreme intervals at distance from the mass of the underlying WTP may be effective at closing the integral, but at the expense of losing the information of too many observations of one sign in the tails (Kannien 1995). In this case, the dependent variable shows little response variation to bid amounts, while there is always the possibility that an unfeasibly high amount may actually be accepted, thereby preventing the upper tail of the estimated cdf asymptotically approaching one as fast as expected. There is now a modest literature on optimal bid design in DC CVM which will be addressed at several points in this and following chapters.

The decision to use a pretest truncation indicator is important in the sense that it places a lot of weight on the pretest sample to determine location and the precise shape of population WTP the distribution. It is clear that the above observations are essentially artifacts of the design process. Several authors have raised this issue and stressed the problems involved in bid distribution process require further research. At present there is no consensus on design criteria. From the author's perspective the number of potential pitfalls in DC design and implementation lend considerable appeal to the open-ended format, particularly as the latter is typically observed to produce conservative welfare estimates relative to the DC variant.

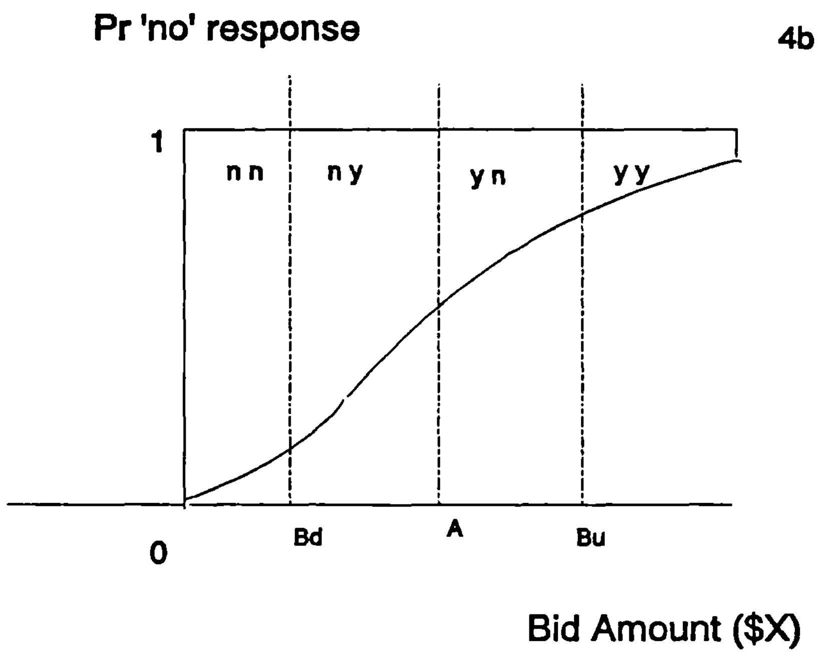
3.9 Functional form

There is a debate about whether responses to DC questions should be modelled in a form which is strictly consistent with utility theory. Consistency involves specifying a functional form for the utility difference index ΔV of (1), that can be traced back to an indirect utility function, and by extension the axioms of consumer choice. Strict observation of theory limits the way in which income and other

Figure 4



E(WTP) using a truncated and normalised c.d.f.



E(WTP) derived using an interval double-bounded format. A is the initial WTP amount, followed by amount Bd if the initial response is 'no' or Bu if it is 'yes'.

explanatory variables enter the index function of the response probability formula (1) above. Certain forms make the Hanemann and Cameron approaches dual to one another. It has also been shown that certain functional forms may impose additional restrictions, such as limiting WTP relative to respondent income and more controversially addressing a priori expectations on whether the good being valued should necessarily provide a welfare improvement. Such considerations 'tighten' the class of utility functions involved in utility theoretic modelling (Hanemann and Kanninen 1996). From a purist's point of view, welfare measures derived from ad hoc models are meaningless. The pragmatic view is that as long as the response probability function slopes upwards and coefficients have acceptable signs, then some minimum theoretical requirements are met, Sellar *et al* (1986).

Qualitative choice models have their origin in biological trials where a censoring response to a stimulus variable (perhaps a chemical dose administered to a subject), need not obey any underlying theory. In highlighting the utility-theoretic basis for DC responses, Hanemann (1984) apparently also showed that the introduction of qualitative choice modelling by Bishop and Heberlein involved the use of a functional form that was strictly inconsistent with a random utility framework. Specifically it was revealed that a form of v that is linear in income v (j, y; s) = $\alpha_j + \beta y > 0$, where j = 0,1[22], y is income[23] and the WTP amount A, generates the utility difference $\Delta V = (\alpha_0 - \alpha_1) - \beta A$, (since income drops out of the difference) giving a statistical DC model in (1) above, where $\alpha = \alpha_1 - \alpha_0$, is only identifiable as a difference, and that this is the only form of the difference which explains responses independent of the individual's income y and thus according to Hanemann 1984 p334 "income effects do not occur". Note that the mean formula for this form given by $\alpha_1 - \alpha_0 / \beta$ can be interpreted as saying the average willingness to pay is obtained as the utility change converted into monetary terms by division with the marginal utility of money β (Kristrom 1990a).

Another consistent form including income, derives from the logarithmic function of v (j, y; s) = $\alpha_j + \beta \log y$ $\beta > 0$ giving $\Delta V \approx (\alpha_1 - \alpha_0) + \beta A/Y$, (see Hanemann 1984). Hanemann and Kanninen (1996) show how both cases are nested within a Box Cox functional form originally adopted by McFadden and Leonard (1992) :

$$U_q = \alpha_q + \beta_q \left[\frac{y^{\lambda - 1}}{\lambda} \right] + \epsilon_q$$

where q=1 or 0, and for lambda =1 provides the linear model above, while lambda =0 the log-

[22] Where the vector s has been suppressed.

[23] McFadden and Leonard (1992) correctly distinguish this as post-tax income.

logistic model.

The log-logistic - including ln(A) - was in fact the form used in Bishop and Heberlein's permit experiment. Thus:

$$P_y=\frac{1}{1+e^{-\alpha+\beta \ln A}}$$

which it was thought, could not be generated from any explicit form of the utility model v (j, y; s), j = 0,1 (Hanemann 1984). Such a model has convenient properties, specifically the log of bid by definition rules out predicted probabilities of negative willingness to pay, possible with a linear model. Literally dozens of studies report parameter estimates from this model and other distributional variants and its use avoids the need to truncate the negative portion of the continuous random variable (figure 3a) and the potential error in mean calculation (Johansson *et al* 1989)[24]. One problem that does arise however, is that the power transformation of the bid values implied by a log-logistic, makes the mean estimate extremely sensitive to high predicted tail values. A similar problem also arises using a lognormal distribution and the two models are in fact hardly distinguishable.

This observation has given rise to a debate over whether functional form really matters as long as the preferred form (utility theoretic or ad hoc) yields unbiased estimates of Hicksian surplus with a minimum variance. Following Bishop and Heberlein there have been many studies specifying the functional forms without direct reference to a utility function (Sellar *et al* 1986; Boyle and Bishop 1988), and others comparing the performance of theoretic relative to ad hoc specifications using a range of fit measures (Bowker and Stoll 1988; Boyle 1990). There is no real consensus on the need for models to be utility theoretic. Empirically Boyle and Bishop found that specifications consistent with utility theory 'may not provide statistically significant coefficients, and some coefficients may have the wrong signs'. Bowker and Stoll's findings are particularly relevant in that in some cases utility theoretic models produce negative medians, which although not a nonsensical finding if the good being valued gives negative utility (eg venomous snakes)[25], is slightly worrying if the good being valued (in this case Whooping Cranes), is supposed to provide positive utility. Equally of

[24]More specifically, the latter form of truncation means that the probabilities do not sum to one.

[25]And supposing respondents were actually invited to indicate whether they wanted to be compensated (as opposed to were WTP) for the species' conservation.

concern is the finding by Sellar *et al* (1986), that some specifications resulted in estimated coefficients which gave upwardly sloping demand functions. Thus some consistency with 'normal' price quantity relationships provides a minimum requirement on model selection[26].

As it turns out, Hanemann's original observation appears unfounded, as it is possible to find better specifications of the utility function. Indeed the restriction of a linear utility function has no theoretical basis, and there are certainly many indirect utility functions that have the usual neoclassical properties which are not linear in the parameters. Furthermore Hanemann and Kanninen (1996) now assert that the Bishop and Heberlein model can in fact be motivated by a general class of separable utility functions allowing response probabilities to be independent of income (see Hanemann and Kanninen 1996). In addition, such forms may accommodate restrictions on responses relative to income and expectations regarding the sign of the welfare change.

This finding does not change the fact that the selection of a functional form in modelling is likely to remain a matter of judgement combined with plausibility checks on elicited surplus (eg the logic of a negative median should be judged according to good being valued). Allowing the data to tell its own story, points to the selective use of many forms and discretion in the calculation of conditional and unconditional measures of central tendency[27].

3.10 Mean versus the median

The issue of the appropriate measure of central tendency in DC welfare measurement is basically the choice between the mean represented by the integral of the cdf of the random WTP (represented by areas X or Z in figure 3), or the median represented by the WTP at which Py= 0.5, or some other percentile of the distribution. In CV models using symmetric distributions these measures may coincide (see Table 1). Elsewhere, the eventual choice in any CV experiment is essentially a value judgement. However like the issue of truncation, the choice can be motivated by both statistical and

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[26]Sellar *et al* essentially define a total value function by substituting equation 1 into 5 and finding the first derivative inverse Hicksian demand curve which must be non-negative. The second derivative Hicksian demand curve must be negative for a downward sloping demand curve.

[27] This also suggests a flexible or generalised approach in modelling. For example, rather than the simplified Bishop and Heberlein log-logistic, a power transformation of the odds

$$\left[\frac{\pi}{1-\pi}\right]^{\lambda}=\alpha+\beta\ln(Bid)+e$$

can be estimated using the Box-Cox transformation to define the appropriate form according to the significance of lambda.

(welfare) economic considerations.

The choice between the mean and the median may be influenced by two related issues. First, the determination of the best estimate of welfare change over all members of an affected population. Second, aggregation of gains and losses over all members of the population for the purpose of benefit-cost analysis. The second issue is typically implicit in the decision on the first, and follows the basic precept that the sum of compensating variations necessarily enables a potential Pareto improvement. Reasons to dispute the latter are advanced by Blackorby and Donaldson (1990).

The median is frequently the only reported welfare measure in several high profile CV studies (see for example Carson *et al* 1994ab). The median approximates the WTP value at which 50% of the population would vote yes and 50% of the population would vote no. In economic terms, the mean is the amount of consumer surplus respondents would receive on average if they were provided with the good in question at a zero price.

In modelling terms the basic difference between the median and the mean is that the former depends only on the location of the response probability curve at the 50% probability, while the latter depends on the location of the whole c.d.f. truncated or otherwise. The median is typically more resistant than the mean to outliers or unusual data observations, and cases where a high proportion acceptances of highest bid. Its use avoids the need to justify truncation decisions if the integral is unduly sensitive to outliers of wherever the response probability function has no closed form representation. Inspection of the data set used in several studies (e.g. Carson 1994b) reveals a fat tail problem, which is (rightly or wrongly) circumvented by use of the median. In addition, it has been found that the mean can be unduly sensitive to apparently minor differences in the method of estimating the structural model, such as generalised least squares or maximum likelihood (Hanemann 1989 p 1060). Except in the most extreme case of price insensitivity or model misspecification - where the response function slope is almost flat or doesn't cross 0.5 - the median is nearly always identified. But the latter issue highlights how the median can also be sensitive to parameter values and therefore regression mis-specification. Similarly, the finding of a negative median requires the judgement about the sign of welfare change for the majority of respondents (eg poisonous snakes)[28]. The issue of price insensitivity is a particularly interesting issue in relation to complex goods and we return to it later.

[28] In fact if respondents are not asked to state a WTA for the implicit environmental improvement, then a resulting negative median should be treated as a product of an inappropriate model.

The choice between the mean or median may derive from a prior choice of a social welfare criterion. As pointed out by Johansson *et al* (1989) the median voter approach may be inconsistent with a Hicks-Kaldor potential Pareto efficient outcome in the sense there is a possibility that a genuine outlying observation (representing someone with a high stake in the change), may simply be disenfranchised by its use. In other words, the use of the median for aggregation immediately implies the adoption of a social choice rule which approximates majority voting (or super majority voting if an alternative quintile were used). This interpretation has some affinity to the NOAA recommendation on the use of a referendum format to frame DC questions (NOAA 1993).

However, when the distribution of WTP is skewed, as seems to be a common occurrence with CVM data, there is not much point using the median as the mass in the tails of the distribution will be ignored. This also implies that the choice of error form in the random utility model can have a substantive economic significance.

It has been argued that aggregation of the median does not have the same "natural" interpretation as a mean aggregation Kristrom (1990a). Use of the mean is perhaps more in tune with the random utility framework and imperfect knowledge of respondents' utility functions. For a given error distribution, the model of the discrete dependent variable is conditional on the regressors which describe the sample, including the those people who accept high WTP amounts. It would therefore seem unreasonable to search for the best fitting conditional distribution only to ignore this information. On the other hand, the reverse may be true. The presence of different respondent categories may be a good reason to represent all 'mental models' by that of the median voter.

It has been suggested that there is a distinction between surplus measures needed for benefit-cost analysis and for damage assessment, Carson *et al* (1992). Specifically, their view is that the median WTP is suitable for the former, but that mean WTA should be used for the latter because of the implied property right and the legal requirement to restore damaged parties to their original state. This assertion is not particularly relevant in alternative institutional environments, nor is it consistent with the implicit redefinition of the property right in damage assessments such as the Exxon case where WTP (equivalent surplus) was used to get at a welfare measure for a hypothetical ex ante rather than a real ex post event, (an approach subsequently endorsed by NOAA 1993). Furthermore there is apparently no justification for supposing that parties to any environmental change are in any sense more disinterested than litigants of natural resource damage whatever the relative proximity to the injury.

The forgoing sections show that the derivation of a welfare measure from DC data requires several value judgements related to the choice of the measure of central tendency, the degree of adherence to theoretical requirements and the need to truncate undesirable model predictions. The following four sections review several further modelling issues related to the refinement of welfare measures.

3.11 Confidence intervals around mean/median WTP

Welfare measures are random variables as they are dependent on the estimated model parameters and by extension an associated error approximation. Estimates of the variance of welfare measures have not always been apparent in many CV studies, which from a welfare perspective is unsatisfactory. The calculation of confidence intervals around the computed welfare measure has direct policy significance[29], and considerable effort has recently been directed to refining methods. Cooper (1994) provides a summary of methods. The method proposed by Krinsky and Robb (1986) to calculate confidence intervals around the elasticities of consumer or factor demand functions first used in CV by Park Loomis and Creel (1991) is described here. Basically all methods commonly used in CV are a variant on bootstrapping and jackknifing procedures for resampling from initial data sets or vectors of parameter estimates. A guide to these techniques can be found in Sprent (1989).

Krinsky and Robb note that functional forms such as the translog now commonly used in demand or production analysis, no longer produce elasticities as parameters. Instead these are typically non linear functions of parameters that have to be estimated either by a potentially erroneous linear transformation or by alternative methods. This is analogous to the situation in CV, since the solutions to E(WTP) are non linear functions of the maximum likelihood parameters of the logit model. It is therefore appropriate to use the method suggested by Krinsky and Robb to estimate the variance of E(WTP).

The method basically consists of the generation of a large sample of regression model coefficients by random draws on a multivariate normal distribution determined with a mean and variance-covariance according to the original model maximum likelihood estimates. For each drawing of the parameter vector E(WTP) can be recalculated according to the favoured equation to derive an empirical distribution of WTP. A $(1-\alpha)$ confidence interval is obtained by ranking the vector of calculated WTP values and dropping $\alpha/2$ values from each tail of the ranked vector. Krinsky and Robb suggest 1000 drawings is sufficient to generate an accurate empirical distribution, although it may be necessary to

[29] In the project context this might be for the calculation of an Internal Rate of Return where the confidence interval actually spans the cut-off rate.

increase the number of draws until no further change to the confidence interval is discernable. Generation of a sample of parameter vector estimates is automatic in a programme like LIMDEP, using commands to save the estimated coefficients and the variance covariance matrix. Desvouges *et al* (1992), show that It is also possible to bootstrap from the original survey WTP values instead of a vector of model coefficients. This has a favourable property of avoiding the assumption that the original coefficients are the 'true' coefficients. Their method performs a monte carlo resampling with replacement from the original data set to constitute a series of new random data sets of the same size as the original. Each of these can then be used to recalculate new model coefficients, each time to be used to recompute the WTP. The repeated re-estimation of the coefficients is somewhat cumbersome, but it allows variation in repeated samples from the same population to determine the variance.

This method only differs slightly from the jackknife in the use of monte carlo generation of samples. Using the jackknife procedure, samples are generated by systematic omission of one observation in the sample.

Note that both of the latter techniques rely heavily on the justification that the sample cdf is the maximum likelihood estimator of the population distribution function of WTP in order to calculate the sample variance analogue to the population characteristics. This fact leads appropriately onto one of the main determinants of the sample likelihood estimator the bid range that respondents are asked to accept or reject.

3.12 Bid Vector

One issue that seems to be fundamental in achieving a well defined response curve is the choice of the bid vector to cover the relevant location of the true WTP represented by the sample mean or median. As previously stated, biased estimation of E(WTP) is essentially due to the disparity of the assumed distribution of WTP implied by the selected bid range and the actual sample willingness to pay (and both may be at odds with the 'true' population WTP)[30]. Ignorance of the shape of the underlying distribution of WTP therefore places considerable weight on the information derived from any open-ended pretest survey. Where no pretest is available, bid placement becomes a guessing game.

[30] A study conducted by Davis and O'Neill (1992), demonstrates the combined problems of an inappropriate bid vector and modelling procedure, producing a median not encompassed by the vector.

The general view is that careful choice of bid vector obviates the need to truncate a fat tail (thereby potentially underestimating E(WTP)), or the need to rely on the median. However there is always a considerable unknown element in DC design as the investigator is never sure not only about how the tails should behave, but also about the effect of certain round bid amounts. Jakobbson (1994), for example notes that examination of response proportions for the WTP for the preservation of Leadbeter's Possum reveals obvious clustering around certain round values - or focal points, see Green *et al* (1995) - which she suspects is a by-product of valuing an unfamiliar good. Similarly unforeseen events such as high non response rate at any particular bid[31] can lead to divergences between the bid design assumption and subsequent best fitting model. The remainder of this section offers observations on bid design criteria.

The choice variables in the bid design process involves the determination of the maximum sample size (N), the number of bid levels (partitions of the assumed WTP distribution) m, their location along the line of real WTP values (corresponding to the actual individual bid amounts b_1 to b_n) and the number of individual respondents assigned to each m to receive each b_i where $m \leq N$. These variables are best thought of as a histogram which approximates the underlying WTP distribution.

Essentially, bid design is tantamount to selecting one of the explanatory variables which along with the underlying parameters of the assumed distribution of the WTP[32] and the sample size, endogenously determine the efficiency and bias of the model parameters used to determine mean WTP. This is not a regular procedure in econometric research. The choices can be governed by the information sought, such that if the median is of primary interest, then we should ideally seek information close to it. Alternatively the determination of the mean implies that the determination of the tails of the distribution is of importance. But in the absence of reliable pretest information or some robust Bayesian reasoning (see Kristrom 1994), design is stymied.

An immediate problem apparent in many studies is that the investigator rarely has a clear idea of the most obvious variables such as the sample size which may be determined by time and financial constraints. The use of mail surveys allows the investigator to be relatively certain in posting bid amounts to a sample, but even here the response rate is not guaranteed. Thus if the sample size is central to the design structure, non-respondents must be systematically assigned a 'no' to the posted

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[31] One might for example, have the problem of a low or zero response rate for a particular bid value in mail surveys or due to interviewer problems.

[32] Continuing for the time being with the assumption of a parametric modelling approach.

bid to avoid uncertainty. The use of dynamic design criteria to update optimal bid vectors is one way to circumvent this problem, but there appears to be a singular lack of research in the area. Despite a growing literature on optimal bid design, in practice most CV studies have used ad hoc bid vectors.

3.13 Optimal Bid design

Original DC exercises used ad hoc methods of selecting typically 5-10 bid amounts usually (but not always), equally spaced within the range indicated from an open-ended pretest survey. An alternative method used in some early studies including that of Bishop and Heberlein involves a log-linear spacing of bids, such that as bids increased spacing increased thereby correctly recognising, that placement (although not the number of bids at each spot along the real line), should roughly correspond to some probability density.

Boyle *et al* (1988) appear to have been the first to attempt a stratifying procedure explicitly to identify the whole response curve, while Cameron and Huppert (1991) simulated the effect of different bid ranges and demonstrated the 'luck of the draw' nature of selection on resulting WTP. Optimal bid design in DC appears to have commenced with Duffield and Patterson (1991) who attempted to circumvent the problems arising in ad hoc bid construction. Notably their approach takes bid amounts as given and determines the unique number of bids levels for a given sample size such as to minimize the variance of the expected willingness to pay. Additionally they note the import of design criteria used in biometrics to select dose levels to minimize the area of the asymptotic joint confidence region for the regression parameters. Cooper (1993) reviews the relevance of existing statistical design criteria such as C and D optimality (see below) for the particular problems involved in the derivation of acceptable welfare values in DC CVM. He also proposes a design algorithm DWEABS - (Distribution Weighted Equal Area Bid System), which is used in chapter four and which is one of several approaches compared by Elnagheeb and Jordan (1995). Hanemann and Kanninen (1996) and Alberini (1995), offer extensive tests of alternative design criteria.

As previously suggested, optimal design of the discrete choice bid vector is essentially driven by the desired properties of the estimated coefficients of the resulting model which ultimately determines the WTP estimator. The desired properties are the bias and efficiency of the estimates, which rely partly on the choice of a bid structure to minimize the asymptotic variance.

On bias, Kanninen (1995) provides view on what can be expected from poor bid design. Essentially the question may be answered by recalling that the mean and median calculations of Table 1 are provided by a combination of the model parameters alpha and beta. Bias in willingness to pay estimates will therefore depend on the direction of the bias in coefficients estimates resulting from an inappropriate bid vector. As it happens bias in both parameters may be offsetting, such that bid design does not substantially bias WTP providing the design does not promote the precision of one parameter over the other. A simple solution to avoid bias is to increase the sample size, but simulations with different bid vectors show that a concentration of extreme bids is inadvisable.

On variance, an intuitive appreciation of the problem can be gained by considering a typical linear regression model:

$$y_i = \alpha + \Sigma \beta_j x_{ij} + e_i$$

where y_i is a response variable, alpha and beta are parameters to be estimated and x_{ij} are explanatory variables which are typically given. Optimal design in CV can be likened to the need to ***design*** the explanatory variables in order to estimate efficiently the parameters of interest. A typical design is one that minimizes the variance of alpha and beta. Let X be the matrix $[1,x_j]$. It can be shown that the variance of the parameter vector β $[\alpha, \beta_1, \beta_2,..]$ is proportional to the information matrix $[X'X]^{-1}$ (see Judge *et al* 1988 pp201). Therefore the problem of minimising the variance of the parameter vector β reduces to minimising $[X'X]^{-1}$, in other words, choosing observations as far away from the mean of x_j as possible.

Moving away from this linear abstraction presents problems. In the case of non-linear models used for DC, the information matrix can be shown to be a non-linear function of a number of unknowns we are trying to get as an optimal solution to a minimum variance problem. More specifically, it can be shown that the Fisher Information Matrix[33] resulting from the second order derivatives of the likelihood function for the model parameters, is equally a function of bid values used (see for example Alberini and Carson 1994). In other words, we have to know something about the distribution of the response variable in order to choose the bids that yield the efficient parameters to describe it! This problem is evident from the (biometric) optimal design literature, and prevents its use in CV. A second best approach is to find the required information in a pretest survey.

From the optimal design literature two main methods have been widely cited in indicating the

[33]Recall that this matrix is the expected value with the reverse sign of the Hessian matrix of second derivatives, and that its inverse is the asymptotic covariance matrix of the MLE parameter vector.

percentile points of an underlying WTP distribution for the placement of bids to avoid efficiency problems Hanemann and Kanninen (1996). The C optimal design method involves specifying a function for the asymptotic property of interest such the variance of mean or median WTP and minimising the function with respect to bid values. Of course the design points are functions of the unknown underlying WTP distribution which cannot be identified before data collection. C optimality is therefore more appropriate for ex post evaluation of which points along the WTP distribution provide the most statistical information.

Extending the simplified example above, D optimal design involves jointly minimising the variance of parameters alpha and beta, by picking design points to minimize some function of the information matrix. It turns out that in the case of a non linear model the information matrix itself is a non linear function of unknown parameters of which will be part of the solution to the minimization problem. Again the method has some appeal for ex post analysis of the effects of certain bids on the objective function. Furthermore, as pointed out by Cooper (1993) this method produces two design points which is not ideal for fully identifying the response function.

The information requirements of optimal design procedures suggests a role for sequential design procedures up-dating bid vectors from at least one open-ended pretest. Kristrom (1994) describes the use of an algorithm for up-dating bid vectors (Robbins Monroe recursion), which has been used in a CV of the Stockholm archipelago. Such methods are not common and given that CV researchers will be restricted by budgets, rules of thumb can be provided by experimentation with different bid vectors and comparison of the asymptotic variance of the mean WTP (Kanninen 1995). Such an approach for a logistic distribution suggests that bias and variance may be reduced firstly by increasing sample size, secondly by keeping bids above the 15th percentile and below the 85th. Observations outside such a range may be wasted. It should be noted that such advice merely prevents a shot in the dark and cannot be generalised to all circumstances (e.g. for other distributional assumptions for which different percentiles will be appropriate). Furthermore, such placement may enforce truncation since bids will not be placed to verify the response pattern in the tail dictated by the chosen distributional assumption. Depending on the particular model, this may create its own problems. Optimal design criteria are given further attention in the following chapter.

This section has shown that existing optimal design procedures do not allow CV researchers to identify optimal bids before collecting data. There are many sources of error in the design of DC CV and several commonly recommended procedures are in fact value judgements that are typically not explained in papers. The lottery nature of bid design can only be reduced by following rules of thumb. These include the use of good open-ended samples in order to determine adequate bid placement in the tails for common distributions. Another important issue relates to the cautious use of models which are particularly sensitive to tail values. At risk of generalisation, CV researchers should use all prior information to identify the vital part of the distribution. This may include values elicited in other similar studies. In conclusion the advantages of the DC approach may be more than outweighed by some of the problems arising from bad design.

3.14 Further issues in CV design: double bounded and bivariate models

A recent modification to the DC format is the addition of a follow-up question in which an initial yes (no) is followed-up with a subsequent WTP amount higher (lower) than the first bid. Respondent willingness to pay is therefore identified by 'tighter' interval censoring combinations of yes yes, yes no, no yes, no no. which gives more information on the underlying willingness to pay and, can be statistically more efficient (Hanemann *et al* 1991, Kanninen 1995).

Extending the modelling procedure for the single response model given above for the random WTP G_{WTP} (A) which traces the probability that WTP $\leq$ A. (or a no response to offered value A), and $P_y = 1 - G_{WTP}$ (A). The extra question partitions the cdf for the random WTP evaluated in A between bounds determined by the initial and follow-up question. Respondent WTP is therefore in one of the four response sequence categories:

$$
\begin{aligned}
\text{Pr yes yes } (A,B_u) &= \Pr[A \leq C \text{ and } B_u \leq C] \\
&= \Pr[B_u \leq C] \\
&= 1 - G_C(B_u), \\
\text{Pr yes no } (A,B_u) &= \Pr[A \leq C \leq B_u] \\
&= G_C(B_u) - G_C(A), \\
\text{Pr no yes } (A,B_d) &= \Pr[A \geq C \geq B_d] \\
&= G_C(A) - G_C(B_d), \\
\text{Pr no no } (A,B_d) &= \Pr[A \geq C \text{ and } B_d \geq C] \\
&= G_C(B_d).
\end{aligned}
$$

as shown in figure 4b.

The relevant likelihood function for this example is given by:

$$L = \Sigma(d^{yy}\log\pi^{yy}(A,B_u) + d^{yn}\log\pi^{yn}(A,B_u)$$
$$+ d^{ny}\log\pi^{ny}(A,B_d) + d^{nn}\log\pi^{nn}(A,B_d)$$

where the response combinations are 1,0 binary variables indicating the relevant range of the random variable. As in the single response case, maximum likelihood estimates are obtained maximising the function for parameters of the model selected to define $P_y \equiv F^n (\Delta v (A)) = 1 - G_{WTP} (A)$. That is, the likelihood of a set of responses to a bid set is simply the probability that WTP lies in the interval between the accepted and rejected range. Defining this problem is relatively straightforward in LIMDEP.

Double bounded models have been applied by in several instances (eg Imber *et al* 1992; Carson et al 1992; Leon (1996)). Leon analyses the use of Survival models for interval censored data which are analogous to the modelling procedure set out by Hanemann *et al* (1991) and readily available for a variety of distributional assumptions in programmes like SAS.

As with the single case there is a related literature on the placement of bids in the double bounded case with a view minimizing parameter variance and that of the WTP estimator (Kanninen 1993; Alberini 1995). The double-bounded model may also be the initial stage of an iterative bidding process, and the question on the adequacy of two bounds has been examined by Cooper and Hanemann (1995) who show that most efficiency gains from extended questioning are gained going from a single to double bounded model. Nevertheless, the double bounded format may be somewhat problematic when responses to follow-up questions represent a disposition to a second question rather than a genuine response related to the good on offer. Hanemann and Cooper have attempted to circumvent this problem with an innovative one and half bound method which is based on telling respondents the cost range A_{Low} to A_{high} of a potential programme, but only following up 'yes' responses for initial offers in the lower range and 'no' questions where the initial offer was in the upper range. On average, the interviewer will follow up with a second bid only half as often as with a conventional double-bounded model. In simulated exercises Cooper and Hanemann find that the format offers the efficiency gains of a double-bounded format while potentially circumventing the problems associated with the affront of a second question.

An added complication in modelling is the suggestion by Cameron and Quiggin (1994), that follow-up responses may not be motivated by the same underlying distribution of WTP as the first response.

In other words, WTP is not immutable and may become unstable between a first and second questions as respondents engage in types of strategic behaviour that most CV researchers would prefer to forget. They introduce bivariate modelling procedures (see Judge *et al* 1988), which relax the assumption of identical distributions implicit in figure 3b and avoid potential estimation bias if WTP are in fact distributed differently in first and second responses. The determination of an additional correlation parameter ρ in theory allows the hypothesis of similar distributions to be tested. Thus where $\rho = 1$ first and follow-up responses are identically distributed as in fig 4b and the basic double-bounded (interval censored) model can be seen as a special case of the bivariate case[34].

In principle, one can choose any distribution function for Cameron and Quiggin's method. In practice, many distributions do not have natural bivariate analogues to the normal distribution although it has been shown that it is possible to approximate other forms by transforming the probit form, Amemiya (1985), Brown *et al* (1994a). Alberini (1995a) has investigated the bias resulting from erroneously fitting an interval-censored model in cases of a less than perfect correlation between distributions. On the basis of the mean square error of the mean WTP, it turns out that the interval-censored model is quite robust even for low values of ρ. This result combined with the uncertain power of the model test, means that a best strategy is to rely on a regular double-bounded model unless it is completely obvious that something bad is going on between responses.

Cameron and Quiggin's empirical findings of greater dispersion in follow-up responses leads to doubt over the incentive-compatibility of the DB model and aggravates existing credibility problems caused by potential yea-saying.

Further biases in the use of this format are likely in common administration formats such mail surveys, where respondents are free to look ahead for a second WTP amount - and probably adjust their views on the first amount proffered.

However Kanninen (1995) has recently suggested an advantage for salvaging poor initial bid design using a follow-up question. Basically a follow-up question gives a second chance to filter some information from respondents who overwhelmingly reply no to an initial extreme value. Interestingly Kanninen's results show a reduction in bias in parameter estimates between a single bounded (when bids were poorly placed in the right tail) and double bounded model using bids equal to double and

[34]There is in fact some doubt about the power of a regular t-statistic to test $\rho = 1$, and therefore to distinguish between an interval-censored and a bivariate model (see Alberini 1995a p.172).

half the initial amounts. The resulting difference in estimates of mean WTP is an empirical issue examined in chapter four.

3.15 Parametric, non-parametric or semi-parametric methods

Given the investigator's ignorance of the distribution of the underlying WTP and the utility difference structure, the need to specify either F^n and ΔV a priori, is regarded as a modelling restriction which may adversely affect parameter estimates and therefore bias WTP estimates. Non parametric and semi-parametric estimation makes the logit assumption less critical. Recent developments relaxing both parts of the model are essentially methods to allow sufficient generalisation of the bid curve.

It is difficult to state *a priori* the extent of any resulting bias by incorrectly fitting a logit model, but model choice is important for at least three reasons: 1) if one wishes to consider distributional aspects of welfare change; 2) if there is reason to believe that the sample is non-random, so that sample means of conditioning variables are not the same as population means; 3) if one wishes to transfer the WTP equation to a different population[35]. There may also be some merit in relaxing some of the more deterministic modelling elements inherent in economic theory.

Pommerehne and Hart (1994) make much the same points in stressing the fundamental advantage of the nonparametric (and presumably semi-parametric) approaches drawing on the field of evolutionary economics which advocates employing a viewpoint permitting rationality and individualism within groups. They point to the evidence provided by verbal protocol analysis (Schkade and Payne 1994), which suggests that subgroups of respondents are likely to be qualitatively differentiated in preference structures, and in the way they process information in a DC survey. Such differences are essentially overridden by the derivation of a representative sample bid function by parametric methods which traces out where any individual will switch from a no to a yes irrespective of the subgroup to which they may belong. In circumstances where respondent subgroups are qualitatively different, aggregation may be problematic. On the other hand, non-parametric methods which rely less on sampling respondents' personal characteristics for modelling, make the explicit assumption that the problem of different mental evaluations, can be circumvented by drawing the biggest sample possible. The only downside of this approach is of cost of large samples and the inability to extrapolate the result for any particular subgroup.

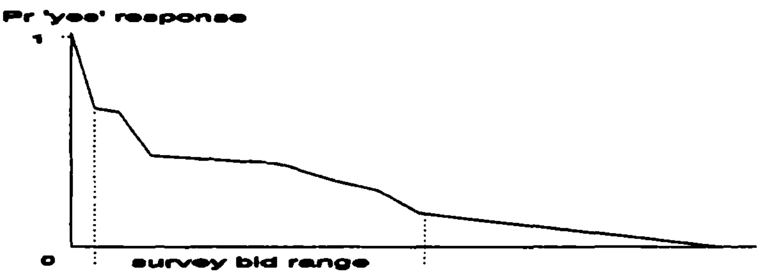
[35] The theory of benefit transfer will not be considered in detail, see Brookshire and Neill (1992), or Bergland *et al* (1995) for reviews of the theory and practice.

As with many of the DC techniques, non-parametric survival approaches to modelling censored data has manly been developed in the area of biological trials or mechanical reliability analysis. Kristrom (1990b) seems to have pioneered its application in CV, drawing on an algorithm developed by Ayer *et al* (1955), which makes for a deceptively simple estimation method. The advantage to the approach is that it appears to be only tenuously related to utility theory. Essentially Kristrom (1990a) points out that this link is "via a first order approximation argument, as the (yes/no) probabilities will depend on the size of the bid". In other words, the survival function has to show reasonable price sensitivity. The downside of the approach is that it makes somewhat arbitrary truncation assumptions at both ends of the resulting function. Also, it is considerably more complicated to incorporate covariates into non parametric models. Nevertheless the approach has been used in several studies, while convergent validity between parametric and non-parametric models has been shown by Kristrom (1990b). Carson *et al* (1994b) have used a generalisation of the Ayer method proposed by Turnball (1976), for interval-censored data from a Double bounded format.

Basically the method uses the sequence of proportions of 'yes' responses observed at each DC bid level, mapping out a function from the lowest bid to the highest. Unlike parametric estimation where predicted maximum likelihood probabilities are a function of the parameter vector of the assumed distribution, the non-parametric estimation models data uniquely as a function of bids. Compared to the parametric likelihood function above, the P_is are unknown parameters themselves, but their maximum likelihood estimates are simply the observed proportion of acceptances at bid level j, $\pi_j = yes_j/no_j$. If bids A_j are arrayed 1...J lowest to highest, then, Ayer *et al* show that if the sequence of proportions forms a monotonic non-increasing sequence of proportions, then this sequence provides a distribution free maximum likelihood estimator of the probability of acceptance (Kristrom 1990b).

In very large sample surveys, the proportions of 'yes' bids would be expected to decrease as bid increases. This may not be the case for small samples (see for example figure 4 chapter four). When the initial sequence is not monotonic (eg $\pi_j \leq \pi_{j+1}$) a simple transformation in the algorithm of Ayer *et al* adds the proportions of successive bids until the function is monotone non-increasing. The transformed sequence is mapped against the offer amounts to define a survival curve. To truncate the function (in cases where π_{j+1} (A) is not already = 0, the investigator must assume a bid value A at which the probability of a yes is assumed to be zero and extend the function by interpolation to that point. This is essentially an arbitrary interpolation over a range of response probabilities. Similarly, to close the function, the lowest bid amount (for which all yes responses are expected), can be at a response probability no higher than that for the minimum observed bid actually offered or where Pr yes = 1. Either approach essentially imposes a spike at zero and forecloses on the possibility of negative WTP.

Figure 5



Empirical survival function E(WTP) - area under fitted graph

Figure 5 shows a standard survival function and the essentially arbitrary interpolation over a range of (beyond bid range) response probabilities by fixing the end points of the graph. The mean is calculated as the area bounded by the survivor function while the median is the bid at which the probability of a 'yes' response is 0.5. This type of function is similar to the Kaplan-Meier survival routine offered by statistical programmes such as SPSS. A recent suggestion by Carson (1995 *personal Communication*) is that two mean calculations using non parametric step functions should bound the value of a correctly selected parametric model. In other words, at each step of the non parametric function which are the response proportions to discrete bid amounts, an expected mean can be calculated as the product of either the lower or upper value of each interval and the response proportion. There is as yet, no empirical proof that this method sets a bound on parametric models.

Semi-parametric estimation associated with Coslett (1983) and used in CV by Li (1996), offers an alternative to non-parametric estimation, by relaxing any parametric assumption about the distributional form of willingness to pay, but retaining the structure of the index function and therefore the potential to use covariates. Creel and Loomis (1994) introduce something similar, but their terminology appears to be confused. Their 'semi-nonparametric distribution free estimator' appears at first sight to dispense with a distributional assumption, although the method is essentially parametric in the estimation technique (Li 1995 *pers.comm.*). In essence their approach relies on an approximation to the error component using a Fourier series with very many terms[36], but which itself has an associated error term which appears to be logistically distributed. The importance of this last assumption diminishes as the number of terms in the series increases. Creel and Loomis p13 cite Gallant (1981, 1982) as showing that the Fourier series has the ability to globally approximate a function of unknown form arbitrarily accurately, as the sample size becomes large. In other words, as the number of terms increases to infinity the transformation will converge on the true distribution. An alternative approach not specified by Creel and Loomis, might be to use flexible non-linear transformations of the original variables while holding the assumption of the error term as logistic or normal. However these forms are somewhat complex to estimate and still outside the domain of CV practice. The main point, is that there is a spectrum of estimation procedures which basically generalise the shape of the bid curve, running from the fully parametric specification of both the random variable and the index function (common in most CV applications), through to Kristrom's fully non parametric estimator. In between are methods pioneered in other areas of discrete choice analysis relaxing assumptions on either or both parts (Li *op cit*, Manski 1985, Matzkin 1992).

3.16 CV and biodiversity: preference uncertainty and extreme responses

It is possible that some values held in relation to the natural world can motivate response patterns limiting the role of CV in resource allocation. At the limit, it is likely that such respondents may not self-select out of the survey process and give yes/no responses which in the absence of further probes will be registered as per normal. Such responses may not be suitable for inferring Hicksian surplus measures. Keeping with the maintained cost-benefit framework, from a policy perspective the issue becomes how to deal with such responses.

In the first instance the question is whether or not such problems can be identified and accommodated in the CV framework. This section focuses on a number of motives giving rise to problems in the DC framework. In the first instance, methods to incorporate uncertainty are reviewed. We then consider

[36] The trigonometric Fourier series is used extensively in engineering to approximate a periodic function (eg waves), by a linear function of sine and cosine terms (see Croft *et al* 1992).

alternative reasons for the apparent rejection of valuation, which range to the most extreme hypothesis of imprecise or non-existent preferences, or the possibility that preferences cannot be heuristically constructed in the typical CV framework. This discussion leads to a consideration of one of the most controversial aspects of CV, namely, the problem of scoping or embedding, which may be one manifestation of imprecise preferences. Irrespective of the existence value controversy, persistent embedding remains problematic for the credibility[37] of CV giving rise to potential allocative inefficiency. The relevance here is that the problem appears to be particularly acute when valuing complex goods such as biodiversity.

Although there are methods to test for insensitivity to scope, there is as yet little consensus on how the problem can be avoided. To the extent that some goods remain prone to the problem, then research on the domain of CV seems appropriate. Several issues are briefly addressed. First, the nature of the goods suitable for CV. Second, modification of CV practice, particularly to address information constraints. Third, the eventuality of not using CV in allocative decisions, and alternative criteria such as Safe Minimum Standards.

As an introductory rationale for subsequent chapters, analysis is based on the view that CV is an appropriate method for measuring total value of some resources, and should (with some qualification), be used in resource allocation. The discussion in this section is based on the observation of a particular response pattern to DC questions which may or may not obviously suggest the existence of a problem. Where analytical problems are evident, the priority involves rescuing as much information as possible from the method or deciding on its limits. A central critique which needs to be addressed, is the charge that CV is least useful where most needed, that is, for valuing goods of a non-use nature.

Preference uncertainty in CV

Faced with an unfamiliar scenario related to use or nonuse of unfamiliar goods, it is reasonable to suppose uncertainty on the part of a respondent. Reflecting this uncertainty in the DC framework[38] is essentially the question addressed in different ways by Svento (1993), Hanemann and Kristrom (1994), Li and Mattson (1995) and Hanemann *et al* (1995). All these approaches assume well behaved

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[37]Scope insensitivity is by no means limited to categories of goods deemed to provide existence value.

[38]A more general review of the literature on welfare measurement under uncertainty can be found in Svento (1994).

preferences, and propose somewhat elaborate fixes to parametric DC modelling procedure.

Representing the individual's preference ordering

$$u(x,z;\ p)$$

where x is a composite private good, z the index of environmental quality, and p an arbitrary parameter for random preferences; uncertainty with regard to either z or p would make it difficult for the individual to state his surplus value with certainty. As a modification to the difference in utility framework described above, Svento (1993) introduces the existence of indifference contingent on the bid value and the assumed quality z of alternative outcomes. In addition to a yes or no answer, the trichotomous choice framework allows respondents to indicate either that they would be equally well-off keeping the WTP amount proffered (and forgoing the change), as they would in paying the sum asked, or that they simply don't know (as opposed to yes or no)[39]. Modifying the likelihood function to account for a new response region (don't know or indifferent), Svento is able to show how explicit estimation of such responses can alter the welfare estimate compared to a cleaned data set, or simply systematically assigning don't know responses to the yes or no category. In practice, such responses do not produce radically different mean estimates and are likely to be few in many surveys. Where they do occur, Svento shows that it may be possible to model the magnitude of what he terms response vagueness.

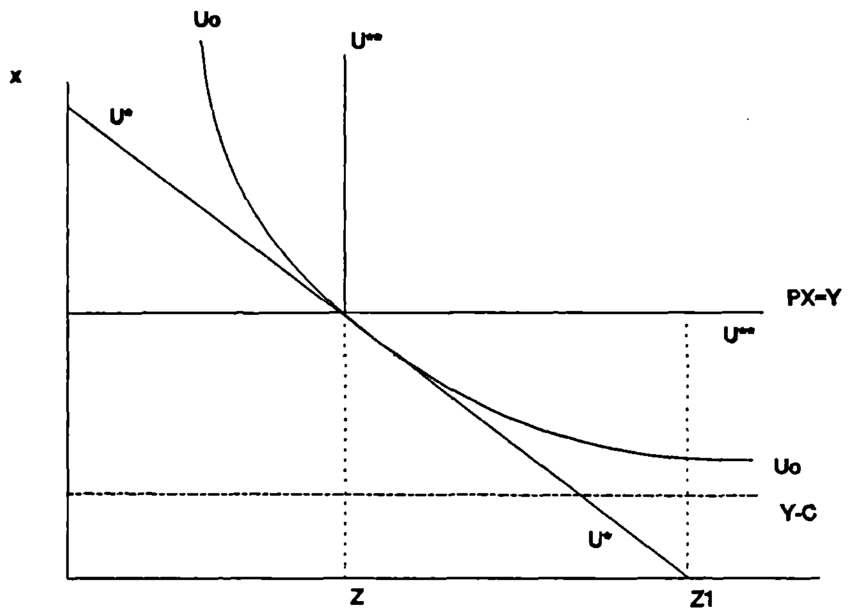
Hanemann and Kristrom (1995), prefer to speculate about the nature of p and why individuals have difficulty in stating a WTP. Two scenarios can be summed up in figure 6 which presents indifference maps showing perfect substitutability u* and perfect complementarity u**, and an intermediate case u^{o} and the budget line PX=Y. Assuming an increase in z to z1, the compensating surplus will be equivalent to the distance between the budget line and a particular indifference curve at z1. In this case the individual's CS could be anything between zero or his entire income y. It may therefore be difficult for the individual to state a WTP. Focusing on the intermediate convex case the location of the indifference curve relative to the budget line Y-C subject to paying the offer bid A for z1, determines the response. In this case the response would be a no.

There have been two attempts to capture the effects of preference variation represented by this model.

[39]Note that the offer of a don't know option was also proposed by the NOAA panel, such that explicit modelling procedures are of some relevance.

Hanemann *et al* (1995) neatly reparameterise the standard binary logistic response model with an alternative cdf of the random variable WTP conditional not only on the true WTP (represented by the bid amounts) but also an additional parameter they claim to represent the uncertainty around response values. Closer inspection reveals that this is akin to an error generalising model proposed by Kristrom (1990a)[40] or a semi parametric estimator simply generalising the distributional assumption and hence the shape of the response probability graph. Recovering this parameter is tantamount to explaining variability of the actual distribution and allows amendment of the expectation to account for uncertainty.

Figure 6 Preference uncertainty



However the approach might be of limited value since the authors admit that it is difficult to net-out any sampling error. Notwithstanding this problem, the same authors have applied the model to the original Bishop and Heberlein data, to confirm that the observed disparity between hypothetical and real WTA can be characterised by the uncertainty model, and can therefore be rationalised using the standard neoclassical apparatus, Li *et al* (1995).

[40]For the error component of the RUM Kristrom (1990) adopts the Aranda-Ordaz asymmetric generalization of the standard logistic cdf $F^n = 1 - [1+\lambda e^n]^{1/\lambda}$ (where $\lambda=0$ yields the extreme value cdf and $\lambda=1$ yields the logistic cdf) this and several other specifications allow WTP distributions to be flexibly generalised beyond two parameters.

The second attempt to approximate preference uncertainty is the use of a "post decisional confidence measure" in a survey to elicit this uncertainty measure directly. This basically boils down to asking people to state in percentage terms how certain they are about their DC response and then using the percentages to weight the likelihood function to be estimated. In the only application of this procedure, Li and Mattsson (1995) find that the weighting has a fairly dramatic effect on the mean estimate. The only criticism of this approach is that it is similar to the percentage split type of question for asking people to allocate WTP over value motives. Cognitively it is not clear that this task should be any easier than stating an open-ended WTP.

3.17 Alternative preference structures

The foregoing models attempt to safeguard the standard model framework. They do not probe too deeply into the underlying reasons for response patterns motivated by preference structures which if held by a significant number of respondents could simply invalidate the Hicks-Kaldor compensation criterion. This issue relates directly to the WTP/WTA disparity through the observation of a willingness to pay any amount of money or, if the opportunity is provided, stating a large willingness to accept value. Such responses have been rationalised by the existence of lexicographic preferences and/or the existence of moral responsibility in purchase and sale decisions.

Lexicographic or noncompensatory preferences

Georgescu-Roegen (1954) questioned the existence of indifference relations in the context of potentially irreducible biological functions, while Debreu (1954) formalised the lexicographic structure in economics. Noncompensatory or lexicographic preferences[41] in the context of CV have been discussed by Stevens *at al* (1991), Lockwood (1994) and Hanley and Spash (1995). In the latter two cases direct reference to valuing biodiversity was used. Both states can lead to the violation of the continuity axiom of the indifference curve, and therefore potentially invalidate WTP or WTA as useful Hicksian welfare measures.

Lexicographic preferences are thought to be a rational expression when a good is essential or ascribed moral (see below) or irreducible form such that potential satiation (and therefore a lapse into compensatory or exchange preference structures), is dominated by psychological or physical need characterised by a threshold. In the case of biodiversity, one might suppose such preferences to apply

[41] Strictly speaking noncompensatory preferences refer to a condition where indifference relations cannot be described. Lexicographic preferences apply to situations where a particular quantity of some attribute is preferred to any amount of another attribute. This may not preclude exchange preferences up to that threshold.

to the *existence* of charismatic species or biota which leads to the difficult part-whole bias problem of eliciting non-lexicographic behaviour for marginal changes. If this supposition is true [42], then the worth of CV is circumscribed by threshold levels which may vary for each and every individual.

On one side of the threshold (one which is characterised by a survival/extinction dichotomy), the individual may be WTP any amount to cross the threshold, or not WTA any amount of compensation to fall below it. As stated by Lockwood (1994), the only way out of this is to work on constructing preference maps to observe such preference discontinuities. In the absence of observable preference maps, the investigator must rely on crude consistency checks on possible lexicographic behaviour, such as those used by Stevens *et al* (1991). These basically ask respondents to agree or disagree with hypothetical trade-off scenarios involving money and environmental change. Such an approach can at best allow the identification of the proportion of a sample suspected of holding lexicographic preferences specific to a particular survey exercise. The framing of such validity checks can however be highly suggestive of motives the respondent may never have actually considered prior to the survey.

A further problem is presented by the need to aggregate individual lexicographic preferences. One decision response to a survey finding such preferences might reasonably be to equate non-compensation to the need for defined safe minimum standards.

Moral responsibility

The role of moral responsibility in provoking non-convexity in choice, is closely related to the issue of loss aversion and reference dependent preferences. Such behaviour is similar to the role of commitment acting as a barrier between personal choice and individual self-interested welfare, as advanced by Sen (1977). The issue here is less one of whether commitment is an allowable motive (e.g. is there any legitimacy in supposing committed behaviour as inconsistent with self-interested welfare?), but more a case of deciding factors in sale and purchase decisions giving rise to committed behaviour, and how to modify a CV accordingly.

Experimental evidence Boyce *et al* (1992) and Peterson *et al* (1995), has shown that some goods may indeed be imbued with a moral element provoking non-compensatory choices in decisions to accept compensation. The scenario can be characterised by the discontinuity in the indifference curve figure 7. Recall that the issue of loss aversion relates to the willingness to accept compensation granted that

[42]See Boyle *et al* (1994)

the respondent has a right to the higher level of provision of the public good Q1 shown by point c on U1. Considering the issue of a proposed reduction in provision to Q1 the Compensating surplus (WTA) may just be ab, but it is possible to imagine a discontinuity in the indifference curve U1 at point c (or any intermediate point to a) which accentuates the required compensation[43]

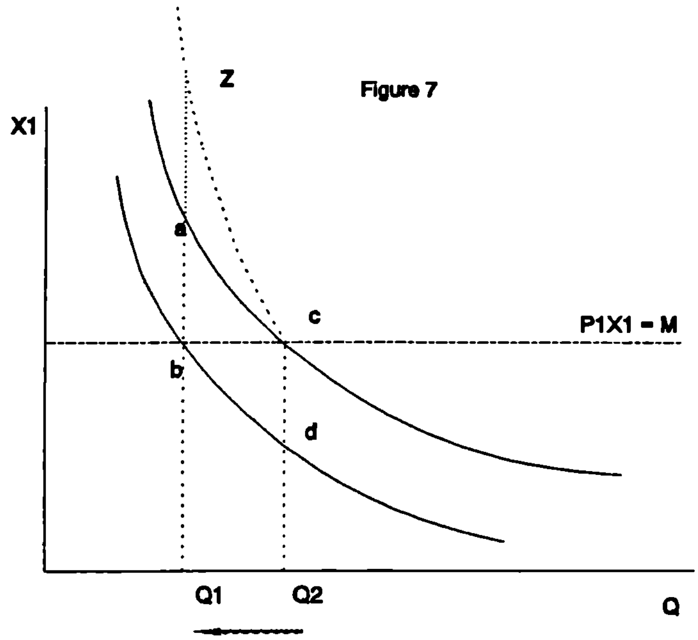
On can speculate that the extent of the discontinuity or the loss aversion is likely to be extreme for some goods such as endangered species, and where the respondent cannot fathom the correct extent of personal versus joint responsibility for the fate of the good in question. In terms of CV design, part of the problem may be avoided by adopting a WTP format which if accepted, affords the respondent a psychological detachment from personal responsibility for the outcome of his decision. On the other hand, if it is correct to use WTA then several issues become vital: Is the good likely to be imbued by a moral position on sale? Can it be correctly described such that the sale decision relates to a marginal decision, and not the fate say, of a whole species? By extension and for response purposes, can the individual respondent be convinced that she is in a position of joint as opposed to sole responsibility for the fate of the good? In their experiment on the WTA to avoid trees being destroyed, Boyce *et al* are not sure that they completely convey the species/individual tree dichotomy to respondents. This accentuates the moral position which they feel is adopted by respondents when taking sale decisions over certain environmental goods.

Citizen versus Consumer responses

In relation to the CV respondent's decision perspective, Blamey and Commmon (1993) evoke Sagoff's claim that there is a category mistake to claim that whatever the question format, individuals will in many circumstances oppose themselves by acting as moral agents and concerned citizens rather than self-interested consumers (Sagoff 1988). Environmental decisions taken in such a capacity are considered to fall within the provenance of "social regulation" requiring guidance through highly informed "ethical rationality" rather than poorly informed preferences. Blamey and Common take this one step further and contest the very existence of utility functions over 'pure public goods of the existence value type' (p.9) which they say, are a requirement for the use of CV in

CBA. The logical conclusion they say, is that CVM is only reliable as a surrogate referendum and

[43]At the limit it is possible to see that if the indifference curve becomes vertical then we have the lexicographic case.



Equivalent surplus (WTA) compensation for a quality reduction where respondents attach a moral responsibility to sale decisions.

not in CBA or damage assessment. This appears to be close to the views of Mitchell and Carson regarding the referendum model rather than the market model as the appropriate gauge of CV responses.

Blamey and Common appear not to distinguish the nature and content of utility over many public goods valued using CV, and the nature or existence of the utility function over existence *per se*. With regard to the latter they make the point that there is simply no objective method to validate any measure that might be claimed as motivated by pure existence. On the former there would appear to be some consensus in the literature that the content of the utility function is over some total value of the good in question, of which some inextricable part may be an existence component (Cummings and Harrisson 1995). Mitchell and Carson were referring only to the much debated issue of whether CV can give a pure existence value if one were (under ideal conditions), asking a group of off-site nonusers, about a remote good, and were probably not admitting that utility functions do not exist over public goods only that as such, CV is still useful since the measurement of total value at least avoids the problems of undervaluation even if existence value is impossible to estimate. As it happens, information on charitable giving may provide some insights to correct the apparent shortcoming identified by Blamey and Common. Exactly parallel research using CV and charitable giving has yet to be undertaken.

Blamey and Commons use of Sagoff's categorization takes the debate into familiar arguments over the of the sovereignty and content of individual preferences. In CV this argument has been well rehearsed since Kahneman and Knetsch's (1992) controversial explanation of embedding as the result of respondent's purchasing of moral satisfaction rather than responding to the extent of the suggested insult. To an extent this argument can be countered by the observation that nothing in economic theory limits the content of individual preferences. As Harrison (1992 p.250) memorably puts it - I call my utility jolly, what you choose to call your utility is, as far as I am concerned your business". Similarly Becker, cited by Hanemann (1994a pp33) states that '[I]ndividuals maximise welfare *as they conceive it*, whether they be selfish, altruistic, loyal, spiteful, or masochistic'. Much the same claim is made by Carson (1995a), citing Samuelson (1993), who reduces the 'warm glow' hypothesis to a failure to understand the very logic and history of consumer demand. However the problem of insensitivity to scope in contingent valuation remains, and it is appropriate to consider why embedding might arise in the context of valuing biodiversity. Prior to doing so it is of interest (from a technical point of view), to reconsider the evidence that Blamey and Common deploy to support their argument of citizen response motives.

Bid Insensitivity.

Blamey and Common re-analyze data from the South East forest CV study conducted by the Australian Resource Assessment Commission (RAC), which, along with the assessment of the mining operations at Coronation Hill (Kakadu), were significant events in the controversial life of CV in Australia (see Bennet and Carter 1994). Of particular interest in the forests case was the finding of bid insensitivity, which manifest itself in the insignificance of the bid variable in the logit equation and flat bid functions over the \$1-\$400 bid vector. Moreover this occurred in regression relationships over three different (randomly assigned) conservation scenarios for different amount of forest conservation, none of which turned out to be significantly different (with obvious implications for derived means or medians).

Leaving the obvious embedding problem (evident from similar bid functions), the slope of the functions provides an interesting diagnostic already discussed in the main body of this chapter. Interpretation is somewhat difficult. Although Blamey and Common use the somewhat heuristic device of including alternative dummy explanatory variables to explain responses[44] in accordance with citizen rather than consumer responses (they say are necessary to validate the use of CV), there is

[44]These questions had been included in the RAC design to identify committed voters who will essentially cause dump regardless of the price or the extent of the injury under consideration.

another possible hypothesis. First, the bid vector may not have been wide enough to provoke a sufficient number of negative responses at higher bid levels. This is somewhat worrying as the range appears to have been fairly wide. The only other explanations were that good was not sufficiently well described, or as suggested by Randall (1995 *pers. comm*), there is something inherently alien about the pseudo-referendum DC format for Australians. Dismissing the last reason as somewhat glib defence of the method per se, the suspicion of indiscriminate bidding or yea-saying for some remote and poorly-defined good cannot be dismissed. The question then becomes whether this may hold as a general rule, and goods can be valued using CV without extensive (and costly) description.

CV well-behaved preferences and embedding

This section offers some observations on the persistent problem of embedding in stated preference methods. The discussion leads to the question what respondents actually value in remote CV studies, or those concerning complex unfamiliar goods and by extension what can be expected in attempting to value biodiversity using the CVM?

Though there may some minor contextual difference between embedding (Kahneman and Knetsch 1992), part-whole bias (Mitchell and Carson 1989) mental account bias (Tversky and Khanemann 1981) and insensitivity to scope (Boyle et al 1993, Carson 1995b) they all refer to the same thing. Basically the respondent is valuing something other than that intended by the investigator. The Adding Up Problem (Hoehn and Randall 1989) is often mentioned in the same context, although this refers to something slightly different, namely the case where the sum of the individual values of say several individual species amounts to more than the value of a programme to save them all. Endangered species programmes are likely to be prone to the problem

Kahneman and Knetsch (1992) define embedding as occurring when "the same good is assigned a lower value if WTP for it is inferred from WTP for a more inclusive good than if the particular good is evaluated on its own". At its most extreme, for example in the South East Forests case, perfect embedding may lead to an equal stated value irrespective of the scale of the environmental change under consideration. The persistent occurrence of embedding or scope insensitivity over certain categories of goods has serious implications for the role of CV in resource allocation, either in cost-benefit analysis or for damage assessment.

There have been several explanations of embedding behaviour in CV studies. Of particular relevance to the current work is the observation by Schulze *et al* (1994) that mental models will hold many complex or 'exotic goods' to be considered as joint in nature. In other words in a CV exercise, it becomes difficult to persuade a respondent to value a species when he has a mental model of jointness between that and an entire ecosystem. Such behaviour is frequently conformed in retrospective reporting and verbal protocols, the value of which are to show the extent of divergence between the mental model the investigator is attempting to convey and that held by the respondent.

Much of the embedding debate has concentrated on showing that some aspects of studies demonstrating the phenomenon are flawed. This has been the approach taken by Smith (1992) in his critique of Khaneman and Knetsch. Likewise the otherwise flawless experiments of Desvousges *et al* (1992) which appeared to demonstrate clear embedding by respondents asked their WTP to prevent migratory waterfowl fatalities in oil ponds[45] and oil spills, highlighted among other things, the age of respondents and the alleged inferior nature interviews by a shopping mall intercept questioning format (Carson 1995b). A more theoretical basis exists to counter the criticism of Diamond and Hausman (1993) who find embedding inconsistent with the consumer choice axiom of non satiation. Hanemann (1994) evokes the degree of substitutability between goods and the familiar case of diminishing marginal utility to show that at least some scoping would be apparent if asking WTP for their joint provision as opposed to asking WTP for each one individually. The question begged by this explanation in the context of the experiments conducted by Desvousges *et al* regards the extent to which marginal utility is observed to decline in provision. As Arrow *et al* (1993) put it, the fact that WTP apparently flips to zero "is hard to explain as the expression of a consistent rational set of choices" (p.11).

A straightforward answer may be that people do not have well defined preferences over species which they know nothing about or may never of heard of. They may prefer them to exist rather than be extinct due to the irreversibility of the latter. However they may be indifferent between existence *per se* and the existence of any number over and above a notional critical threshold which may be scientifically-based or some purely personal construct. More specifically, 'lots more' of any specie may be associated with diminishing or insignificant changes in marginal utility.

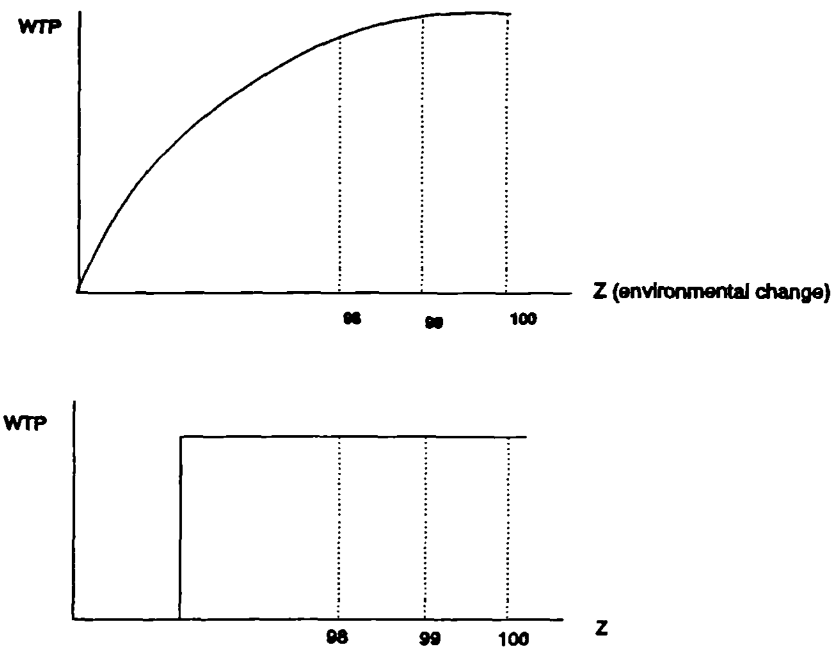
This is more or less the view advanced by Boyle *et al* (1994) and more cogently expressed by Fisher

[45] The Desvousges study found that WTP for a programme that saved 2% of the birds was not significantly different from WTP for a programme that saved 1% of the birds. The percentages refer to programmes to get the population back up to 100% of what it would have been in the absence of oil ponds.

(1994). WTP will essentially be invariant if respondents are asked to value numbers beyond the threshold because marginal utility has dropped off or declined sharply. Such a case can be shown in figure 8. Fig 8a shows WTP as a function of provision z which may be equated to the hypothetical waterfowl population recovery scenarios offered by Desvousges *et al.* Following the axioms of consumer theory for utility in z1, WTP is also concave in z1. If marginal utility is in fact positive, it is evident that the WTP for the 98-100% restoration programme should exceed the WTP for the improvement between 99-100%. Fig b demonstrates an alternative scenario in which WTP is zero until some population threshold is reached, then suddenly jumps to show the notional WTP for existence. Once existence is perceived assured, WTP becomes invariant for additional population increases. It is possible to speculate that such a scenario describes the findings of numerous studies including that of Desvousges *et al.* In other words, their bird population rescue scenarios were located in this flat preference surface such that the elicited WTP for the two alternative marginal scenarios above was the same.

This seems a plausible explanation for behaviour regarding endangered remote species and a finding which is not necessarily ruled out by economic theory (Carson 1995b). The crucial aspect in such circumstances, is how respondents interpret such thresholds (location etc). In the Desvousges *et al* study the 'population safety threshold' would appear to be located somewhere below the lowest population depletion scenario of 98% which respondents were invited to consider. If on the other hand, the threshold happened to be between 99 and 100% then WTP would be the same to go from 98-100 as 99-100 - apparent embedding behaviour. Two possible caveats should be borne in mind. First, it is possible that respondents are simply incapable of distinguishing the benefit derived from what amounts to two small recovery percentages. Second, if existence itself is scale and price invariant, then the WTP expressed for some part of an increase say, from 0-1% of the whole population, may well be the same as for the both the foregoing scenarios. In both cases however, the use of CV in valuing different programmes is somewhat limited. In the case where flat preference functions are not admissible (for increases in population), Boyle *et al* draw conclusions for the problem of valuing marginal changes. Specifically most changes in environmental assets requiring evaluation for both natural resource damage assessments and cost-benefit analysis consist of marginal effects that are small percentages of the total stock of the assets. If theory does not allow preferences to be flat over the posited changes in the quantity of the resource in question, then the only alternative explanation for observed scope insensitivity is a cognitive one. That is, the nature of the marginal change was not (or cannot be) communicated in an appreciable manner for most respondents.

Figure 8 Concave and flat preference areas and embedding



This underlines the role of accurate and detailed provision of information provision in the survey. In most species recovery programmes one would like to observe some concavity in the relationship between supply and WTP, even if thresholds are apparent. Methods which have been used to avoid embedding and scoping problems basically attempt to make the respondent aware of the irrationality of the act. Thus, describing both the larger entity and the subunit to be valued is one way to deal with the scope problem. Alternatively, asking about the value of the whole unit and then allowing respondents to allocate values to different parts of the resource is possible (Willis and Garrod 1993, Brown *et al* 1995). Other authors have suggested the use of disembedding questions to openly invite respondents to indicate th extent to which they embed (Schulze *et al* 1994). Similarly many standard approaches from psychology such as post-survey retrospective reports, verbal protocols and the development of complete context survey instruments (McClelland *et al* 1992) are rapidly becoming standard elements of survey design.

Whatever the cause, forms of embedding are routine and will increase with the unfamiliarity of the good. Carson (1995b) suggests 'remedies to the problem are straightforward in concept but often difficult and expensive to implement. The respondent must (i) clearly understand the characteristics of the good they are asked to value, (ii) find the CV scenario elements related to the good's provision plausible, and (iii) answer the CV question in a deliberate and meaningful manner". Even such advise is somewhat sweeping given the cognitive complexity of the issue.

3.18 Conclusion

This chapter has addressed numerous methodological problems to be confronted in using CV to value biological resources, which must be set against the main strength of the approach, the measurement of inclusion of non-use value and by extension the valuation method considered most appropriate for resource allocation decisions (involving complex goods) when more direct methods are unavailable.

Although the emotive and cognitive content of biodiversity presents a particularly stringent test for CV, a reliable modelling methodology is all important if the method is to stand up to scrutiny of its role in resource allocation and damage litigation. Some convergence in methodology would be desirable, although given that no two CV studies are attempting to value the same thing, this may be unrealistic. This chapter has outlined diverging features of different but plausible estimation procedures in DC CVM and attempted to reconcile the diversity of response motives with economic theory.

There are many potential pitfalls involved in modelling discrete choice data such as the choice of functional form, the error specification (or generalisation), integration limits, and bid vector design, that which need not be confronted in the use of the relatively more transparent but cognitively loaded open-ended format. For some authors (eg Desvousges *et al* 1992), the potential disparities that may emerge as a result of minor modification in modelling, are sufficient to cast doubt on the suitability of the DC approach for use in natural resource damage assessment. It is impossible to conclude in favour of either elicitation approach, although the merits of the open-ended approach should not be underestimated. By the same token the literature seems to suggest that many practitioners may have been seduced by the apparent rigour lent by the adoption of binary choice models and do not fully appreciate some of the associated problems. It is worthwhile recalling however that just as there is no theoretical distribution of WTP, there is essentially no definitive model of the strategies that people use to answer CV (DC or OE) questions. As such, it would seem less preferable to impose the Hanemann type format rather than to let people tell their own story. In other words, using current techniques, much more of what comes out of a DC model can be shown to be an artefact of that model. This chapter has described some of the methods to relax some of these modelling restrictions which are likely to grow in importance. For the time being the best recommendation is for surveys to be designed using the biggest open-ended pretest survey possible, which should ideally be split, not only to provide a bona fide open-ended sample, but also to investigate at least two DC bid vector designs. In this way convergent validity tests between formats are possible. A final point to make regarding the use of DC models in resource policy is that they are not user-friendly for those unfamiliar with statistical methods.

This chapter has eschewed further debate about the role of CV in measuring existence value. Suffice to say that in relation to the motives commonly ascribed to environmental valuation, the method accounts for existence value within a total value discussed by Cummings and Harrison (1995).

Motives matter, and actions such as charitable donations show that some non use element exist. But saying this is one thing while consistent measurement of somewhat arbitrary value categories for a range of goods is another. It may be possible to isolate an existence value residual from total value minus use. In the context of CV however, there are considerable problems in convincing respondents to temporarily suspend all use-related motive so that the investigator may identify the nonuse element. Even if this were possible the existence element is seemingly prone to a part-whole bias which, in the extreme invalidates the whole enterprise of CV. This is a controversial conclusion, but while it holds, the resource allocation implications of any use of CV predicated on the fiction of free-standing existence values is suspect. Quite simply the distinction between the use of CV to measure existence value as opposed to total value has not been clearly articulated, to the extent that rulings such as that made by the US court of appeal on non use value must be questioned (Cummings and Harrison 1994). This distinction notwithstanding, the true value of CV needs to be assessed for cases where existence values are thought to be significant and where they should be included in allocative decisions.

The conclusion held by the author is that CV can be used in many circumstances with certain caveats: First, the uniquely non-use case cannot be validated. Second scenarios involving complex or remote goods are difficult to describe and are likely to cause difficulties such as acute embedding behaviour. This may be circumvented by extensive focus group work but in reality some line should be drawn on the ability of respondents' to construct meaningful values for certain categories of goods. Again, it is impossible to state with any authority what these limits should be although some authors have attempted to categorise (see for example Wiestra *et al* 1995). Both caveats lead to somewhat negative conclusions regarding the role of CV. In the context of nonuse value, one can only concur with the conclusion of both Cummings and Harrison and Blamey and Common on the absence of any method for the objective validation of nonuse values. Combining this conclusion with the restriction on subject goods, inevitably leads to the somewhat negative conclusion that CV is least reliable where it is most needed. Indeed the least controversial uses of CV do still appear to be those related to goods in the use category such as water and sanitation provision although even here the merit nature of such goods is something approximating an existence value.

Several aspects covered in this chapter will be the subject of two CV applications in following chapters. It is however appropriate at this point to give some pointers on further potential research areas related to the above conclusions.

Other alternatives to CV do exist. Stated preference methods such as Contingent Ranking offer alternative ways of presenting multi-dimensional information sets, although the method can often limit choice and be statistically demanding (ERM 1996, Bergland 1994, Brown *et al* 1992). A more extreme option is to dispense with stated preference methods entirely, in favour of cost-effectiveness or safe minimum standard criteria. Both methods are more pragmatic policy tools, and in some sense switch the burden of proof onto the nature of the costs of conservation rather than the benefits. However, as stated in chapter one, much more research is required to determine exactly what constitutes a minimum standard for many species and ecosystems. Furthermore even an institutionalised SMS approach does not obviate some consideration of the role of existence values for the marginal case of costs being deemed excessive.

A further promising area of investigation in biodiversity valuation involves the development of experimental markets to circumvent the fundamental informational restrictions of typical CV surveys. Lab-based methods would take a subset respondents through an involved process of preference learning and value formation for highly unfamiliar aspects of ecosystems. The resulting values could be used in their own right or as suggested by Kask *et al* (1994), to calibrate[46] field CV values, a process they term CVM-X (see also Swallow 1994).

The CVM-X idea might also be used in conjunction with Values or Citizens' Juries which have a history of use in some European countries as a consensus approach to public policy on complex

[46]The calibration of CV responses is not new and was an aspect about which the NOAA panel solicited comments.

ethical and social issues (Stewart *et al* 1994)[47]. Basically the approach relies on the educated opinion of a carefully selected lay jury who are systematically briefed on all aspects of the issue at hand by a panel of unbiased experts. The process comes very close to the construction of a contingent market and has independently been suggested as an alternative to CV in the US by Brown *et al* (1995).

[47]In the UK the first consensus jury on plant biotechnology under the auspices of the Science Museum was held in November 1994 (see UK NCCPB 1994). In other countries such as Denmark and the Netherlands conferences have also been held on subjects such as food irradiation and transgenic animals.

Chapter 4

Valuing a Tropical Wetland Ecosystem: A Contingent Valuation Study

4.1 Introduction

Chapter 3 showed that a legitimate concern about CV is the feasibility of valuing complex goods such as ecosystems or species. An exhaustive pursuit of these issues requires some attention to areas such as cognition (definition of the policy issue and scenario comprehension), data analysis, public choice and other standard issues of theoretical validity. This chapter addressES these issues in the context of a fairly rigorous application of the method and standard procedures of response analysis. Specifically, what problems can be detected from the data and can these guide the limits of applicability? A relatively novel set of issues arises as a result of application in a tropical wetland ecosystem in a developing country, Brazil.

4.2 Issue

The Pantanal contingent valuation study was designed to investigate the value a sample of respondents would place on a healthy ecosystem. The study is similar in spirit to recent applications to other complex goods such as aquatic habitat (see Whittington *et al* 1994), ground water cleanup (McLelland *et al* 1992), valuation of landscapes (Willis *et al* 1995) and acid rain (Macmillan *et al* 1995).

The study area: Pantanal

Pantanal (Portuguese for plain) is a unique wetland environment shared between the Brazilian states of Mato Grosso and Mato Grosso do Sul (MS), with fringes in eastern Bolivia and Paraguay (figure 1). A vast food plain for the seasonal inundation of the Paraguay and tributary rivers, it forms the largest freshwater wetland in the world and is an unrivalled wildlife habitat of endemic and migratory species. The environment of the area is subject to several forms of extractive (subsistence) and non extractive uses (tourism), from which market-based valuation information may be derived. The passive damage nature of several current environmental threats to the area does however suggest a role for contingent valuation to explore aspects of total value not captured by revealed preference methods.

Centro de Pesquisa Agropecuaria do Pantanal (Pantanal Agricultural Research Centre a regional research arm of the Empresa Brasiliera de Pesquisa Agropecuaria - the national research network body of the ministry of Agriculture) identifies a number of threats to the Pantanal environment, three of which form the basis of a damage scenario presented to survey respondents. These comprise:

a) Mercury pollution from informal gold and mineral mining (known as the *garimpo);*
b) Agricultural run-off and sedimentation resulting from land use change in the adjacent highland plateau or *planalto* (fig 1);
c) Agrochemical residues.

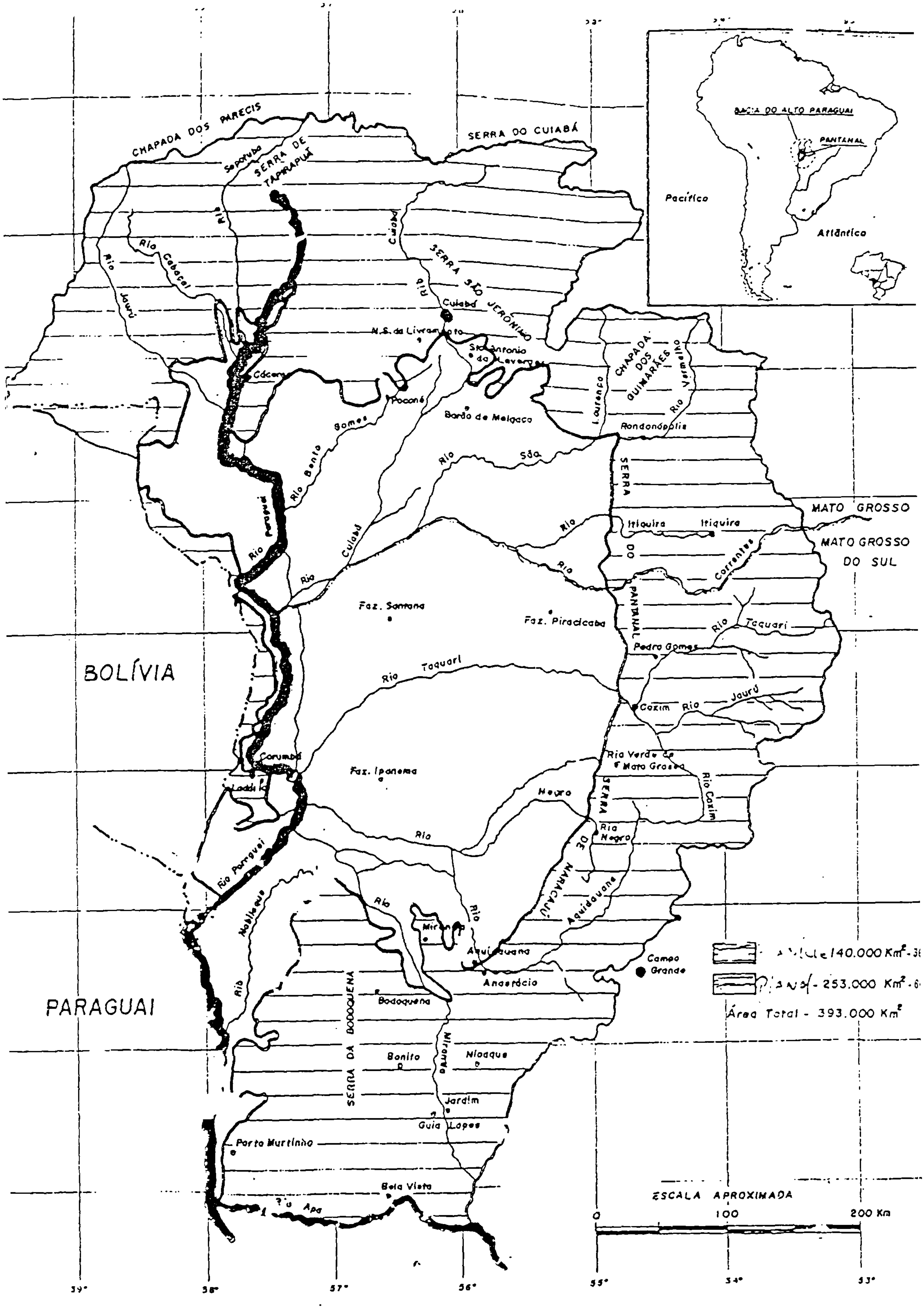
In addition, the area is threatened by a major engineering project (Hidrovia), which proposes to dredge a stretch of the Paraguay river between the towns of Corumba and Caceres for navigational purposes. The potential perturbation and long term damage has elevated the region to a global conservation *cause celebre* (see *New Scientist*, 3 June 1995; Sutton, 1995). The extent of the project, as well as the nature of resulting damage to the ecosystem, are subject to considerable uncertainty and existing impact assessments have not begun to account for the social cost of project alternatives.

The study reported in this chapter was initiated during a visit to Corumba (MS) in July and August 1994 and continued until November of the same year.

4.3 Why Contingent valuation?

Against this background of environmental change, the adoption of CV has much to do with the plurality of inseparable values associated with the resource, and motives held by users and non users of the Pantanal. Revealed value studies offer the only alternative validity check on hypothetical values, but the impacts of interest in the current study are difficult to capture using recreational demand models. In relation to CV, Cummings and Harrison (1995) highlight the impossibility of isolating (use) motive-free pure non use measures and suggest that total value is the correct construct for interpreting CV responses. If this is the case the site/off-site distinction is less relevant in designing CV studies, although users are most likely to give the best reflection of total value of the resource. Either way, direct inquiry provides the only means of identifying a total attribute-related surplus measure.

Aside from providing a more accurate reflection of resource value, the main advantage of using CV is the flexibility of application. In fact, the only limit of application relates to the ability to link reliably the monetary expression of use motives to the quality change of interest. The bounds of the latter are very much in the gift of the survey designer, who has the responsibility to communicate the issue in such as way to ensure respondents arrive at a value statement using relevant information rather than default/heuristic assumptions. This problem is not trivial, and few of the protagonists in the debate seem comfortable to delimit the boundaries of the method accordingly. An obvious question



CHAPADA DOS PARECIS
SERRA DE TAPIRAPUÃ
SERRA DO CUIABÁ
BACIA DO ALTO PARAGUAI
PANTANAL
Pacífico
Atlântico
Rio Sepotuba
Rio Cabaçal
Rio Jaurú
Rio Cuiabá
SERRA SÃO JERÔNIMO
Cuiabá
N.S. do Livramento
Sto. Antonio do Leverger
CHAPADA DOS GUIMARÃES
Rio Vermelho
Lourenço
Cáceres
Poconé
Barão de Melgaço
Rondonópolis
Rio Bento Gomes
Rio São Lourenço
SERRA DO PANTANAL
Rio Paraguai
MATO GROSSO
MATO GROSSO DO SUL
Itiquira
Rio Itiquira
Rio Correntes
Rio Piquiri
Faz. Santana
Faz. Piracicaba
Rio Taquari
Pedro Gomes
BOLÍVIA
Coxim
Rio Jaurú
Corumbá
Faz. Ipanema
Rio Verde de Mato Grosso
Rio Coxim
Rio Negro
SERRA DE MARACAJÚ
Rio Aquidauana
Rio Nabileque
Rio Miranda
Miranda
Aquidauana
Anastácio
Campo Grande
140.000 Km²
253.000 Km²
Área Total - 393.000 Km²
PARAGUAI
SERRA DA BODOQUENA
Bodoquena
Bonito
Nioaque
Jardim
Guia Lopes
Porto Murtinho
Bela Vista
Rio Apa
ESCALA APROXIMADA
0
100
200 Km
59°
58°
57°
56°
55°
54°
53°

to ask of the current study therefore, relates to the validity of responses and the absence of any objective criteria against which the derived information can be gauged. That contingent valuation has apparently been 'successfully' applied extensively in similar circumstances is insufficient rationale for unqualified adoption. In the absence of further cognitive validation, such a statement is nevertheless a commonly observed conclusion.

4.4 Survey design

The survey instrument for this study can be found in appendix 1 and 1A. The associated show cards used with the survey are included in appendix 2. As in many other studies, the NOAA guidelines (summarised in appendix 3) have (rightly or wrongly) been adopted as a default point of reference for several of the design features:

- Face-to-face administration of the survey;
- Discrete choice (although not as a referendum);
- Willingness to pay (equivalent surplus);
- Allowing a don't know option;
- Reminding respondents of the availability of substitutes.

However the distinction between the CV requirements for damage litigation - the concern of NOAA - as opposed to benefit-cost analysis remains unclear, and the need to adhere rigidly to all points listed in appendix 2 is questionable.

For cost reasons, mail surveys are the preferred method in the majority of published studies. Following design criteria such as those suggested by Dillman (1978), respectable response rates can be achieved, while the method allows the investigator to target the sample more accurately. A disadvantage of the method is that bias may emerge from essentially arbitrary treatment of non-respondents, while the identity of the actual respondent is never certain. In practice face-to-face surveys circumvent many of the drawbacks encountered using the mail format. The approach does introduce the problems of interviewer and social desirability bias.

The preliminary version of the questionnaire consisted of 30 questions accompanied by eight show cards presenting multiple choice answers and providing supplementary information to a verbal scenario. Questionnaire versions were designed for administration to a sample frame of tourist/visitors passing through the towns of Corumba and Miranda[1] (figure 1). The majority of these visitors have engaged in recreational angling during their visit to the area. Four trained interviewers were employed in Corumba and a further 2 in Miranda over a period of 4 months between August and November 1994. Interviews of approximately 20-25 minutes were on the basis of casual intercept at ranger stations through which visitors must pass to weigh catches and purchase an official permit/laisser passer called a *lacre*.

Questions 1 through 19 of the questionnaire mainly elicit reasons for a respondent's visit and travel cost information. Several variables such as fish catch, repeat or first visit, group/family size and days spent in the area may also serve as validatory covariates in bid functions. Questions 20-30 and related information comprise the contingent valuation exercise, and collect further information on respondents' socio-economic characteristics.

Scenario

The damage scenario was perhaps the least satisfactory element of the design. The somewhat contradictory position created by complex goods relates to the NOAA requirement of checking respondent understanding (NOAA 1993). The CV representation of a complex good is in fact an artefact of the information the investigator considers important. A review of much of the literature shows scenario communication to be a particularly weak area in reducing complex and emotive issues to the dimensions of heuristic devices such as photographs, pie charts and quality ladders. Use of such methods in risk communication has been shown to bias resulting welfare measures (Loomis and duVair 1993), and even extensive pretests cannot guarantee cognitive validity. The two main issues are (a) whether the respondents' interpretation of a scenario corresponds to that intended by the researcher, and (b) whether or not the scenario actually corresponds to the eventual outcome of a particular project.

Given the three principal causes of damage, the first problem is what to ask about. Photographic evidence was not considered sufficiently subtle to convey an idea of ecosystem damage. The preferred scenario informed respondents of the damaging effects of a decline in water quality in the Pantanal, and its direct influence on plant and animal diversity. Damage was attributed to factors a) - c) listed above, with additional effects related to sporadic deforestation and 'civil constructions' such as roads, minor dams and dykes.

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[1]Situated approximately 200km apart. These towns are considered to receive different visitor groups.

Some of the complexity of conveying damages to respondents was avoided using a species box showcard (suggested by Macmillan *et al* 1996). The scenario (see card 5 appendix 2) suggested 3 ecosystem damage stages based on current patterns of resource use in the Pantanal and placed respondents before an impending reduction from stage B to stage C over the period from 'today' to the year 2010. A further map showcard informed respondents that the extent of damage might be locally severe in specific areas. The highlighted areas corresponded to zones of known environmental damage as a result of open-cast gold and mineral mining and was designed with a view to test for differences in on-site and off-site perceptions given that respondents were previously asked whether or not they had actually visited any of the areas shown. Scenario presentation was interrupted by a single question probing respondents' prior knowledge about pollution in the Pantanal. The question was mainly for validatory purposes but equally designed to break up the monotony of a fairly lengthy scenario communication process. The scenario continued to inform respondents of the costly nature of new and additional control technologies through charges over and above current payments made by all taxpaying Brazilians as well as through current use charges for individual anglers, which are paid in two ways. These contributions notwithstanding, controlling water pollution in certain areas like the Pantanal was described to require additional expenditure. One important consideration in the determination of fund allocation was the value individuals placed on the characteristics of the Pantanal environment and the leisure opportunities it afforded. Hence the reason to elicit individual willingness to pay for the maintenance of current water quality conditions.

As a prelude to an initial filter question on the willingness to pay (anything), several important qualifying statements were read out to respondents:

a) To reassure regarding the exact and sole purpose of any payment;

b) To inform respondents about precisely what they were paying for;

c) To inform respondents of the likely outcome if sufficient funds were not forthcoming;

d) To remind respondents to consider their own current and future uses;

e) To remind respondents to consider their budget constraint;

f) That all users would pay, and that the decision to pay should be taken in the knowledge that such a payment related only to environmental quality in Pantanal (Mato Grosso do Sul). In other words, reminding informants of substitutes, (in this case the potential use in Mato Grosso or alternative areas nearer points of origin).

Choice of Payment Vehicle

The choice of an appropriate payment vehicle has been found to have a significant affect on WTP, Bateman *et al* (1995). The Brazilian experience with CV has been limited to one study on rural water (Briscoe *et al* 1990), which used an hypothetical monthly tariff for connection. Forms of payment for less familiar goods are controversial everywhere, although initial doubts that such scepticism would add to considerable suspicion of authority and distrust of government were largely unwarranted. A tax option offers considerable advantages in being an egalitarian means of payment. Casual probing during pretesting was sufficient to suggest the potential rejection of anything apparently controlled by a remote authority in Brazilia.

Alternative use-related vehicles include an annual licence payable by all recreational anglers[2] at a cost of R$34, or the previously mentioned *lacre* (seal). Seals are fixed on catch boxes at a cost of $R4 per box[3] (volume). Neither mechanism is wholly ideal as a payment vehicle. The liability of the lacre is clearly dependent on catch volumes and some respondents may have ruled out future liabilities on this basis. On the other hand, there is a definite sense in which a catch-related payment offers respondents a closer approximation to a payment for the environment, and essentially circumvents scepticism over a wasted payment. Alternatively the higher priced licence translates more clearly into an annual payment and may be more appropriate. However the obligation to pay anything extra is conditional on the decision to renew. This uncertainty dictated a split-sample approach as follows:

Pretest 1: Open-ended WTP an amount additional to the cost of a lacre, with WTP amounts suggested by payment card (see card 4 appendix 2). The payment card was favoured because of uncertainty about respondents' ability to state open WTP amounts. Values were arbitrarily selected over the range R$0 - R$4000.

Pretest 2: Open-ended WTP an amount additional to the cost of a licence. WTP amounts freely stated (no payment card).

Willingness to pay questions

The exact wording of scenarios and the open-ended and discrete-choice willingness to pay questions may be found in appendix 1. A novel aspect is the question sequence summarised in table 1, which includes a 'don't know' option answer at three stages of the discrete choice format. This enlarges the analytical permutations considerably (although analysis of these options will not be pursued here).

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[2]It should be noted that the definition of recreational angling differs considerably from its UK counterpart, with a mean catch per visit of 128kg of fish for the two pretest samples.

[3]Over the period of the survey the Brazilian Real stood at approximately $R1 = $US0.9.

The willingness to pay question elicited a WTP corresponding to the equivalent surplus of a potential quality decline. The format makes an implicit assumption regarding the property right, which was not contested by respondents.

Sample Frame and Administration

Determination of a sample frame from an affected population presented a problem for the current study (and potentially a problem for many non use CV studies in developing countries). A sample frame bias arises where the sample frame "does not give every member of the population chosen a known and positive probability of being included in the sample" (Mitchell and Carson 1989). The problem jeopardises the accuracy of projecting the results of this survey to any population beyond the socioeconomic group comprising the majority of respondents who themselves will be a subset of potential interviewees.

While the current study is dealing with a unique and world renowned resource (ie. with a high non use component which potentially widens the population to the whole world), the potentially effected population was in the first instance determined to be direct users of the resource[4]. In other CV studies this is typically a homogeneous group to the extent that an unmodified instrument is considered appropriate for all prospective respondents.

In the current study the population of users was determined to span a range of socioeconomic classes and income groups which could not reasonably be bridged by a single survey instrument similar to the one used in this study. In particular, it was felt that an unmodified survey asking poorer groups about water quality in relation to biological diversity would probably result in a high protest rate. On the other hand, use of multiple survey versions raised the prospect of having to adopt specific communication devices and payment vehicles, with no guarantee that the subject good is perceived equally across versions.

The issue of how to sample from a highly heterogeneous user population therefore presents a problem for CV surveys. A single survey instrument rules out probability sampling (as advised by NOAA), the objective of which is to obtain a sample similar to the parent population. We deliberately restrict the population to the subset of anglers, and the resulting selection bias is accounted for when

[4]Users are distinct from nonusers who do not actually use the resource, but may hold non use value as part of a total value expression in a CV response (see Cummings and Harrison 1995).

Table 1. Question sequence open-ended and discrete choice questionnaires

	Pretest 1	Pretest 1A	Discrete choice
Q.21	WTP (any amount)? yes/no/don't know	WTP (any amount)? yes/no/don't know	WTP (any amount)? yes/no/don't know
Q.22	maximum WTP as increase in price of *lacre* (payment card)?	maximum WTP as increase in price of *licence* (without payment card)?	WTP amount $RX over price of lacre? yes/no/don't know
Q.23	Willing to pay any more? yes/no/don't know	Willing to pay any more? yes/no/don't know	WTP amount 0.5* $RX or 2*$RX (from Q.22) over price of lacre? yes/no/don't know
Q.24	Maximum amount additional to amount stated Q.22	Maximum amount additional to amount stated Q.22	Maximum WTP over current price of lacre?
Q.25 & 26	Verbal validation of WTP and/or reasons for zero	Verbal validation of WTP and/or reasons for zero	Verbal validation of WTP and/or reasons for zero

Table 2. Summary of responses

	Open-ended	Discrete-choice
Total(1)	186	400
refusals	-6	-10
Non response WTP	-9	-13
of which 'Don't know	-	6(3)
Other item non-response	-31	-88
protest	-7	-28
Total(2)	133	267

Notes: 1. Total surveys actually administered (OE) or the number designed (DC) of which some may have been spoilt; 2. Total analyzed. 3 Don't knows are treated as no responses, therefore the deduction for the non response WTP is only 7.

aggregating over the relevant population.

A series of mock interviews cum focus group exercises conducted with CPAP employees allowed several potential design problems to be identified. Pretesting took place in July 1994, with surveying conducted between August and November. Pretesting enabled several other issues to be investigated, namely the choice of elicitation format, payment vehicles, and the determination of the bid vector to be used in subsequent design of the discrete choice survey

A total sample of 586 visitors of a total recorded annual population of 110,000[5] were interviewed at wildlife ranger stations. Willingness to participate was ascertained by casual intercept (i.e. random requests to participate), a method has been employed in conjunction with both interview and self-administered CV studies (see Boyle *et al* 1994). Despite the need to maximise interview numbers (and a low visitor throughput at selected sites), interviewers were instructed only to interview group members simultaneously, and to approach male and female visitors equally.

4.5 Obtaining a clean data set

Issues typically dealt with in cleaning the data set are the treatment of protests/zeros, cross tabulation of responses for consistency and plausibility, and a decision regarding other missing observations. In addition, some decision may be taken on the identification of outlying responses, although this is dealt with in the following section. For many CV studies these procedures have been erratic at best[6].

For both the open and closed-ended versions of the survey considerable importance must be attached to the treatment of zero and non respondents. To an extent this task is complicated using face-to-face surveys, which on the one hand give the interviewer the chance to probe respondent motives, but at the same time increase the likelihood of interviewer and social-desirability bias. With mail surveys such problems can be by-passed by making an arbitrary decision about non-respondents when the questionnaire is not returned. That is, to treat them as zeros (the conservative option), or to exclude from the sample. For returned protests and zero bids, both face to face and mail responses may be validated with a follow-up question such as question 26 in the current survey.

[5]This figure relates only to the state of Mato Grosso do Sul and is based solely on the issue of lacre by the policia florestal. It probably underestimates the number of actual anglers. At least an equal number of visitors might be assumed for Mato Grosso state.

[6] See for example the literature review Appendix A in Desvousges *et al* (1992).

Table 2 summarises response categories. Where a choice has been necessary, the tendency is always to err on the conservative side. Not shown are genuine zero bids deduced from a standard validatory question. Don't knows are treated as zeros and included in the final analysis. Detailed cross tabulation was considered unnecessary as standard database[7] queries revealed no disproportionate WTP responses relative to income or age. There are several methods to impute missing values for independent variables and for predicting the missing dependent variable or simulating the response probabilities of different sample income groups (see Whitehead 1994 and Whittington *et al* 1994 respectively). Respondents with missing observations on any of a limited number of independent variables were simply omitted. Although the literature suggest this could give rise to a sample selection problem if the dropped observations are a non-random sub-sample, most missing observations were not noted to be from the sensitive variables like income. As such we do not find the problem of self-selection by lower income groups.

For the purpose of discrete choice analysis it is possible to check the sensitivity of E(WTP) by arbitrarily including a number of the item non-respondents (failure to respond to specific qualifying questions), don't knows and WTP non respondents as 'no' responses in the data set. This process augments the data set by a further 97 observations (for a total of 364 respondents to the single-bounded question), and will be termed the 'conservative' data set in any further analysis.

Pretest data - analysis validity

Data analysis is described in two stages[8] corresponding to the analysis of two open ended pretest data sets, and the use of all (or a subset) of these responses in the design of an optimal discrete choice format.

Initially two pretest versions (total of 186 surveys) were administered. Q26 was used to identify genuine zero values and protest responses. Table 3 summarises some of the important characteristics of the sample as well as showing the pretest means which are shown in more detail in table 4. Noteworthy in table 3 is the stated mean monthly income (all sources), which is approximately R$4,800 and clearly above the national average. The sample is equally skewed in terms of educational attainment and sex.

[7]Codification and cross-referencing using the database programme Borland-Paradox.

[8]All data and programmes detailed in subsequent analysis can be obtained from the author.

Table 3 Pre-test and general sample characteristics

	mean	median	spread	std. dev.	N
WTP[1]	52.76	30	0 - 300	57.75	78
WTP[2]	89.745	70	4 - 300	70.43	55
Income[3]	4394.2	4250	1000 -10,000	2499.5	482
Sex	0.987	-	0 - 1	0.109	495
Age	43.1	43	21 - 76	8.63	467
Education[4]	5.45	6	3 - 7	0.86	476
Days spent[5]	6.47	6	3- 15	1.47	338

Notes: 1. open-ended pretest sample 1; 2. open-ended pretest sample 2; 3. total monthly income from all sources in \$R (US\$1 = \$R0.9), 3. Ranging 1 (lowest) - 6 (highest) see Appendix; 5. total days normally spent during a visit to the Pantanal

Table 4 Comparing pre-test samples

	Mean A (n=78)	Mean B (n=55)	t-test	Mann-Whitney
WTP[1]	52.76	89.74	-3.21	R (p = 0.0000)
WTP[2]	33.64	59.76	-3.77	R (p = 0.0000)
Income	4166.6	4181.8	-.04	NR (p = 0.5057)
Age	43.29	44.87	-1.02	NR (p = 0.1866)

Note: Sample A (lacre) sample B (licence)

t-test computed without assuming equal variances. NR cannot reject null of equal means at p = 0.05. WTP[1] is the sum of WTP Q.(22) and follow-up amount Q.(24), WTP[2] is the WTP Q(22) only

Table 4 makes clear that sample means (for the two pretest samples) for income and age are not significantly different. However the mean WTP amounts differ[9], with higher amounts paradoxically elicited from the higher cost licence than from the lower cost lacre. There are several possible explanations for this observation. In the first place, respondents to the lacre question actually considered the implications a higher price lacre and the uncertainty of multiple payments related to fish catch. On the other hand, respondents asked to consider the licence vehicle would probably have been aware of the implied annual payment. Based on pretest information, several questions need to be addressed. First, is it possible to validate theoretically the pretest information? Second, which vehicle is most suited for use in subsequent discrete choice questionnaires (in terms of avoiding unnecessary bias)? Third, what information can be obtained on the underlying distribution of willingness to pay for the design of a discrete choice bid vector?

4.6 Bid functions

The theoretical validity of CV responses can in part be gauged using bid functions both to check parameter signs conform to a priori expectations, and as a device to extrapolate characteristics of sample behaviour. Bid functions are usually *ad hoc* constructions since they cannot be integrated into some known utility function. Despite this, some authors (eg see Bateman *et al* 1993) stress the theoretical undesirability of common linear forms due to the implied restrictions on marginal willingness to pay and the need to avoid the prediction of negative WTP values[10]. It is possible to exploit more acceptable forms, e.g. log forms (see Garrod and Willis 1995) or flexible forms and transformations such as the Box-Cox (Soguel 1994). Most open-ended studies have shown low explanatory power over several forms and the purpose here is not to extrapolate from the open-ended functional form. Arguably a level of confidence should be expressed in the data contingent on the confirmation of a priori expectations. However open-ended functions do not directly determine the magnitude of the welfare measure, and while they may serve as a useful cross-check on subsequent DC analysis, they will not receive exhaustive treatment here.

Table 5 summarises the subsequent regression information.

Regression 1: employs a WTP (LACRE) dependent variable regressed on five explanatory variables.

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[9]Considering both the amounts stated in the initial WTP Q.22 and the sum of Q.22 and follow up Q.24.

[10] By using log WTP as the dependent variable.

Table 5. Bid functions (open-ended responses)

Dependent Variable	Constant	Income	Catch	Age	1st Visit (1-0)	Know. Poll (1-0)	Vehicle (1-0)	
WTP (LACRE) n = 78,	-9.87 (-0.46) 0.64	0.00017 (0.103) 0.92	0.043 (1.596) 0.11	0.79 (1.534) 0.12	-7.75 (-1.008) 0.31	11.26 (1.204) 0.19		Adj R^2 = 0.03, F[5,72] = 1.5, BP* (5) = 28.5
WTP (LICENCE) n = 55	17.75 (0.472) 0.63	0.0069 (2.51) 0.01	0.161 (2.13) 0.03	-0.163 (-0.25) 0.79	11.83 (1.187) 0.23	-1.52 (-0.189) 0.84		Adj R^2 = 0.11, F[5,49] = 2.46, BP (5) = 8.3
TOTWTP (LACRE) n = 78,	-25.1 (-0.76) 0.44	-0.001 (-0.43) 0.66	0.054 (1.61) 0.10	1.56 (2.02) 0.043	-10.33 (-0.91) 0.38	25.58 (2.06) 0.48		Adj R^2 = 0.09, F[5,72] = 2.6, BP (5) = 21.6
TOTWTP (LACRE) n = 68,	-12.46 (-0.60) 0.54	-0.001 (-0.78) 0.43	0.067 (1.74) 0.08	0.99 (1.88) 0.60	0.99 (0.16) 0.87	-1.30 (-0.22) 0.82		Adj R^2 = 0.10, F[5,62] = 2.5, BP (5) = 21.0
TOTWTP (LACRE&LICENCE) n = 133,	0.57 (0.03) 0.97	0.0022 (1.43) 0.15	0.062 (2.36) 0.01	0.26 (0.64) 0.51	-0.93 (-0.14) 0.88	7.60 -1.15 0.24	28.03 (4.23) 0.00	Adj R^2 = 0.11, F[6,126] = 3.9, BP (6) = 29.6

Notes: t-stats. in parentheses, significance level below
*BP is the Breusch-Pagan tests for heteroscedasticity and can be compared with a chi-squared distribution with the number of degrees of freedom dictated by the sample size and number of variables. The t-values have been computed using White's heteroscedasticity covariance matrix estimator (see Green 1993 pp391)

A coefficient is said to be significant if the significance level is below 0.05; the t-test for each coefficient refers to a two-sided test. Accordingly the model performs poorly with a low adjusted R^2 of 3% and a F test of model significance = 1.5 (p = 0.18). Furthermore the Breusch-Pagan statistic suggests the presence of heteroscedasticity, which along with a low fit statistic are characteristics of cross-sectional data sets. In the presence of heteroscedasticity parameter estimates are not minimum variance, effecting the reliability of t-statistics and the conventional F-test. As there is no obvious re-specification of the equation to eliminate the problem, the use of White's consistent estimator for revising the covariance matrix allows the result of OLS to be salvaged without knowing the form of heteroscedasticity (see Greene 1993 pp391).

Regression 2 repeats the above exercise regressing WTP elicited with the licence payment vehicle on the same covariates. For a smaller sample size there is a notable improvement in the significance of the income and catch variables, as well as the overall fit. Additionally the Breusch Pagan statistic indicates that the null of homoscedastic errors cannot be rejected (compared to a critical value of 11.07 for the chi-squared distribution with 5 d.f.), suggesting improved model specification. Interestingly the signs on the coefficients for age and 1st visit have reversed, although a negative sign on age is in keeping with other CV studies. The sign on respondent knowledge of the pollution problem is unexpected but will not detain us further.

Regression 3 uses a total WTP (lacre) dependent variable, the sum of the first WTP question and a follow-up requesting an additional payment above the initial amount (Q22+Q22).
In theory, the two questions should elicit a more accurate measure of surplus although the results presented here are not encouraging. Specifically, only age shows any significance, although the sign reversal may indicate that the follow up question may not in fact have elicited a systematic reappraisal of WTP.

Regression 4: To further investigate the findings of the previous regression an attempt was made to clean the data set by considering the presence of potential outlying observations. The interpretation of problems caused by the presence outlying WTP observations were briefly mentioned in chapter three. Methods suggested in the literature are essentially arbitrary variants of resistant fit models or windsorised[11] data sets and the current analysis employed a statistically convenient method to handle outliers. A distribution-free boxplot (See Emerson and Strenio 1983) of the WTP data used in regression 3, was used to throw light on the structure of the data. The box plot (figure 2) conveys

[11]Based on the simplistic principle that all data sets are normal in the middle.

visually and simply information about the distribution of the data and provides clues relevant to the distribution of the bid vector for the design of the discrete choice format. While distribution free, a boxplot shows features of location and width analogous to a normal distribution. Outliers and a standard confidence interval are defined by the data rather than an assumed distribution, and the plot requires the median and fourths of the distribution which are resistant to up to 25% of the data being arbitrarily large.

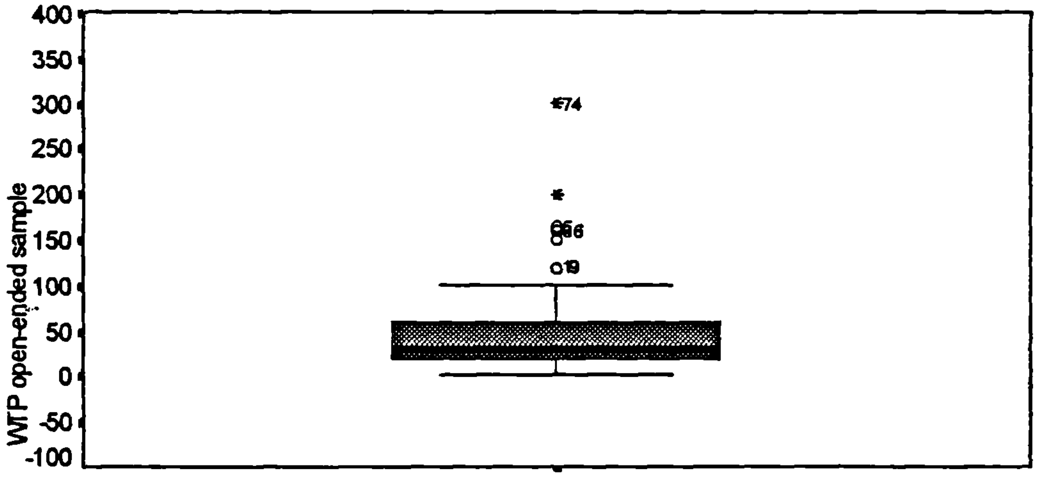
In the plot, the horizontal straight line through the shaded box shows the median of the open-ended data set lying towards the bottom of the box, indicating that the data are positively skewed (confirming the observed disparity between the mean and median in table 3. The interquartile range (ie the 25th to the 75th percentile or 50% of observations) defines the upper and lower boundaries of the box. Outlying willingness to pay observations are defined by the "whiskers" spanning three halves the interquartile range (small circles) or extreme values more than 3 box lengths from the 75th percentile (denoted by asterisks). For the open-ended data set (lacre and payment card), the procedure identified approximately 10 observations[12] to be removed from the data set prior to re-estimation. It is worth recalling that the removal of these observations is simply for exploratory purposes rather than for mean calculation. As with the mean/median argument (welfare measure), the removal of genuine high bids risks the loss of respondents with genuinely most to lose from the environmental change in question, and is therefore to be discouraged unless convincingly validated by reference to respondent income or age.

Regression 4: In the event, the removal of the outliers brings only a small improvement to the estimated model which while significant ($p = 0.034$) is somewhat counter-intuitive for the income and prior (pollution) knowledge variables.

Regression 5: As a final cross check to the information presented in table 4 both sets of pretest (total) WTP values were stacked and re-estimated with a dummy payment vehicle variable (licence = 1, lacre = 0). As expected, the freely stated WTP additional to the licence has a significant positive effect on willingness to pay consistent with the difference in means shown in table 4.

[12] The 6 points indicated on fig 2. (eight outliers and two extreme observations) cover multiple observations at $R200 (3 observations) and $R150 (3 observations). Numbers indicated in figure 1 are data set reference numbers and not amounts.

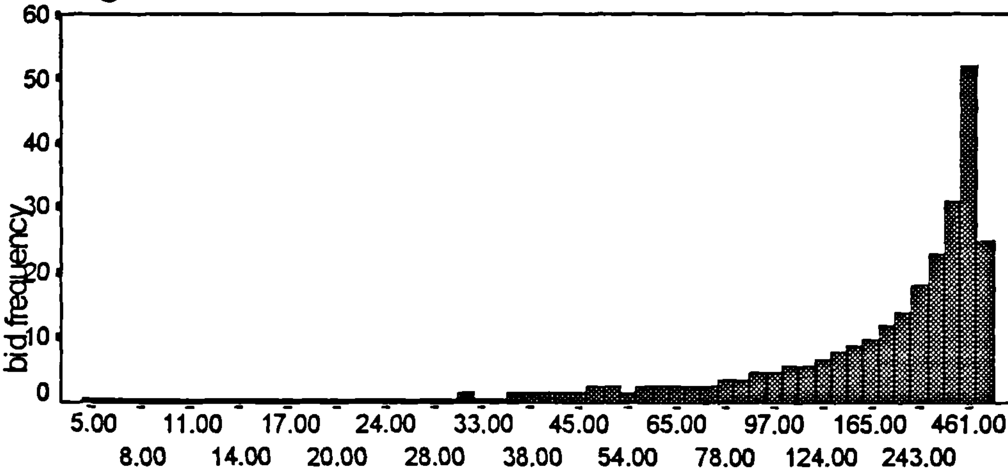
Fig.2



sample size =78

open-ended data (lacre and payment card) outliers and extremes shown

Fig.3



bid

lognormally distributed bid values

4.7 Dichotomous choice format design

Open ended pilot surveys have typically been the starting point for the design of a dichotomous choice (DC) question format. As noted in chapter three, the design of the vector of bids to be offered in DC questionnaires is subject to considerable uncertainty in that the distribution of responses is unknown before placement. Bid selection therefore involves a trade-off between optimally locating bids where they provide most information on the underlying WTP, while assuring the bid curve is well defined. In essence the latter is taken to mean that lowest offer amounts should elicit an overwhelming 'yes' response while the highest bid amounts should on the whole be rejected. The latter requirement avoids the need for arbitrary truncation of the distribution which might lead to the underestimation of E(WTP) (see for example Randall and Kriesel 1990), or - as sometimes occurs - the need to deploy additional surveys posting higher bids only after the problem has become apparent (See Macmillan *et al* 1996, Soderqvist 1995).

Placement of bids raises several questions. First, how high should the bid range extend given the need to identify the tail of the distribution but maintain consistency with economic theory (eg restrictions on WTP relative to income)? Should the range be bounded by an open-ended pilot survey and how much weight should be located in its tails in order to validate the acceptance of extreme bids and predicted probability estimates for out of sample observations? Placing too many bids far out may result in poor fit with little variation in the dependent variable. In designing a discrete choice experiment such questions are common, and it becomes apparent that the DC format merely reinterprets the problem of extreme observations typical with the open-ended alternative.

There is currently some disagreement over design methods, the conventional wisdom being to locate sufficiently high bids into the tail of the distribution in order to minimise an open tail problem and the potential underestimation of the resulting willingness to pay estimate. Recently Kanninen (1995) has provided a rule of thumb (see below) maximising the observed responses in the area of the distribution most likely to contain the true population mean and thereby minimise the bias and variance of model parameters. Either way, consistent design depends largely on the information derived from a pretest open-ended survey to set the bounds of the referendum format.

4.8 Optimal bid design

Much of the recent CV literature has attempted to learn from biostatistical design criteria which were mentioned in the previous chapter. In essence, that literature tends to produce unsuitable bid designs for CV, while requiring knowledge of the WTP distribution which the researcher will not have. The discrete choice design method here followed Cooper's optimal equal area bid design method for minimising the mean square error of the resulting willingness to pay estimator (Cooper 1993)[13]. The two stage design algorithm requires a distributional assumption for the preferred open-ended data set to define the parameters on a bid selection process. Cooper's design places considerable weight on getting the 'right' open-ended sample for the design, recommending several preliminary samples to assess response consistency and to avoid the possibility of diffuse information in small samples. As this is not always an option, some discretion is necessary when following the design procedure.

For a given sample size, the process allows the investigator to specify the optimal number of bid amounts and the optimal sample design or the sample design for the unique number of bids. In practice the latter is favoured for most CV studies limiting survey design to offering 5-10 discrete bid amounts which is convenient for purposes of administration and codification particularly of mail surveys. It is of some interest to experiment however and to contrast approaches to survey design.

Bid Design

The design method sets DC bids according to the information derived from the open-ended survey. At the outset a decision on an open-ended data set should be justified. Somewhat contrary to the findings presented in the preceding section, bid design is based on the information provided by the set generating the more conservative (total) open-ended WTP estimate elicited using the lacre and payment card method. However, for the purpose of DC survey administration, the licence vehicle was favoured over the lacre to avoid any confusion related to annual versus multiple payments. The box plot figure 2 shows that the selected open-ended data set was asymmetric with a highest bid at approximately $R300. If this is the underlying structure of willingness to pay then it seems reasonable to distribute bid amounts accordingly.

In determining the distribution of bids in discrete choice CV, economic theory provides no a priori guidance on the distribution of WTP. Income may be one of the main determinates of WTP, and if this is typically skewed, then one might expect WTP to be similarly distributed. However this assumes much about the role of the income elasticity of demand (or more correctly the WTP

[13]I thank Joe Cooper of the USDA Economic Research Service for providing a copy of the algorithm (DWEABS - Distribution-Weighted Equal Area Bid System).

elasticity)[14] which - as the scoping debate seems to suggest - should also depend on the substitutability of the good. We might also expect a random utility model with non negative preferences to imply a skewed rather than a symmetrical distribution.

A Box Cox test was run on the data to test the null hypothesis that the pretest data was lognormally distributed versus the alternative of normally distributed. The lognormal is perhaps the commonest asymmetric form, and the test assumes that there exists a value λ by which the random variable y is transformed so that:

$$\frac{(y_t^{\lambda}-1)}{\lambda}=x_t$$

Where if $\lambda = 0$, y_t is distributed lognormally and if $\lambda = 1$, y_t is distributed linearly. The test revealed a value of $\lambda = 0.26$ which did not allow the null of lognormality to be rejected at the 95% level[15].

Cooper's optimal bid design (Cooper 1993) provides an algorithm to design a bid vector which minimises the mean square error of the willingness to pay estimate. The design is akin to the C-optimal design criterion in that it minimizes of a function of the parameter estimates. To circumvent the distributional design handicap presented by CV (discussed in chapter three[16]), the method simply substitutes information derived from the distributional assumption inferred from the pretest data for the unknown (and a priori unknowable) parameters of the true WTP distribution.

Recall that the design problem involves the selection of a number of bid levels m, the amount of each bid level b and the number of respondents n to receive each b and sample size $N = n*m$, (and typically $m \leq N$). In typical survey design there has been little systematic about the choice of m

[14]See Flores and Carson (1995) for the difference between these two measures.

[15]A non-parametric Komolgorov-Smirnov test (Conover 1980) confirmed that the open-ended data were distributed approximately lognormally ($p = 0.31$ for the null of normality for the random variable ln WTP).

[16]Namely that the maximum likelihood problem for minimum variance parameter estimates from a DC model can be shown to include bid threshold information in the Fisher information matrix (derived from the Hessian of the problem).

except probably the convenience of say 5-10 bid levels, with b occasionally determined by log-linear or simple equally spaced increments between the ms and the limits bounded by the extremes of an open-ended survey. N in most cases, will be an educated guess, as the investigator will necessarily need to make allowances for incomplete questionnaires and other unforeseen events reducing the response rate.

Having selected a lognormal distribution to approximate the pretest bids, the algorithm[17] divides the density into equal areas and sets the bids at the borders between the areas such that the number of unique bids is the number of equal areas minus one. The algorithm provides the option of limiting m to a pre-specified number or allows it to be dictated by the distribution. The bid amounts are given by $b_i = F^{-1}(P_i)$, where $F^{-1}()$ is the inverse of the cumulative distribution function and P_i (the order of the distribution corresponding to the quantile b_i) is defined as:

$$P_i = i/(m+1),\ i = 1,2....m.$$

Placement of m by equal area is dictated by the shape of the distribution. This means that bids are closest together in the region of highest density but as the tail gets thinner, spaces between m increase at a rate which is dependent of course on the pre-specified distributional assumption. There is no truncation limit except the fact that the area of the density is finite. In the case of the lognormal assumption, the thick right tail will inevitably influence the spacing as well as the allocation of the number of individuals to each b_i, which is the second stage of the design process.

The objective is the minimisation of the mean squared error which - striking a balance between unbiasedness and variance - is a generally accepted measure for the performance of an estimator, given as

minimize $$MSE(WTP^*) = (WTP - WTP^*)^2 + var(WTP^*)$$

subject to:

$$\sum_{i=1}^{m} n_i = N$$

[17]Description of the bid assignment procedure follows Cooper (1993).

where $n_i \geq 0$ for $i = 1 \ldots n$,

where WTP* is the design program estimated willingness to pay (in other words that corresponding to the programme-determined optimal bid distribution, or combination of m,b_i and N), to be distinguished from WTP which is that of the true population mean (both of which to be distinguished from WTP resulting from the parameters estimated using the ultimate bid design and logit model).

In the second stage given b, the mean square error minimizing n (the number of individual respondents at each m) is determined according to this minimization criterion.

The algorithm employs a discrete linear approximation of the continuous mean and variance of a non zero random variable (here the lognormal distribution). This is essentially a trapezoidal approximation of the integral of the function fitting the open-ended data, giving for the mean :

$$WTP^* = \sum_{i=1}^{m} \Delta b_i q_i$$

$$\Delta b_i = (b_{i+1} - b_{i-1})/2, i = 2, \ldots\ldots m-1$$

In other words as m gets bigger[18] the better the approximation to the integral.

and for the variance:

$$Var(WTP^*) = \sum_{i=1}^{m} (\Delta b_i)^2 q_i (1-q_i)/n_i$$

where $q = n_i(\text{yes})/n_i$ is the proportion of 'yes' responses to b_i which is unknown (before the survey takes place) and is approximated in the algorithm by 1 - $F(b_i)$ from the assumed cumulative density function for the open-ended data above.

[18]m is a choice variable and can be set by the investigator or automatically determined by the programme for a given distribution and sample size.

With this information it is possible to set up a lagrangian to minimize the expression for the MSE(WTP*) above, with respect to n_i for given m. This turns out to be the same as minimizing the expression for the variance, since the bias portion is not a function of the n_i's. The resulting expression n^*_i the variance minimising respondent allocation to b_i (see equation 6 Cooper 1993) and that for the WTP* can then be plugged into the original MSE expression in which WTP is the average from the open-ended survey. These substitutions give a minimum MSE for given m, MSE*.

Cooper's program[19] carries out these steps by conducting a scan over the integer values of m = 1 to N and choosing the m that minimises MSE and then defining b_i^* , the vector of optimal bids for the whole sample, i = 1,2,...n.

Essentially the bid selection criterion aims to minimize the possibility that the distribution of bids is different from the distribution of actual sample willingness to pay, thereby minimizing the likelihood of underestimating E(WTP) or losing information by disproportionate placement in the tails. It is important to note that the design method places considerable weight on the validity of the open-ended survey to predict how responses will be distributed. There is no guarantee that this will the case even when resources are available to conduct large open-ended pretests. For example, one strand of opinion is that respondents will simply process the information available in the discrete-choice format differently than when arriving at an open-ended response. To the extent that this is true then response format will be one reason for distributions to divergence.

The bid vector

Figure 3 shows the lognormal frequency distribution of the amounts shown in table 6 used in the main survey. The initial survey design for 300 was subsequently enlarged to 400 when a higher than expected response rate became apparent.

Several things are worth noting. First, the number of bid amounts is not typical and is a result of allowing the DWEABS algorithm to design an optimal vector without specifying the number of bids. Second, use of the lognormal bid structure was the very obvious location of numerous high observations into the right tail of the density. While this may assure that FB = 1 as b_i goes to infinity, there is a potential loss of information entailed by erroneously positioning bids. Furthermore, the structure entailed by the placement leaves relatively few observations at lower bid levels, and

[19]The algorithm is written in GAUSS format and took about 5 minutes on a 486 desktop with a Pentium processor for an optimal design and a sample size of 300.

Table 6 Bid Vector for discrete-choice analysis

BID ($)	SAMPLE SIZE		
5.00	1.00	78.00	4.00
6.00	1.00	84.00	4.00
7.00	1.00	90.00	5.00
8.00	1.00	97.00	5.00
9.00	1.00	105.00	6.00
10.00	1.00	114.00	6.00
11.00	1.00	124.00	7.00
12.00	1.00	135.00	8.00
13.00	1.00	149.00	9.00
14.00	1.00	165.00	10.00
15.00	1.00	185.00	12.00
16.00	1.00	210.00	14.00
17.00	1.00	243.00	18.00
18.00	1.00	287.00	23.00
19.00	1.00	352.00	31.00
20.00	1.00	461.00	52.00
21.00	1.00	701.00	25.00
22.00	1.00		
24.00	1.00		
25.00	1.00		
26.00	1.00		
28.00	1.00		
29.00	1.00		
31.00	2.00		
33.00	1.00		
34.00	1.00		
36.00	2.00		
38.00	2.00		
41.00	2.00		
43.00	2.00		
45.00	2.00		
48.00	3.00		
51.00	3.00		
54.00	2.00		
57.00	3.00		
61.00	3.00		
65.00	3.00		
69.00	3.00		
73.00	3.00		

forecloses on any form of robust within-bid analysis should this be considered important. Use of the lognormal bid structure therefore requires faith in the location of the true mean and this highlights the luck of the draw nature of optimal bid design.

4.9 Model estimation - single discrete choice model

Following the question formats presented in table 1 (yes, no, don't know), there are several possible permutations for analysing respondent willingness to pay and for deriving the preferred estimator. This section presents parametric and non-parametric results from the single DC question, plus the parametric interval and bivariate model of the double-bounded format. The choice between the models is essentially down to professional judgement, and the requirement to opt for conservative estimation procedure consistent with the NOAA panel recommendations. In cases where the rationale for a decision is ambiguous, 'the option that tends to underestimate WTP is preferred' (Arrow *et al* 1993; p.4612). Accordingly mean values are estimated for both the clean and 'conservative' data sets.

Logistic and log-logistic regression

Irrespective of the selected distribution of the bid vector it seems impossible to speculate a priori about the best fitting model of actual responses. The best procedure should be to vary the distributional form and compare models on the basis of predictive ability. Several forms such as the logistic, the log-logistic, the Weibull and the log-normal can be used although most studies automatically adopt the first two of these (see Lee 1992 for the relative properties of these distributions). Inspection of the plotted proportions for the single DC response figure 4, also provides an additional clue about behaviour of the tail of the distribution. The log-normal may be discounted on intuitive grounds because of its fat right tail which in the absence of truncation is likely to lead to an inflated mean[20]. In this case the choice between the remaining models was made on the basis of separate univariate regressions of the bid (or log of bid) variable on the response probability. These revealed similar log- likelihood statistics[21] indicating the models provided roughly similar estimators of the parameter vector. For purely computational ease therefore, logistic and log-logistic regressions were selected to analyze the single response DC questions, on the basis that the incentive-compatible

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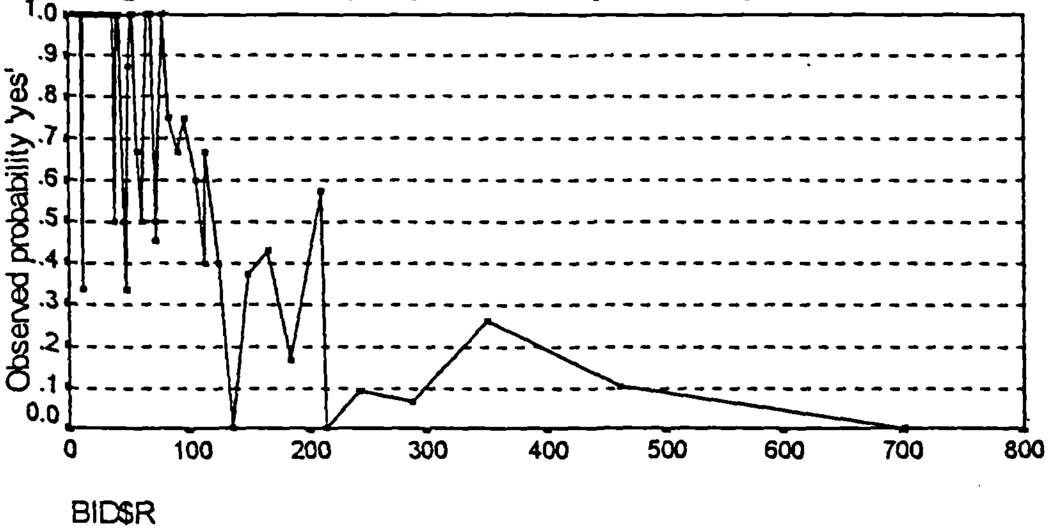
[20]Inspection of the results of other studies reporting the means derived from a lognormal model confirms this (eg Imber *et al* 1991; Leon 1996).

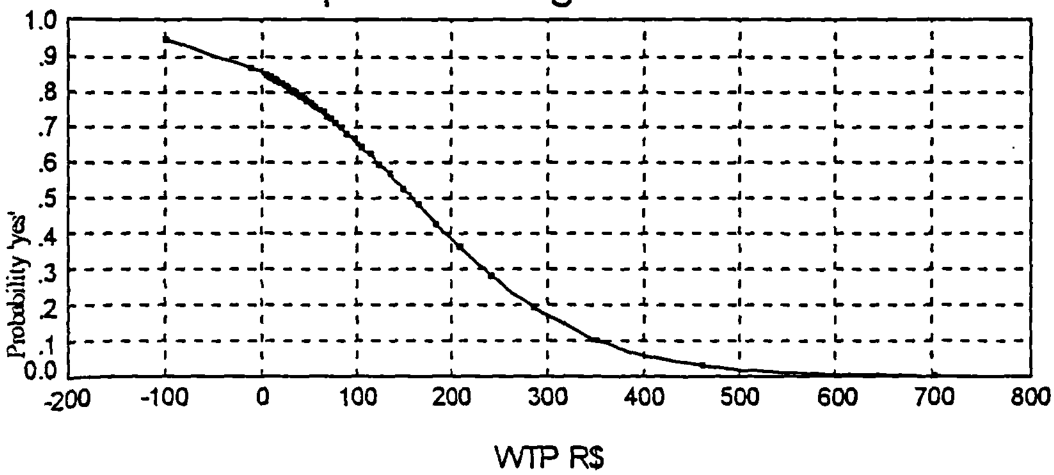
[21] The relative size of the log-likelihood statistic of two models differentiated by a distributional assumption allows the choice of the estimator which maximises the probability of observing the response probabilities actually observed. Cramer (1991) reviews diagnostic statistics for logit models.

Fig.4 Plotted proportions 'yes' response



Dichotomous choice (first response)

Fig.5 Parametric analysis predicted logit function



Dichotomous choice (first response)

properties of the format should best be reflected in the single question. The logistic version will be the only one generalised to evaluate the effect of additional covariates.

$$Probability'yes' = \frac{1}{1+e^{-z}}$$

where z represents the favoured utility index or some generalized form including additional covariates.

Equation forms for deriving the mean and median from the linear and log (logistic) functional forms were presented in chapter three. Tables 7 and 8 present the estimated model coefficients which will be used to calculate mean willingness to pay. Accordingly for the linear form this restricted model correctly predicts 79% of the yes and no responses actually observed with the bid variable highly significant. It is as well to note that in contrast to OLS[22] the omission of other relevant explanatory variables has the effect of biasing the coefficient β towards zero (see Cramer 1991, p.37 for a proof). As such, it is necessary to compare the performance of the model with additional covariates, to be sure to avoid any bias in mean/median WTP. Which variables to include is a matter of debate. In particular the income variable, and how to reflect the marginal utility of money between two utility states. Omitting income is tantamount to assuming constant marginal utility of income between states under consideration. Desvousges *et al* (1992 p67), point out that this can be the case only for perfectly replaceable commodities. Including income allows for a test of perceived uniqueness in the sense that the marginal utility of income varying with the level of environmental change (implied by a significant coefficient) would not occur if perfect substitutes were available.

The log-logistic (table 8) apparently provides a superior fit for the restricted version of the model, predicting 82% correctly. Judgement on the use of this model is reserved for the following section. Further issues of model mis-specification are examined by Ozuna *et al* (1993). In the case of the double-bounded model (below) fit issues are addressed by Kanninen and Khawaja (1995).

Initially five additional variables were codified to be included in the regression analysis. Table 9 presents the maximum likelihood parameter estimates of the same variables and related diagnostic statistics. Variables Bid, Income and Know are significant at less than the 1% level and have the expected signs. In the latter case it is expected that informed respondents are more disposed than those

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[22]where the effect of omitted variables only shows up as increased residual deviance provided omitted variables are uncorrelated with retained regressors.

Table 7 Logit model (first bid)

Logit regression, (1/0 dependent variable)		
Variable	Coefficient	t-stat.
Constant	1.7663	6.975
Bid	-0.01113	-7.709
n=267 Log-Likelihood = -123.83 Restricted Log-Likelihood (slopes = 0) = -184.39 McFadden's R^2 = 0.33 % correct predictions 79% y=1: 124, y=0:143		

Table 8. Log-logistic (first bid)

Log-logistic regression, (1/0 dependent variable)		
Variable	Coefficient	t-stat.
Constant	8.6962	8.258
ln Bid	-1.8299	-8.553
n=267 Log-Likelihood = -112.36 Restricted Log-Likelihood (slopes = 0) = -184.39 McFadden's R^2 = 0.39 % correct predictions 82% y=1: 124, y=0:143		

Table 9. Multivariate logit

Additional covariates in logit regression			
Variable	Coefficient	t-stat.	Definition and coding
Constant	-0.56403	-0.54874	
Bid	-0.01195	-7.46889	WTP discrete amount $R (Q.22)
Income	0.00023	3.40298	Selected household annual income band (all sources) (Q.27)
Visit	-0.01664	-0.67697	Total number of visits made to the Pantanal area (Q.6)
Know	1.26476	3.61938	Prior knowledge of the pollution problem 1 yes, 0 no (Q.20)
Catch	-0.00413	-1.28193	Total fish catch by party (all species) in kg (Q.19)
Age	0.02399	1.04733	Respondent age in years (Q.26)

n=267
Log-Likelihood = -100.26923
Restricted LL. (slopes =0)= -184.39
McFadden's R^2 = 0.40
% correct predictions = 83%
Chi-squared (6) = 148.25 (critical value 12.59)

learning of the problems for the first time. This is a potentially interesting finding as the knowledge issue remains a sticking point in CV deliberations. In particular can people value things they don't know about and what types of prior information should 'count'.

Variables Catch and Age (only significant at 0.19 and 0.29 levels respectively), provide somewhat counter-intuitive results. In the case of age, the positive coefficient suggests WTP increasing with age - typically the reverse is true - while WTP seems to be inversely related to the variable Catch. A plausible explanation for the latter relates to the interpretation of the last variable 'Visit', which is the least significant variable of the regression, although of negative sign. Either of these variables are the quantity variable in a demand relationship. Both are in the range $-1 < d < 0$ which is consistent with economic theory and implies that the Hicksian demand curve will be downward sloping.

Because the dependent variable of the regression is a probability, the effect of any continuous independent variable must be viewed in terms of a quasi-elasticity defined as:

$$\delta P_i / \delta x_{ij} = \beta_j \, P_i(1 - P_i)$$

In other words, the regression coefficient gives the change in the log odds for a one unit increase in the variable. In general, for an increase in m units in the variable, the log odds ratio is equal to m times the logistic regression coefficient. The value of the elasticity will vary with X and it is usually evaluated at the sample mean.

Overall this model is an improvement on the restricted single bid predicting 83% of the observed responses and a likelihood ratio test of favouring the unrestricted model (distributed χ^2 with 5 d.f. compared to a critical value of 11.07).

4.10 Mean/median estimation

Plugging the estimated coefficients from table 7 into the logit model we have:

$$Probability(odds)'yes' = \frac{1}{1 + e^{-[1.7663 - 0.01113(\$A)]}}$$

which is the predicted function[23] in figure 5. Reading off from probability 'yes' = 0.5 shows the median lying around 150 real. This can be validated by solving the above for Pr (yes) = 0.5.

$$\log(odds)\frac{0.5}{1-0.5}=1.7663-0.01113(\$A)$$

giving

$$\$A=\frac{1.7633}{0.01113}=158.69$$

which of course is also the formula α/β for the unrestricted mean given in table 1 in chapter three. Note however that extending the plotted function shows high probabilities of negative willingness to pay (eg the probability of a yes response to WTA \$100 is around 0.95). Ruling out such a finding, the restricted mean is:

$$E^{+}(WTP)=\frac{\ln(1+e^{1.7633})}{0.01113}$$

which truncates negative values giving a higher mean of 172.92. Use of the log-logistic model parameters offer an alternative to the truncated function and figure 6 shows the response function of the model using the parameters from table 8. As well as the desirable property of avoiding negative values, the non linear ln(bid) transformation does introduce the undesirable problem of making the mean much more sensitive to predicted tail values than using the raw bid data. In fact, for certain parameter values ($-1 < \beta < 0$), the mean of the distribution is undefined or infinite (Hanemann 1984). As can be seen, the parameter value -1.83 still makes the right hand tail quite sensitive to small changes in beta. Thus, while the median of \$R116 comes close to the confidence interval of the open-ended data, the mean of \$R6638 derived using the appropriate expression is clearly well outside

[23]Recall that for the unrestricted model the correct term for the unrestricted mean is $1/\beta(\alpha+\Sigma\gamma_i s_i)$ where γ_i is the estimated parameter corresponding to the socioeconomic characteristic s_i, which is evaluated at its mean.

Fig.6 Parametric analysis predicted function (log-logistic)

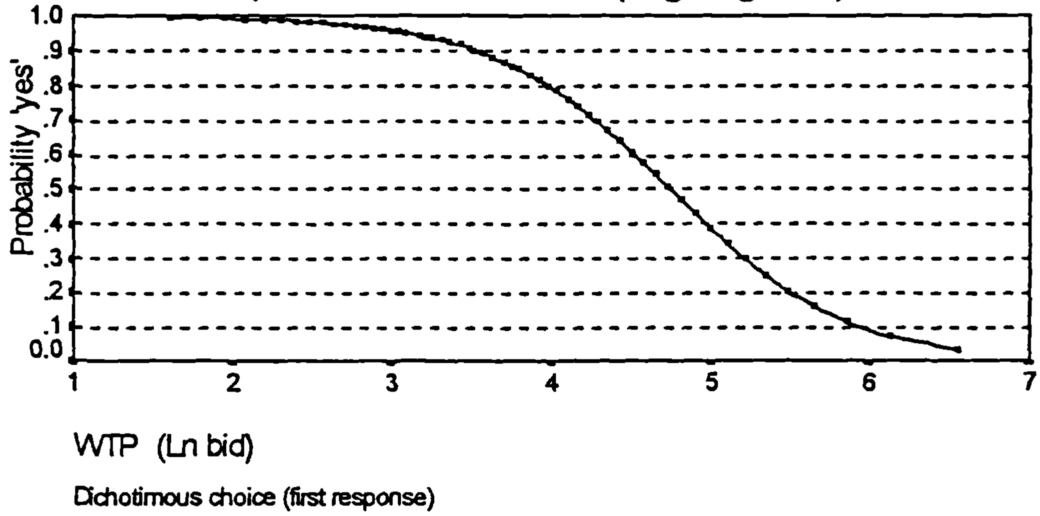
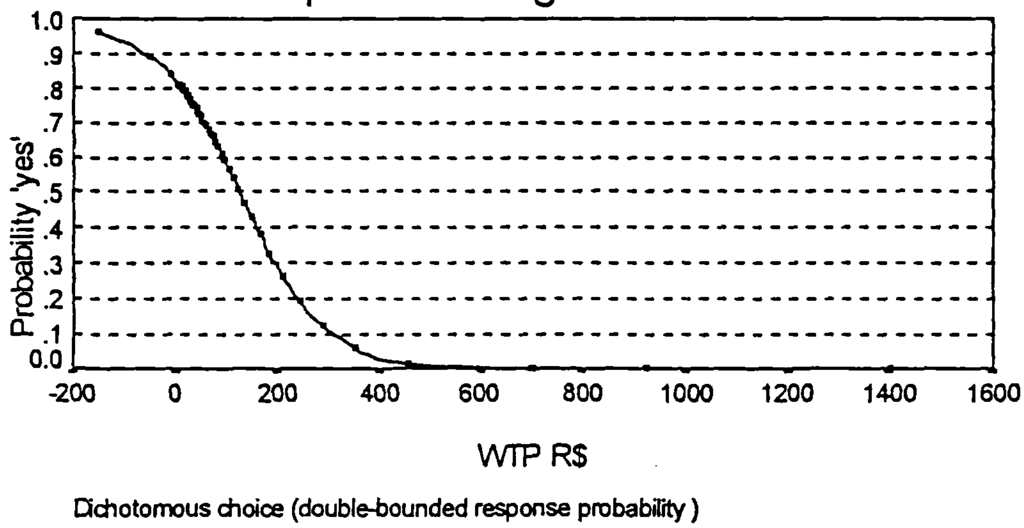


Fig.7 Paramteric analysis predicted logit function



both open and closed ended bid ranges[24]. Since there is no a priori reason to truncate bid values used in the study, the linear model is preferred. Unless otherwise stated, and consistent with the view of the conservation of the Pantanal providing positive utility the restricted (mean) estimate shall be used.

Figure 5 makes several other issues clear. First, the predicted function shows that response probabilities are at least price sensitive, which might not be the case were responses driven by any social desirability or symbolic motive. Related to this is the magnitude of the slope parameter if the mean is defined by the area under the positive function. Second, the function does not have a fat tail, indicating the need to truncate and potentially underestimate the mean. Note that from the predicted function the probability of observing a yes response at the highest offer amount 701 is only 0.002391, little more than 1 in 1000. This raises the issue about the wisdom of bid placement according to a lognormal distribution. The highest value appears to have identified the tail of the distribution, although inspection of the plot of proportions in figure 1 suggests that this might have been identified at a much lower value. In other words, the bid placement may have over-estimated WTP and lost information from locating a disproportionate number of bids at distance from the true sample mean and median.

4.11 Double-bounded and bivariate estimation

As noted, there is a growing literature on optimal design for double bounded estimation. Much of this was unknown to the author at the time of the design, and the bid amounts were arbitrarily doubled and halved. Individual WTP is therefore contained within intervals B, B_u and B_d which are values double and half the bid vector used in the single response model above. This design choice is qualified by recent findings below.

Because of the question sequence used in the survey (table 1 above), the double-bounded format can be analyzed in a number of ways to include the 'don't know' responses.

The user defined Minimize command in LIMDEP 6.0 was used to mimic the log-likelihood function for the double-bounded likelihood estimator in chapter three. The method requires starting values to

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[24] Extreme results of this nature are not uncommon in DC modelling. Reviewing the forms tested by Imber *et al* (1991), reveals that mean estimates ranged from Aus $160.64 for the logistic to Aus $756,181.811 for the lognormal! For the log-logistic the authors suspiciously only provide the median. The use of a log transformation makes both models highly sensitive to the behaviour of the tails.

speed convergence by Newton's method (see Cramer 1991 p.19), and these can be taken from the output of the logit command used to estimate the single DC response model. The Matrix command can also be used to generate the variance covariance matrix of the MLE parameter vector, for use in the generation of confidence intervals by the method of Krinsky and Robb.

Table 10 contains the parameter estimates of a model plotted in figure 7 and used to calculate the mean and median WTP presented in table (below). Note the similarity of the response probability functions apart from the now extended right tail, which shows that the predicted probability of observing a yes to 701 has reduced to 0.000862 while that for the highest possible bid 1402 is only $7*10^{-7}$. Evidently the double-bounded model is giving less weight to amounts that were not accepted than the single response model, while the integral of the function beyond the original highest bid value adds a negligible amount to the density. This is precisely because the follow-up provides an opportunity to retrieve some of the respondents who refused the original highest bids thereby generating more information about the shape of the distribution somewhat nearer the true sample moments. As noted by Kanninen (1995), this is one of the advantages of the interval model when bid design is poor. The double-bounded model also typically leads to a lower mean estimate than the single format.

Bivariate model

Recall from chapter two that Cameron and Quiggin (1994) entertained the idea that first and second responses in the double-bounded format might be motivated by separate underlying WTP distributions. The determination of a correlation coefficient between both sets of responses in theory enables model selection between the interval model ($\rho=1$ eg identical distributions) or the theoretically correct bivariate alternative. However there is some uncertainty about both the validity of the test and the extent to which an imperfect correlation will seriously compromise the reliability of the interval model parameters (Alberini 1995). With certain survey designs (eg mail) it is reasonable to suspect that strategic behaviour will be an issue, and as a result, at some point the coefficients underlying the first response will be an inappropriate description of the second. If so, mean estimation becomes problematic. As a diagnostic check the bivariate probit model revealed a ρ = 0.635 suggesting imperfect correlation. However Alberini (1995a), simulates the effects of different values on the bias and efficiency of mean estimates, and shows how the bivariate model inflates the mean considerably relative to the interval model. The use of the bivariate probit is above all handicapped by the maintained hypothesis that the data are distributed as bivariate normal.

An alternative test is to analyze first and second responses separately and to conduct some type of

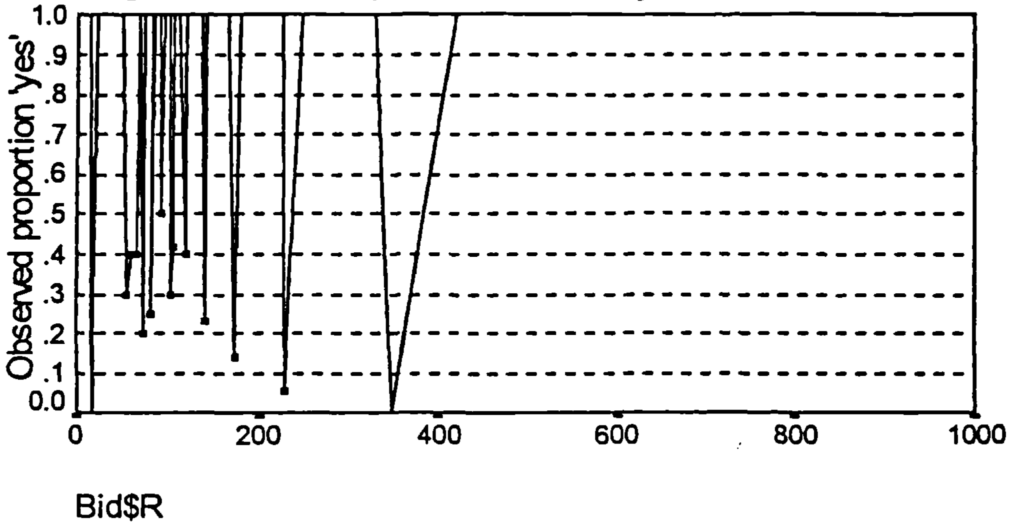
Table 10

Double-bounded model		
Variable	Coefficient	t-stat.
Constant	0.61334	2.102
Bid	-0.01296	-11.04
Income	0.00020	4.213
n=265 response cases: yes-yes = 78; yes-no = 44; no-yes = 36; no-no = 107 Log-Likelihood = -264.832		

Table 11 Logit model (second bid only) and estimated means

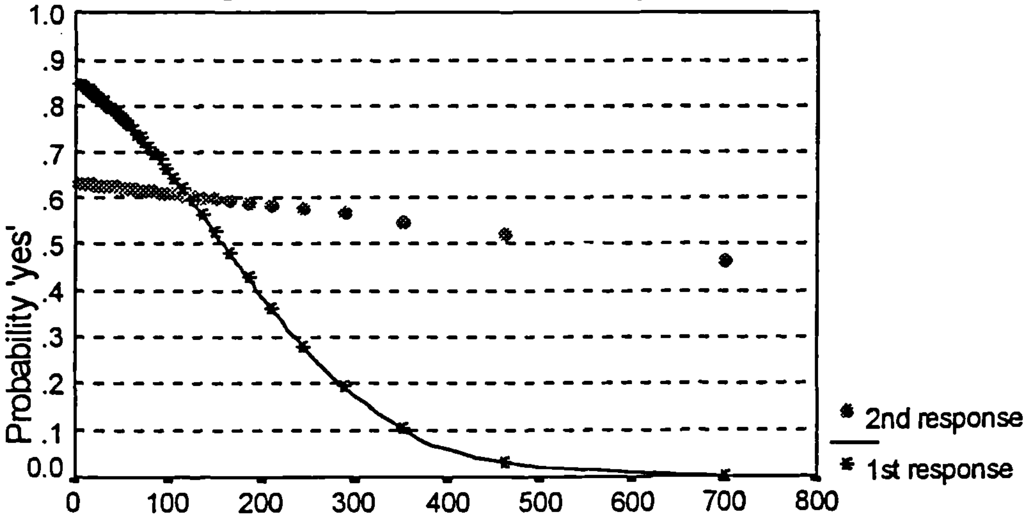
Logit regression, (1/0 dependent variable)		
Variable	Coefficient	t-stat.
Constant	0.55396	2.99229
Bid	-0.00098	-1.21249
Restricted mean point estimate 1028.7 (465.84 - 20081.74)		
n=265 Log-Likelihood = -178.00 Restricted Log-Likelihood (slopes = 0) =-178.74 McFadden's R^2 = 0.004 % correct predictions 56% χ^2 =test for regression significance = 1.48 (significant only at 0.22) y=1: 158, y=0:107		

Fig.8 Plotted proportions 'yes' response



Dichotomous choice (second response only)

Fig.9 Parametric analysis



Dichotomous choice: predicted functions for 1st & 2nd responses non-parametric distance test between two distributions. For present purposes it is sufficient to analyze the second responses of the sample in isolation from the first response[25] to investigate the cause of any likely divergence between responses.

Table 11 repeats the single response logit estimation this time for the second response. The consequences are somewhat alarming. Firstly, the regression is only significant at the 22% level, and predicting 56% of responses correctly. The plotted proportions of figure 8 - showing universal acceptance (rejection) for several very high (low) values helps to validate such poor predictive power, while figure 9 shows that the predicted logit function is almost flat, such that the extrapolated tails are likely to be fat for very high positive and negative values. The result is that integrating for the restricted mean, results in a very high point estimate of 1028.7 (465.8 - 20081.74) whereas an unrestricted mean of 565.33 is not significantly different from zero[26] at the 95% level of confidence. Similarly in contrast to the first response, the median is located in the vicinity of $R550. Clearly these results in their own right would be unacceptable.

There is one very likely and another possible explanation for this observation. The possible explantation relates to latent yea-saying behaviour on the part of respondents which is exacerbated by a follow-up question. Clearly the plotted proportions suggest that some very high values have been accepted as follow-up bids, resulting in insufficient response variation relative to the 'typical' function associated with the first response. Yea-saying behaviour is possibly motivated by some form of symbolic commitment or social-desirability, and the extent to which it is inevitable in follow-up questions is as yet, an open issue. The issue will be further addressed in the next chapter.

The alternative likely reason for the shape of the predicted function relates to the arbitrary bid positioning resulting from simply doubling and halving the original optimal design. Table 6 shows how bids are concentrated by the lognormal distribution, such that many low values and their follow-up multiples are typically universally accepted. However, the sparsity of multiple observations at every amount for very low (and a few moderately high) values means that only one (or very few) unexpected rejections or acceptances are sufficient to upset the behaviour of the whole function. In other words, the result clearly demonstrates how the variance of the function parameters can be reduced by simply increasing sample size at each bid value. In truth, the result found here is likely

[25]Strictly speaking the second response is conditional on the first, such that it is unlikely that response motives will be entirely unrelated.

[26] -13.04 - 1143.71

to be due to a combination of yea-saying behaviour and poor design. However the simple analysis goes someway to suggesting the poor fit offered by the bivariate model and yet again, the perils of poor bid distribution are evident. The important question however, is to what extent such a finding invalidates the double-bounded model? Some guidance on the placement of follow-up bids will be given in a later section.

4.12 Nonparametric analysis

As previously stated nonparametric methods make no assumptions about the form of the distribution of the random error, while the monotonicity of the standard survival function amounts to a much weaker assumption about preference structures.

Although there is no reason to suspect that the logistic distribution is inappropriate for modelling initial responses, it is of some interest to see how the removal of this assumption effects the estimate of mean WTP. Many non-parametric survival and product life routines are available, and this section reports the results from the analysis of the single response DC data using the Kaplan Meier routine (see Greene 1993 p.725). The approach has approximately the same properties as that suggested by Kristrom (1990), and has been used in other CV studies (e.g. Imber *et al* 1991).

The survival function is closely related to the familiar response probability and merely plots the probability than an individual survives past certain events which are the discrete WTP amounts. The bids are arrayed in ascending order $t_{(1)} \leq t_{(2)} \leq \ldots t_{(n)}$ so that the survivorship function S at amount $t_{(i)}$ can be estimated as

$$S(t_i) = n\text{-}i/n = 1\text{-}i/n$$

where n - i is the number of individuals in the sample willing to pay more than $t_{(i)}$. The routine simply allocates the right censored data into a step function where the steps are the maximum likelihood proportions corresponding to each WTP threshold. The routine imposes the self consistency property[27] when there are multiple or overlapping estimates for a particular interval such that the survival function is (weakly) monotonically declining in price. The necessary

[27]Recall from chapter 3 that this basically involves adding successive proportions to make sure the function is monotonic.

Fig.10 Nonparametric analysis
survival function (Kaplan Meier)

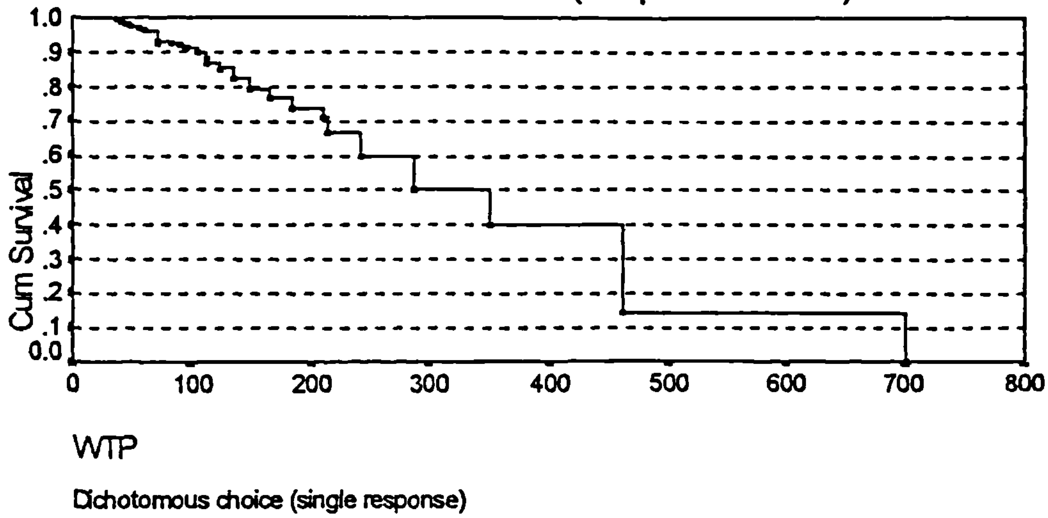
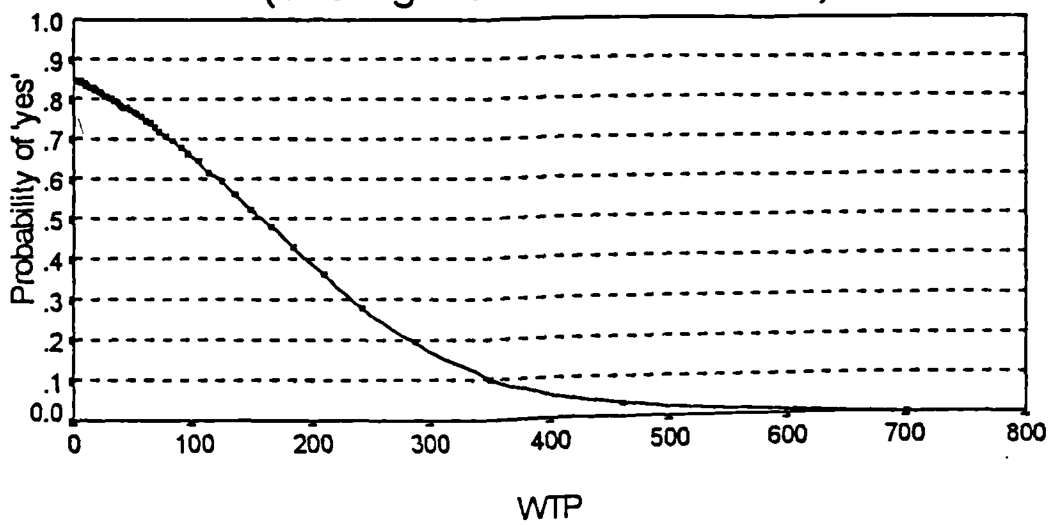


Fig.11 Parametric analysis
(two highest values removed)



assumptions about the end-points to close the function, are typically that everyone survives a WTP of 0, while nobody survives the highest amount. These restrictions on the end points mean that the non-parametric mean estimator - the area under the function[28] - is suitable for comparison with the restricted mean of the logit model (if of course one abstracts from the influence of the distributional assumption).

Figure 10 shows the estimated function for the single response data. Note that, as in the procedure used by Kristrom (1990), it is also possible to linearly interpolate between bid values which essentially amounts to an alternative distributional assumption about the censoring behaviour between bid amounts. Although the points are not interpolated here, this may be preferable for two reasons. First, the interpolated function will down-play the importance of the long right tail of the function where bids are considerably spaced. Second, in this case the step at probability = 0.5, the median is not unique. A description of confidence intervals for non-parametric estimators - standard output from routines in programmes such as SPSS -can be found in Lee (1992).

4.13 Comparison of means

Table 12 presents the mean estimates from the cleaned data set from for the models discussed above, as well as the conservative means for the single and double-bounded models which are expected to underestimate the welfare measure. All DC means are at least inside the bid vector from which they were obtained[29], although the variability between DC means and their size relative to the open-ended estimates is disconcerting. Taking the DC variability first, it is possible to rationalise the disparity on the basis of the bid vector and the selected model. First and foremost, the asymmetric design of the bid vector may have been inappropriate in extending the right tail of the density over excessively high values. This has circumvented the fat tail problem but potentially biased the mean.

Estimation by Maximum Likelihood, results in consistent parameter estimates as the sample size increases. Bias and variance are also a function of the sample size, and can also be affected by the parameters of the WTP distribution and the explanatory variables (the bids).

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[28]Carson *et al* (1994) propose an alternative conservative mean estimator based on the lower bound of each interval. Thus if 20% of the sample happen to be in the interval of R$10-$25, the lower-bound mean is calculated by assuming that this 20% of the sample is WTP exactly R$10.

[29]All but the nonparametric estimate are also bounded by the open-ended bid range.

Table 12. Mean estimates

Model	Mean WTP*	Conservative Mean**
Open-ended	52.76[1] 35.09 - 70.39 89.74[2] 74.2 - 103.8	
Single-bounded	168.29 144.34 - 200.15[3]	137.24 119.43 - 159.66
Double-bounded	137.51 121.71 - 156.15	112.93 98.70 - 128.32
Bivariate probit	215.25 196.37 - 235.80	
Nonparametric single-bound (Kaplan-Meier)	346.10 315 - 376.86	

Notes: * n = 265; ** n = 364. 1. Iacre payment vehicle and payment card; 2. licence payment vehicle, no payment card; 3. 95% confidence intervals by Krinsky & Robb method

Using a result derived analytically by Copas (1988), Kanninen (1995) has evaluated the bias of parameters from hypothetical "true" values and the resulting bias[30] and variance[31] in WTP from various alternative bid vectors Table 13. The designs are two point (the minimum number to identify the two parameter distribution) and various multiple point combinations of the two point designs. For the present purpose, the telling information is the extent to which the bias of alpha and beta work in opposite directions for the various designs. None of the designs has a drastic effect on WTP, and the biases can be reduced by simply increasing sample size. Clearly the design which places values in one tail (relative to the location of the true mean), has the effect of increasing the variance. This turns out not to be such a problem when a double-bounded model is used because of the 'retrieval' opportunity offered by a second bid. As for the single response, table 13b offers some rules of thumb for the placement of follow-up amounts for the double-bounded format. Again, these are based on the logistic distribution and therefore do not apply to other forms which might reasonably be fitted to the follow-up responses. The interesting finding is that even for the design which completely misses the true mean, both bias and variance results are surprisingly resistant.This suggests a merit of the double-bounded approach despite the finding of figure 9.

Taken together these findings suggest that in covering the whole function we can be reasonably confident of avoiding bias problems, although the logic of the number of bids in the tail dictated by a lognormal vector is questionable. For a logistic distribution - which in this case turns out to be the correct distribution (and the one on which Kanninen's design criteria are based), the likelihood of observing 'yes' responses in the tails beyond say 12% is very small, such that at best, extreme placement simply wastes observations. At worst, consistent yea saying can distort the fit of the model, and therefore WTP. As it happens, this has not occurred here and the derived mean seems fairly reliable.

The resulting rule of thumb is that bids should be kept out of the 15th and 85th percentiles for the single response model which in this case means a range of roughly 50 -350 or that around 40% of the values were sub-optimally located in terms of their remoteness from the true mean. Kanninen's results seem to suggest that the Double-bounded mean is more reliable. There are two important caveats to the rule of thumb however. First, it is extremely difficult ex ante, to judge the degree of

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[30]The bias in estimated WTP is the difference between the actual mean and the estimated mean: - $-\ln(1+\exp(\alpha)/\beta - \ln(1+\exp(\alpha)/\beta)$.

[31]The asymptotic variance of estimated median WTP is calculated using the delta method for a function of two normal random variables (Kanninen 1995 eqn 3).

Table 13a Analytical Bias and Asymptotic Variance of WTP - Single-Bounded Logit Model $\alpha = 2.5$, $\beta = -0.01$ are the "true" parameter values median WTP = \$250, mean WTP = \$258, number of observations = 250

Bid Design	% Bias α	% Bias β	% Bias Mean WTP	% Bias Median WTP	Asymptotic Variance of Median WTP
\$200, \$300	0.83%	-0.83%	0.09%	0.00%	170.21
\$100, \$400	1.14%	1.14%	0.12%	0.00%	268.19
\$5, \$500	1.90%	-1.90%	-0.22%	-0.03%	588.60
\$5,\$200,\$300, \$500 (8%, 38%, 62%, 92%)	1.94%	-1.95%	-0.20%	0.00%	258.85
\$300, \$400 \$500 (62%, 82%, 92%)	4.15%	-3.62%	0.11%	0.51%	1779.19

Table 13b Analytical Bias and Asymptotic Variance of WTP - Double-Bounded Logit Model
$\alpha = 2.5$, $\beta = -0.01$, median WTP = \$250, mean WTP = \$258, number of observations = 250

Bid Design	% Bias α	% Bias β	% Bias Mean WTP	% Bias Median WTP	Asymptotic Variance of Median WTP
B = \$250 (50%) Follow-up = B ± \$100	0.53%	-0.53%	-0.06%	0.00%	128.15
B = \$250 (50%) Follow-up = B ± \$200	0.56%	-0.56%	-0.06%	0.00%	135.41
B = \$250 (50%) Follow-up B + \$250 or B - \$245	0.66%	-0.66%	-0.07%	0.00%	141.45
B = \$300, \$400, \$500 Follow-up B ± \$100	0.90%	-0.81%	0.00%	0.09%	217.11

Source: reproduced from Hanemann and Kanninen (1996) Note that bias calculations are based on a logistic distribution and therefore can at best only hold approximately for other forms.

bias likely to arise since the correct vector, response probabilities, and true parameter values are always unknown. Second, the rule of thumb would seem to prevent identification of the tail of the density. In other words, this would have to be dictated by the distributional assumption made in modelling and therefore makes a somewhat important case for parametric estimation and a choice with a flat tail.

A comparison of the parametric and non-parametric means would seem to add to this last point. The highly inflated non-parametric mean suggests that the influence of the logistic distribution has a tempering effect on the mean, in particular by down playing the long right tail. However the advise to 'bunch' the bid vector will itself influence the corresponding non parametric estimate and an interesting project would be to compare relative behaviour of parametric and non parametric estimates using different bid vectors.

This study adds to the growing list of studies finding a disparity between open-ended and dichotomous choice welfare estimates (table 14). However in making such a comparison, it is important to note that a particular distributional assumption is made for the discrete choice responses which does not apply to the open-ended data. In drawing conclusions, the extent to which inappropriate models and distributional assumptions are driving the disparity is uncertain.

Because of the advantages offered by the double-bounded format relative to the single response, the double-bounded mean is the favoured estimate from the DC format. This mean is nevertheless some 2.6 times the most conservative open-ended mean.

There has been a considerable amount of speculation over the cause of the disparity, which questions the alleged incentive-compatibility of the DC relative to open-ended format. Essentially psychometric anchoring and yea-saying behaviour cannot be ruled out and there is a suspicion that such behaviour may increase when the good being valued is unfamiliar thereby accentuating the problem presented by an inappropriate bid vector. The open-ended-DC disparity may therefore have other largely psychological explanations, which cannot be explained away by relative economic incentives inherent in the format. As Greene *et al* (1995 p.20) put it, the claim of relative incentive compatibility, is "misleading because in the case of purely hypothetical CV questions there are no economic incentives at work, and because both protocols can be framed to be incentive-compatible if subjects are economically rational and believe there is some probability they will be decisive." Unfortunately there has been too little definitive experimental work on the psychological motives beyond the

Table 14 Other CV Comparisons of Dichotomous Choice and Open-ended WTP

Author	Good	Survey administration	Independent samples?	Mean WTP		
				Dicho. choice	Open-ended	Ratio (dc/oe)
Dubgaard (1996)	Landscape	mail	yes	71DK	44DK	1.6
Willis et al (1995)	Landscape	Face to face	yes	138.37	33.65	4.1
Bateman et al. (1994)	Wetland valuation	Face to face	yes	£140	£67	2.08
Bishop et al. (no date)	deer hunting	mail	yes	$37	$32	1.16
Boyle et al. (1993)[32]	moose hunting (expost)	mail	yes	775	709	1.09
"	moose hunting (exante)	mail	yes	701	484	1.45
Desvousges et al. (1992)	small oil spill effects	mall intercept	yes	240	129	1.86
"	all oil spill effects	mall intercept	yes	354	81	4.37
Gilbert et al. (1991)[33]	Eastern wilderness	mail	no	10	7	1.47
Johnson et al. (1990)	river recreation	mail	yes	33	53	1.62
Kealy & Turner (1993)[34]	candy bar	classroom	no	0.65	0.58	1.12
"	acid rain reduction	classroom	no	18	8	2.20
Kristrom (1990, 1993)[35]	forest preservation	mail	yes	395	202	1.96
Loomis et al. (1993)[36]	forest preservation	mail	no	224	100	2.24
Seller et al. (1985)	lake recreation	mail	yes	42	9	4.78

[32] The ex post sample hunted moose earlier that year and valued that past experience; the exante applied for but did not receive a permit to hunt moose that year, and thus valued a hypothetical hunting experience.

[33] This study reported 2 comparisons, one of which is presented here.

[34] Median, rather than mean, WTP.

[35] The dichotomous choice mean is from Kristom's (1990) Table 3 based on the Bishop and Heberlein (1979) method. The open-ended mean is from Kristrom's (1993) Table 1 for sample B. SEK were converted to U.S. dollars by dividing by 6, as suggested by Kristrom (1990).

[36] This study reported 3 comparisons, one of which is presented here.

supposed economic pay-off.

Given the need for contingent valuation studies to err on the conservative side, the promotion of the referendum format by the NOAA panel seems somewhat questionable. To the extent that a specific distribution of bids helps to maintain conservative bias (and abstracting from the influence of the distributional assumption), the bid vector in this study seems to have dampened the response function by the combination of few respondents assigned to low amounts and relatively more to each of the higher values. The observed response function in figure 4 shows that the erratic proportions at lower values have been important in weighting the predicted function. Erratic proportions at high bid values (possible if relatively few individuals are assigned the high amounts), would be more damaging.

4.14 Outliers and truncation issues

The disproportionate influence of extreme bid values on mean estimation can be assessed by simply dropping values and re-estimating the model. The impression given by figure 4, is that the tail of the density may be estimated around $R200, and that yea-saying may be behind responses observed beyond these values. To check the effect of the higher amounts, the two highest values (corresponding roughly to the 80th percentile), were sequentially dropped and the single (bid) logit model re-estimated. Interestingly, the omission of the highest value actually increases the restricted mean by around 3 %. Essentially there is information preventing the function approaching zero as fast, and the function in is now predicting a slightly fatter tail, figure 11. This indicates that the original placement does have some value in identifying the tail[37]. The further omission of 461 reduces the mean, although not significantly from the original estimate. This result seems to suggest that the bid vector is not as reckless as first appears. The result is also contrary to the findings of Cooper and Loomis (1992) regarding the disproportionate influence of the tails, and backs up the assertion of Kanninen and Kristrom (1993) that the finding by Cooper and Loomis is essentially an artefact of an inappropriate model rather than the vector. Thus, if the data appear to be drawn from an underlying logistic distribution, the removal of particular tail bid values does not have much effect on the estimated mean. The message gives some credence to the use of any prior information from the open-ended bid distribution. Similarly, it stresses the importance of making the most of prior beliefs about the location of the mean and locating bids accordingly. Similarly it seems worthwhile placing some bids in the tails. However fitting an appropriate model to the data is also important. In this regard the

[37]This result contrasts with the finding of Desvousges *et al* removing the highest bid ($1000) from a six-bid structure ($10, $25, $50, $100, $250, $1000) reduced the mean WTP by between 46% for one version and 71% for the other.

type of simple truncation exercise proves a useful diagnostic.

4.15 Aggregation

For the purpose of aggregation, the extent of the market is already pre-determined by the decision to restrict the survey to a sample of visiting anglers. This decision essentially violates the condition for probability sampling (giving each individual a known probability of being selected), and has inherent limitations for aggregation and prediction. In the former case, this means that the appropriate population for aggregation are the number of anglers registering fish catches in any one year in the Pantanal. This most likely under-estimates the social value, particularly when the resource has a high non use value and where high subsistence use.

Another drawback of the restricted sample frame is that the model is essentially calibrated with parameters drawn from a population which in all likelihood is unrepresentative of the majority of resource users. In other words, the model cannot be used to predict the willingness to pay for individuals other than those in the sample frame. Where the latter is derived from optimal probability sampling there is no problem in aggregation. A problem occurs in the event that resource use is characterised by heterogeneous socioeconomic groups who cannot all be sampled using the same survey instrument. A compromise involves either the preferences of one high-profile subsample of users, or the design of alternative questionnaires with the potential pitfalls this might entail in reconciling information provision and cognition. This problem is likely to be accentuated in developing countries and has not received much attention in the literature.

The issue can also be related to recent contributions to the debate over the income (or WTP) elasticity of environment[38], and attempts to infer magnitudes from CV data (Kanninen and Kristrom (1992), Kristrom and Riera (1996)). If anything can be inferred from CV, it would seem to be more interesting to conduct experiments across countries sufficiently differentiated by income status, rather as is the case in the Kristrom and Riera study, a group of largely homogeneous European states.

Table 15 sets out the aggregated figures for the study. To be clear about what is being aggregated, recall that the choice of payment vehicle implies a one-off annual payment additional to the current licence price. Aggregate value is therefore only based on this surplus value.

[38]See Pearce (1980) for an early review of the debate.

4.16 Further design issues

Numerous problems were foreseen before and during the pretest and redesign process and several only retrospectively. For brevity, and consistent with the need to provide the decision maker with information on key assumptions made, these are summarised in table 16 which also offers an observation on the direction of any likely resulting bias. In the final instance, more extensive tests of the instrument and innovative use of split sample tests were constrained by funding[39] and time constraints. Furthermore in research terms, given the increasing number of CV studies now appearing, it is simply difficult to be innovative. It is difficult to be equivocal about the direction of the overall bias, although the limited sample frame implies a downward bias. On the other hand, since willingness to pay is undoubtedly constrained by ability to pay, total resource value elicited by CV should be sensitive to user income constraints. Income constraints were not a problem for the respondents in this study.

Table 15 Aggregation scenarios

Pantanal Mato Grosso do Sul (110,000 visitors p.a.)[1]		
mean	Aggregate value	95% C.I.
Open-ended	5,803600	3,8589900-7,742900
Double-bounded DC.	15,126100	13,388100-17,176500
Notes: 1. for lower bound figure for all Pantanal multiply by 2		

4.17 Conclusion

On the basis of similar evidence to that found in this case study, many CV applications have concluded on a note of optimism regarding the worth of the method. In as far as a link can be made, the results appear to be consistent with demand theory (price sensitivity of the bid curve and quantity price variables etc). Such conclusions are not misplaced, but the adequacy of these diagnostic assessments may require further qualification. A more fundamental question which cannot be so

[39]The cost for conducting this study amounted to around £2000.

easily validated - due to the absence of diagnostic measures on cognition - is what were respondents actually valuing? This concern is particularly germane to the issue of valuing species and ecosystems, but the answer might only be found with the aid of deeper psychological probing than is typically conducted by CV researchers. It is conceivable that such analysis would present some rigorous challenges to the standard theories of consumer choice.

The problem of sample frames (and subsequent aggregation) in CV studies presents a difficulty and has not been adequately addressed in the current survey which adopted a default 'representative' sample of recreational visitors. Appraisal of alternative sample frames leads to the conclusion that there are likely to be more unanticipated problems in applying CV in LDCs than was first thought.

In terms of the elicitation method, DC has become the method of choice for its desirable market-like properties. It is fair to say that the incentive compatibility of the DC format may be more than outweighed by many of the design problems inherent in the approach, in that apparently simple design criteria may invalidate an otherwise reasonable study. Design issues involve separating out the problems which are artifacts of the bid vector and subsequent modelling selection process, from those that can be traced to cognitive processes apparently causing the disparity caused by different attitudes towards open and closed-ended formats. The latter has not been reasonably explained in economics.

This study shows that there are many elements which liken the DC design process to a lottery. Essentially the bid vector based on poor open-ended information can be either too long or too short, and may locate the density at an inappropriate spot, thereby enforcing truncation bias. Optimal design has been -albeit inadvertently - helpful in defining a bid range. The range used was actually more conservative than that which would have been selected otherwise. For example without reference to an open-ended survey, Cooper and Loomis (op cit), select some extraordinarily long bid vectors (eg $5-1,200) for a hunting study. What this does is to confirm the finding by Kanninen (1995), that bid vectors do not have to be as wide as is thought and, for a single DC, should avoid missing the mean completely. However, the bid vector in this case does not seem to have been disastrous. Information from the optimal design literature suggest that bid placement was erroneous only in the sense that bids were wasted. If yea-saying were a problem in both response formats, then this would have exacerbated bid placement problems.

An important finding from the design literature which is partially confirmed here, is that the double-bounded format is preferred to the single-bounded because even with an extreme bid design and apparent yea saying in the second response, the second question has reduced the bias and seems to point to a more conservative estimate. The lower variance is also now a standard result. However the strategic issue has not been solved and the result from figure 9 is somewhat worrying. If the single response is preferred, a possible suggestion to circumvent the disproportionate effect of potential outliers is to adopt the type of bid sample boot-strapping procedure described in chapter three. In other words, a bootstrap to calculate a confidence interval using the WTP values themselves. Such a procedure gives every observation a probability of being excluded from the sample and provides an expected value less dependent on any outliers. Of course one problem which could arise in this procedure is that excluded observations may actually upset the fit of the model itself. A more accurate procedure would therefore include a model search procedure for the remaining observations after every draw on the WTP set. Computationally burdensome indeed!

In conclusion, it is farer to state that 'optimal' bid design is something of a misnomer. It makes sense to place bids according to the information sought and according to some notional probability distribution, and even according to what can be inferred from similar studies. Relatively simple rules such as log-linear spacing achieve this to some degree, and approximate the rules of thumb on placement. More sophisticated designs should make the most of pretest information. The main criticisms of optimal design is that most of it is ex post. There are no theoretical grounds for generalisation about the distribution of WTP for the multitude of goods valued in CV surveys. Furthermore, unlike chemical trials, most CV studies are expensive to conduct, and cannot go through a recursive procedure nor extended pretests to be sure of the distribution of bids is eventually 'optimal'. Apart from the pretest information, a lot of guesswork and judgement is still involved and even then there is no guarantee that other problems will not arise[40].

By extension an important weakness of the dichotomous choice format is that it only conceals the problem of extreme values that apparently dog the open-ended format. This is probably intentional, since the extreme bids remain a puzzle which many CV researchers would prefer to forget. Increasingly the trend seems to towards fixing a specific conservative bid range and the use of non parametric analysis (e.g. Carson *et al* 1994ab). The resulting arbitrary truncation at the highest bid

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[40]For example in the only other use of Cooper's optimal design algorithm in the UK, Macmillan *et al* (1996) found that 47% of respondents were willing to pay the highest bid of £396 per year for acid rain reduction.The predicted function was similar to that of fig.9. The dispatch of a revised upper amount of £798 was still accepted by 18% of respondents leaving a fat tail. The authors do not explain the distributional assumption they made for setting the bids, but one suspects that a lognormal distribution would have been appropriate.

combined with the absence of any distributional assumption that might (inconveniently?) dictate behaviour in the tails essentially rules out the problem of extreme values upsetting calculation of a reasonable mean. Use of common forms like the log-logistic or the lognormal necessitate the use of the median which may be inconsistent with the preferred social welfare objective. Given that so many studies have found WTP to be distributed lognormally the avoidance of these models may seem suspect.

From a non-statistical point of view and echoing a point made previously, the adoption of the DC format is questionable. Given that individuals are supposed to know their preferences, the decision variables and potential errors outweigh supposed incentive compatibility advantages. The findings in this chapter lend support for innovative designs based on the open-ended alternative.

Table 16 Diagnostic table for interpreting Pantanal CVM.

Problem	Issue	Direction of potential bias	Potential solution
What is being valued?	Uncertainty that damage communication devices are adequate	Uncertain	Focus groups, protocols and retrospective reports when testing instrument to ascertain that scenario being valued is consistent between investigator and respondent.
Ex ante study problem	Good being valued may be at odds with actual ex post environmental change	Uncertain	Investigator can only work off the best information on likely perturbation. Outcome of continued pollution and\or Hidorvia project both uncertain.
Bid design	The bid vector was inadequate (too long/short)	Up - a typical finding from open-ended/discrete choice comparisons. Also related to model choice.	Use open-ended format or a combined format which commences with an open-ended statement and follows-up with a discrete amount some percentage higher. Otherwise use a *non parametric estimator* with a conservative bid range. The latter
Model choice	Parametric versus non parametric DC models. Inappropriate choice of error distribution for RUM	Up	Test all parametric distributions prior to using an *non* parametric model. For conservative design stick with an *open-ended* format.
Embedding and Scoping	Vague description leads to some multiple or subset of the dimension of interest being valued (spatial or temporal)	Uncertain	As per 'what is being valued'. Is it possible to assess plausibility of values given available substitutes for the good? If possible induce scoping in a conservative direction or use of de-scoping question. Split sample testing for embedding in terms of both spatial and temporal provision. If good is too complex, may be impossible to avoid
Data preparation	Treatment of non response categories, whether or not to impute missing values for the dependent or independent variable	Discretion of analyst	Best to err on conservative side. Treat WTP non responses as nos. Evidence to suggest that statistical imputation procedure can improve models (see Whitehead 1994)
Aggregation	Sample selection problem limiting the population to be aggregated over. Problem is also likely to be enforced in LDCs where one instrument inappropriate	down	Identification of stakeholders and design of questionnaires accordingly
Overall study bias	All elements considered	Probably downward for all Pantanal	Study might be validated using other methods eg production function approach (see Lynne *et al* 1981, Mäler 1992)

Appendix 1 Survey Questionnaire

MINISTÉRIO DA AGRICULTURA DO ABASTECIMENTO E DA REFORMA AGRÁRIA

EMPRESA BRASILEIRA DE PESQUISA AGROPECUÁRIA - EMBRAPA
CENTRO DE PESQUISA AGROPECUÁRIA DO PANTANAL - CPAP

LEVANTAMENTO DE ALTERAÇÕES AMBIENTAIS NO PANTANAL - CONFIDENCIAL

NÚMERO DE SÉRIE: /_/_/_/_/-/_/

NÚMERO DA ENTREVISTA: /_/_/_/ - /_/_/

/_/ Completo
/_/ Incompleto
/_/ Não pagamento
/_/ Protesto

DATA: /__/__/1994

Hora do início da entrevista (em 24 horas) ___ : ___ horas.

INSTRUÇÕES AO ENTREVISTADOR:

1. Entrevistar somente uma pessoa individualmente, evitando que outras pessoas do mesmo grupo participem (mas se pode entrevistar mais de uma pessoa de cada grupo, desde que individualmente)
2. Marque as respostas claramente; anote seus comentários pessoais em caso de dúvida.
3. Normalmente você não deve entrevistar pessoas com menos de 18 anos de idade.
4. Se estiver entrevistando uma família, você deve procurar entrevistar o chefe da família.
5. **Tente entrevistar homens e mulheres, assinalando o sexo: 1. /_/ Masc. 2. /_/ Fem.**
6. Leia em voz alta o seguinte texto ao entrevistado:

INÍCIO DO QUESTIONÁRIO:

Alô, eu sou ___________ (NOME) da EMBRAPA. Nós estamos realizando uma pesquisa com as pessoas que visitam o Pantanal e eu agradeceria se você pudesse responder algumas questões. A informação que você irá fornecer permanecerá **estritamente confidencial** e será usada somente para análises estatísticas. **Eu não irei perguntar seu nome nem seu endereço particular.**

(Se a resposta for SIM, então continue).
(Se a resposta for NÃO, agradeça e retire-se polidamente).

Primeiro, eu gostaria de obter algumas informações básicas relacionadas a sua visita.

1. Você está de férias, trabalhando ou vive aqui?
   1. /__/De férias (Vá para a Questão 2)
   2. /__/ Trabalhando na região (Vá para a Questão 6)
   3. /__/ Vive aqui (Vá para a Questão 7-C)

2. Esta é sua primeira visita ao Pantanal?
   1. /__/ Sim (Vá para a Questão 3)
   2. /__/ Não (Vá para a Questão 6)

3. Pretende visitar novamente?
   1. /__/ Sim (Vá para a Questão 4)
   2. /__/ Não (Vá para a Questão 5)
   3. /__/ Não sei (Vá para a Questão 7-B)

4. Nos próximos 12 meses quantas vezes você pretende voltar? __________ vezes. (Agora vá para a Questão 7-B).

5. Porque você não pretende visitar novamente?

____

(Agora vá para a Questão 7-B).

6.A) Quantas vêzes você já visitou o Pantanal? _______ visitas.
B) E nos últimos 12 meses quantas visitas você fez ao Pantanal? (Inclua a visita atual como uma visita) __________ visitas. (Vá para a Questão 7-A).

7.A) Quantos dias você normalmente fica no Pantanal? ________ dias. (Vá para a Questão 7-C).

B) Quantos dias você ficou no Pantanal? __________dias.

C) Você já visitou algum dos seguintes municipios do Pantanal? (Leia um por um e assinale).
   1. /__/ Coxim (MS) 2. /__/ Taquari (MS) 3. /__/ Poconé (MT)
   4. /__/ Cuiabá (MT) 5. /__/ Nossa Senhora do Livramento (MT)

8. Quantas das pessoas em **seu grupo** hoje, incluindo você, tem:
a) 16 anos ou mais?______ b) Menos de 16 anos?______ c) Não estou com grupo

9. Quantas pessoas da **sua familia** que **não** estão com você aqui hoje tem:
a) 16 anos ou mais?__________ b) Menos de 16 anos? ______

10. Onde você mora? (Cidade/distrito e estado, **não o endêrêço**)

____

____

11. A que distância fica daqui? (quilômetros) __________ Km.

12. Como você chegou até o Pantanal?
   1. /__/ Via rodoviária (Vá para a Questão 13).
   2. /__/ Via aérea (Vá para a Questão 16a)
   3. /__/ Vive no Pantanal (Vá para a Questão 16a)

13. Você veio diretamente de sua cidade ao Pantanal ou passou em outros lugares antes de chegar até aqui? (Por quaisquer razões: passear, conhecer, pescar, reunir grupo de amigos, etc).

1. /__/ Vim direto ao Pantanal. (Vá para a Questão 16a)
2. /__/ Passei em outros lugares antes de chegar aqui. (Vá para a Questão 14)

14. Qual a última localidade em que você esteve antes de chegar até aqui?

____

____

15. A que distância fica daqui? (quilômetros) ____________ Km . (Vá para a Questão 16b)

16. a) Aproximadamente quantas horas você viajou de sua cidade até aqui?________ h. (Vá para a Questão 17)

b)Aproximadamente quantas horas você viajou desta última localidade até aqui? ________ h.
(Vá para a Questão 17)

FALE (OU LEIA) EM VOZ ALTA E CLARA O SEGUINTE:

Eu gostaria agora de fazer-lhe algumas perguntas mais específicas sobre o que você valoriza no Pantanal e o quanto você gastou aqui.

17. Agora eu vou lhe mostrar um Cartão com algumas razões para você ter vindo ao Pantanal e gostaria que você selecionasse a razão **principal**. Escolha somente **uma** das alternativas. (**Mostre o Cartão l**). CIRCUNDE O NÚMERO DA RESPOSTA.

a. Possibilidade de capturar peixes de grande tamanho	01
b. Possibilidade de capturar muitos peixes de qualquer tamanho	02
c. Possibilidade de capturar diferentes espécies de peixes	03
d. Proximidade e acessibilidade de onde você vive	04
e. Proximidade em relação a outras regiões de pesca	05
f. Possibilidade de ver animais	06
g. Qualidade do ambiente (beleza natural, não poluido)	07
h. Outros (por favor especifique) ________________________	08

18. Agora eu vou lhe mostrar um Cartão com os possíveis gastos que **você e seu grupo** de pesca podem ter tido nesta viagem de pesca. Gostaria de saber aproximadamente quanto **vocês** gastaram em cada um dos itens do Cartão. (Se não tiver gasto em algum dos ítens, coloque zero). (**Mostre o Cartão 2**). LEIA OS ITENS E ANOTE AS QUANTIAS DADAS PELO RESPONDENTE.
Quantia total gasta

a. Combustível para a viagem (veículo automotor)	R$_______________
b. Apetrechos e equipamentos de pesca	R$ _______________
c. Passagens aéreas (por pessoa)	R$ _______________
d. Isca e gelo	R$ _______________
e. Serviços de guias de pesca	R$ _______________
f. Aluguel de barco e/ou motor	R$ _______________

g. Gasolina e óleo para o motor do barco R$ ______________
h. Alimentação e bebida R$ ______________
i. Pagamento de todos os serviços em barco-hotel R$ ______________
j.Outros____________________________ R$ ______________

ATENÇÃO ENTREVISTADOR:

1. Se o entrevistado responder em quantidades (litros de combustível, número de iscas, etc), anote as quantidades no ítem correspondente, sem tentar calcular valores.
2. Se o entrevistado tiver dificuldades de fazer as estimativas por ítens, então peça-lhe que estime o TOTAL de gastos: R$ ____________________

19. A) Agora eu vou lhe mostrar um Cartão com algumas espécies de peixes que **você e seu grupo de pesca** podem ter capturado nesta viagem. (**Mostre o Cartão 3**). Por favor liste o número total e/ou sua melhor estimativa do peso total para cada uma das espécies. **Não considere peixes comprados.**

	Captura		Número	Peso Total
	Sim	Não		(Kg)
Pintado/cachara	______	______	________	________
Dourado	______	______	________	________
Jau	______	______	________	________
Pacu	______	______	________	________
Curimbatá	______	______	________	________
Piranha	______	______	________	________
Tucunaré	______	______	________	________
Piraputanga	______	______	________	________
Barbado	______	______	________	________
Outros	______	______	________	________

B) Você comprou peixes para levar? 1. /__/ Sim. Quantos quilos?_______ kg. 2. /__/ Não.

FALE (OU LEIA) EM VOZ ALTA E CLARA O SEGUINTE:

Agora eu gostaria de apresentar a você algumas informações relacionadas a possíveis mudanças nas condições ambientais do Pantanal.

DÊ AOS RESPONDENTES AS SEGUINTES INFORMAÇÕES:

Alterações ambientais, principalmente no Planalto adjacente, tem **influência negativa** na qualidade da água dos rios do Pantanal . **A qualidade da água influi diretamente na abundância dos animais e plantas** que vivem na água ou que de alguma forma usam a água como um recurso.

As principais **causas** das alterações ambientais no Pantanal tem origem no planalto adjacente e são:

1. A agropecuária, que provoca **assoreamento** dos rios (devido ao desmantamento e erosão) e lança **agroquímicos** nas águas do Pantanal.
2. A mineração do ouro, que também **assoreia** os rios e lança **mercúrio** nas águas do Pantanal.

Dentro do próprio Pantanal também há alterações ambientais, causadas por **desmatamento** (para introdução de pastagens cultivadas) e pela **construção civil** (barragens, díques e estradas).

Agora eu vou lhe mostrar um cartão que descreve os **danos que tem ocorrido** em algumas áreas do Pantanal em decorrência dessas alterações ambientais, e os **danos esperados no futuro** se nada for feito para controlar suas causas. Gostaria que você lesse com atenção. **(Mostre o Cartão com o Cenário e dê tempo para o respondente ler tudo).** DEIXE O CARTÃO EM FRENTE AO RESPONDENTE.

APÓS O RESPONDENTE TER LIDO TUDO, EXPLIQUE:

Os danos dessas alterações nessas regiões estão no **Estágio B.** Se nenhum controle for feito, até 2010 poderá se atingir o **Estágio C.** Utilizando **tecnologias** para reduzir estes danos, os cientistas esperam que o ambiente não se degrade até o **Estágio C.**

As regiões onde estas alterações ocorrem **com maior evidência** são mostradas neste mapa, nos círculos verdes (MOSTRE O MAPA DO PANTANAL), mas elas poderão vir a ocorrer também em **muitos outros rios do Pantanal.** DEIXE O MAPA EM FRENTE AO RESPONDENTE

20. Você já conhecia alguma coisa sobre os problemas de poluição por mercúrio ou assoreamento no Pantanal?

1. /_/ Sim 2. /_/ Não

Se já conhecia, de que fontes obteve este conhecimento?

____

____

CONTINUE FORNECENDO AS SEGUINTES INFORMAÇÕES (FALE OU LEIA EM VOZ ALTA E CLARA):

A implementação de ações de controle implicam em custos elevados, pois as tecnologias de recuperação do meio ambiente são bastante caras. Um certo nível de qualidade da água é atualmente mantido no Brasil por receitas de impostos. Os usuários como você também pagam diretamente por benfeitorias, serviços e facilidades (públicas e privadas) existentes nos locais de lazer e recreação.

Entretanto, em certas áreas como o Pantanal, as receitas provenientes desses pagamentos podem ser insuficientes para garantir a manutenção da qualidade da água. Assim, é necessário mais dinheiro para manter as condições gerais de qualidade e evitar que a degradação continue.

Um fator importante para levar o governo a gastar mais dinheiro na manutenção da qualidade da água é verificar o quanto os visitantes valorizam um ambiente com as características do Pantanal e as atividades de lazer que se podem realizar nele.

Para ter uma idéia deste valor nós estamos fazendo algumas perguntas às pessoas sobre o quanto elas estariam dispostas a pagar para assegurar que as atuais condições de qualidade da água dos rios do Pantanal sejam mantidas.

Ao responder a este tipo de pergunta, por favor, tenha em mente as seguintes considerações:

1. Será assegurado que todas as receitas provenientes desses pagamentos serão usadas
exclusivamente para manter a qualidade da água do Pantanal.
2. Com essas receitas será possível manter as condições de qualidade da água (e em consequência as oportunidades de lazer) em seus níveis atuais (**Estágio B**).
3. Se não forem obtidos recursos suficientes através desses pagamentos, a qualidade da água
certamente atingirá o **Estágio C.**
4. Você deve basear suas respostas nos tipos de lazer que você faz hoje e naqueles que você poderá fazer no futuro.
5. Se você estabelecer uma quantia como pagamento para assegurar a qualidade da água, esta quantia irá sair de seu orçamento familiar, e portanto, você não poderá usá-la para outras atividades.
6. Todos os usuários dos rios (inclusive as operadoras de barcos), irão pagar a mesma quantia
através de diferentes formas de pagamento. Estes pagamentos serão válidos para as atividades de lazer desenvolvidas somente no Estado do Mato Grosso do Sul.

Você entendeu bem estas considerações? Gostaria que eu repetisse alguma?

21. Tendo em mente estas considerações, você está disposto a pagar alguma soma em dinheiro para manter a qualidade da água dos rios do Pantanal?

1. /__/	Sim	(Continue com o texto abaixo)
2. /__/	Não	(Vá para a Questão 26)
3. /__/	Não sei	(Continue com o texto abaixo)

Se houver um aumento no preço da licença de pesca, que hoje custa R$ 34,00, EXCLUSIVAMENTE com a finalidade de obter receitas para investir em um sistema de controle da qualidade da água, eu gostaria de saber qual é a quantia máxima que você estaria disposto a pagar para a implementação deste sistema.

22. Qual é o aumento **máximo** que você poderia pagar pela licença? Por favor estabeleça qualquer quantia que você pensa que é apropriada.
REGISTRE A QUANTIDADE : R$ ______________
(Se o valor escolhido for R$ = 0, vá para a Questão 26).

23. Se esta quantia não for suficiente para assegurar a manutenção da qualidade da água do Pantanal você pagaria mais (qualquer valor a mais)?

1. /__/	Sim	(Vá para a Questão 24)
2. /__/	Não	(Vá para a Questão 25)
3. /__/	Não sei	(Vá para a Questão 25)

24. Qual é o valor máximo **adicional** (isto é, **além** do valor dado em sua resposta anterior - Questão 22) que você estaria disposto a pagar? R$
____

25. O que você quis dizer quando respondeu que pagaria R$ ______ (VALOR DADO NA QUESTÃO 22)? Este valor que você estabeleceu é sua verdadeira disposição a pagar?

____

____

(Vá para a Questão 27)

26. Se o valor escolhido for R$ = 0, porque?
    1. /__/ Não tenho recursos para pagar, mas gostaria.
    2. /__/ O valor para mim é zero. Porque? ______________
    3. /__/ Não me importo com a poluição da água.
    4. /__/ Não quis estabelecer um valor . Porque? ______________
    5. /__/ Penso que isto é responsabilidade de outros: governo, etc.
    6. /__/ Já pago muitos impostos atualmente, etc.
    7. /__/ Não respondeu.

Finalmente, eu gostaria de saber alguns detalhes que nos permitam caracterizar sua família. Isto é necessário para assegurar que, ao final de nossa pesquisa, nós tenhamos entrevistado uma parcela ampla e heterogênea da população.

27. Agora eu vou lhe mostrar um cartão com diferentes grupos de renda. Você poderia indicar em qual dos grupos deste quadro sua **renda familiar mensal total** se enquadra? **(Mostre o Cartão 5)**.

INFORME QUE É ANÔNIMO E REGISTRE O CÓDIGO DO NÍVEL DE RENDA: ______

28. Em que ano você nasceu? _________
29. Qual o seu nível de educação formal? **(Mostre o Cartão 6)**.

1. /__/ Nunca estive na escola	5. /__/ Científico
2. /__/ Algum grau escolar	6. /__/ Graduação (faculdade)
3. /__/ Primário	7. /__/ Pós-graduação
4. /__/ Ginásio	**Profissão**: ____________________

30. Por último, você é membro de alguma organização para a conservação da natureza?
    1. /__/ Sim.   2. /__/Não

    Se SIM, qual (quais)? ______________________________

OBRIGADO POR SUA AJUDA E ATENCÃO!

Hora do fim da entrevista: ___ : ___ horas.

Double bounded format

Você entendeu bem estas considerações? Gostaria que eu repetisse alguma?

21. Tendo em mente estas considerações, você está disposto a pagar alguma soma em dinheiro para manter a qualidade da água dos rios do Pantanal?

1. /__/ Sim (Continue com o texto abaixo)
2. /__/ Não (Vá para a Questão 26)
3. /__/ Não sei (Continue com o texto abaixo)

Uma estimativa **conservadora** dos custos necessários para a utilização de uma tecnologia despoluidora, capaz de manter o nível de qualidade da água do Pantanal no **Estágio B**, sugere a necessidade de um incremento no preço da licença de pesca (que hoje custa R$ 34,00) de

R$ ___________

22. Você estaria disposto a pagar este valor **adicional** pela licença de pesca? (**acima** do valor atual de R$ 34,00)? 1. /__/ Sim (Vá para a Questão 23.A)
2. /__/ Não (Vá para a Questão 23.B)
3. /__/ Não sei (Vá para a Questão 23.B)

23.A) Nós ainda não sabemos exatamente quanto será necessário investir para manter o nível de qualidade da água do Pantanal. Se o custo **final** estimado para a utilização da tecnologia despoluidora mostrar que será necessário **R$** __________ **(O DOBRO DO VALOR ACIMA)** você estaria disposto a pagar este valor **adicional** pela licença de pesca (**acima** do valor atual de R$ 34,00)? 1. /__/ Sim (Vá para a Questão 24)
2. /__/ Não (Vá para a Questão 24)
3. /__/ Não sei (Vá para a Questão 24)

23.B) Nós ainda não sabemos exatamente quanto será necessário investir para manter o nível de qualidade da água do Pantanal. Se o custo **final** estimado para a utilização da tecnologia despoluidora mostrar que será necessário **R$** ___________ **(A METADE DO VALOR ACIMA)** você estaria disposto a pagar este valor **adicional** pela licença de pesca (**acima** do valor atual de R$ 34,00)? 1. /__/ Sim (Vá para a Questão 24)
2. /__/ Não (Vá para a Questão 24)
3. /__/ Não sei (Vá para a Questão 24)

24. Qual é o valor máximo **adicional** (i.é., **acima** do preço atual de R$ 34,00) que você estaria disposto a pagar pela licença de pesca? R$ _________________

Appendix 2: Survey Show cards

CARTÃO 1

Da lista abaixo por favor selecione a razão *principal* para você ter vindo aqui hoje. Escolha somente *uma* das alternativas.

Possibilidade de capturar peixes de grande tamanho	01
Possibilidade de capturar muitos peixes de qualquer tamanho	02
Possibilidade de capturar diferentes espécies de peixes	03
Proximidade e acessibilidade de onde você vive	04
Proximidade em relação a outras regiões de pesca	05
Possibilidade de ver outros animais	06
Qualidade do ambiente (beleza natural, não poluido)	07
Outros (por favor especifique) ________________	08

CARTÃO 2

Aproximadamente quanto *você e seu grupo de pesca* gastaram em cada um dos seguintes itens nesta viagem de pesca? (Se você não tiver gasto em algum dos ítens, coloque zero).

Quantia total gasta

a.	Combustível para viagem (veículo automotor)	R$________
b.	Apetrechos e equipamentos de pesca	R$________
c.	Passagens aéreas (por pessoa)	R$________
d.	Isca e gelo	R$________
e.	Serviços de guias de pesca	R$________
f.	Aluguel de barco e/ou motor	R$________
g.	Gasolina e óleo para o motor do barco	R$________
h.	Alimentação e bebida	R$________
i.	Pagamento total dos serviços em barco-hotel	R$________
j.	Outros____________________________	R$________

CARTÃO 3

Usando a seguinte tabela, por favor liste o número total e/ou sua melhor estimativa do peso total para cada uma das espécies de peixes *capturadas* por você e seu grupo de pesca nesta viagem de pesca.

	Captura		Número	Peso Total
	Sim	Não		Kg
Pintado/cachara	___	___	______	________
Dourado	___	___	______	________
Jau	___	___	______	________
Pacu	___	___	______	________
Curimbatá	___	___	______	________
Piranha	___	___	______	________
Tucunaré	___	___	______	________
Piraputanga	___	___	______	________
Outros	___	___	______	________

CARTÃO 4

Por favor escolha qualquer quantia do cartão que você pensa que é adequada ou indique qualquer outra.

Em R$:

0

0,50 1,00 3,00 5,00

10,00 20,00 40,00 60,00

80,00 100,00 300,00 500,00

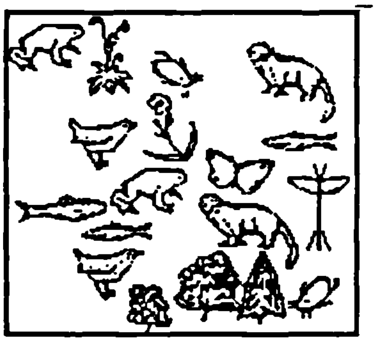
700,00 1000,00 2500,00 4000,00

Card 5

Condições Ambientais no Pantanal

ESTÁGIO A - 1970

A riqueza e a abundância da fauna e da flora indicavam um ecossistema sem perturbações ambientais significativas.



ESTÁGIO B - HOJE

Mundanças ambientais decorrentes de atividades humanas tem provocado alterações na qualidade da água dos rios do Pantanal, comprometendo o equilíbrio do ecossistema e reduzindo as populações de várias espécies de animais e plantas.

ESTÁGIO C - 2010

Os reflexos negativos da degradação do ambiente estarão conduzindo a uma rápida e contínua redução das espécies mais sensíveis, ameaçando-as de extinção.

CARTÃO 5

Você poderia indicar em qual dos grupos de renda deste quadro, sua *renda familiar mensal total* se enquadra (incluindo qualquer benefício/pensão do governo, rendimentos de investimentos, etc, de modo a incluir *toda* a sua renda antes do imposto de renda).

Em R$

a. Menor que 1.000,00
b. De 1.001,00 a 1.500,00
c. De 1.501,00 a 2.000,00
d. De 2.001,00 a 2.500,00
e. De 2.501,00 a 3.000,00
f. De 3.001,00 a 3.500,00
g. De 3.501,00 a 4.000,00
h. De 4.001,00 a 4.500,00
i. De 4.501,00 a 5.000,00
j. De 5.001,00 a 5.500,00
l. De 5.501,00 a 6.000,00
m. De 6.001,00 a 6.500,00
n. De 6.501,00 a 7.000,00
o. De 7.001,00 a 7.500,00
p. De 7.501,00 a 8.000,00
q. De 8.001,00 a 8.500,00
r. De 8.501,00 a 9.000,00
s. De 9.001,00 a 9.500,00
t. De 9.501,00 a 10.000,00
u. Maior que 10.000,00

CARTÃO 6

Por favor indique entre os ítens abaixo o seu nível de educação formal (curso completado integralmente).

1. Nunca estive na escola
2. Algum grau escolar
3. Primário
4. Ginásio
5. Científico
6. Graduação (faculdade)
7. Pós-graduação

Appendix 1a Survey Questionnaire

MINISTRY OF AGRICULTURE AND AGRARIAN REFORM

EMPRESA BRASILEIRA DE PESQUISA AGROPECUÁRIA - EMBRAPA
CENTRO DE PESQUISA AGROPECUÁRIA DO PANTANAL - CPAP

SURVEY ON ENVIRONMENTAL CHANGE IN THE PANTANAL - CONFIDENTIAL

SERIES NUMBER: /_/_/_/_/-/_/

INTERVIEW NUMBER: /_/_/_/ - /_/_/

DATE: /__/__/1994

/_/ Complete
/_/ Incomplete
/_/ No payment
/_/ Protest

Time at start of interview (24 hours) __ __ : __ __ **hours.**

INSTRUCTIONS TO INTERVIEWER:

1. Interview one person at a time while avoiding the participation of other group members if possible (do interview several group members individually if possible).
2. Mark all responses clearly, taking note of any additional comments if necessary.
3. You should only interview individuals over 18 years old.
4. If you are interviewing a family, try to speak to the head of household.
   5. Try to approach male and female respondents equally, noting the sex: 1. /_/ Masc. 2. /_/ Fem.
6. Read aloud the following text :

START OF THE QUESTIONNAIRE:

Hello, I'm ___________ (Name) from EMBRAPA. We are conducting a survey among visitors to the Pantanal and I would be grateful if you could respond to some questions. The information that you provide will remain strictly confidential and will only be used for statistical purposes. I will not be asking for your name or your private address.

(If the response is yes then continue).

(If the response is no then withdraw politely).

First, I'd like to ask you some basic information in relation to your visit.

1. Are you on holiday, working or do you live here?
   1. /__/ Holiday (Go to Question 2)
   2. /__/ Working in the region (Go to Question 6)
   3. /__/ Live here (Go to Question 7-C)

2. Is this your first visit to the Pantanal?
   1. /__/ Yes (Go to Question 3)
   2. /__/ No (Go to Question 6)
3. Do you intend visiting again?
   1. /__/ Yes (Go to Question 4)
   2. /__/ No (Go to Question 5)
   3. /__/ Don't know (Go to Question 7-B)

4. How many times do you intend visiting in the next 12 months? ___________ times .
   (Now go to Question 7-B).
5. Why do you not intend coming back?

____

(Now go to Question 7-B).

6.A) How many visits have you already made to the Pantanal? ________ visits.
B) In the last 12 months how many times have you visited the Pantanal? (Including this visit as one visit)___________ visits. (Go to Question 7-A).

7.A) How many days do you typically stay in the Pantanal? _________ days. (Go to Question 7-C).
B) How many days have you stayed in the Pantanal? ___________days.
C) Have you already visited other areas in the Pantanal? (Read the following list and mark).

1. /__/ Coxim (MS) 2. /__/ Taquari (MS) 3. /__/ Poconé (MT)
4. /__/ Cuiabá (MT) 5. /__/ Nossa Senhora do Livramento (MT)

8. How many people in **your group are**:
   a) 16 or above ?_______ b) Less than 16 years old?_______ c) Not in a group

9. How many memebers of **your family not with you today** are:
   a) 16 or above?___________ b) Less than 16 ? _______

10. Where do you live? (Town/district and state, **not address**)

____

____

11. How far is that from here ___________ Km.

12. How did you get to the Pantanal?
   1. /__/ Road (Go to Question 13).
   2. /__/ Air (Go to Question 16a)
   3. /__/ Live in the Pantanal (Go to Question 16a)

13. Did you come directly to the Pantanal or did you come via other towns to get here? (For whatever reason: tourism, fishing, meeting friends etc).

1. /__/ Came directly to the Pantanal. (Go to Question 16a)
2. /__/ Came here via other places (Go to Question 14)

14. Where was the last place you visited before you got here?

____

____

15. How far is that from here ____________ Km (Go to Question 16b)

16. a) Approximately how long did it take you to get here from your place of residence?________ h.
(Go to Question 17)

b) Approximately how long have you been travelling from your last stop-off point?
________ h.
(Go to Question 17)

READ THE FOLLOWING ALOUD:

Now I'm going to ask you some more specific questions about the things you value about the Pantanal and about what you spent here.

17. Now I'm going to show you a card with some of the reasons you might have for visiting the Pantanal and I'd like you to choose the main reason for coming. Select only one of the options. (SHOW THE CARD AND MARK THE RESPONSE).

a. Possibility of catching large fish	01
b. Possibility of catching a lot of fish of any size	02
c. Possibility of catching different species of fish	03
d. Proximity and access from place of residence	04
e. Proximity in relation to other fishing locations	05
f. Possibility of seeing animals	06
g. Quality of the environment (natural beauty and lack of pollution)	07 h.
Other (please state) ______________________________	08

18. Now I'm going to show you a card detailing some of the expenditures you or your group may have made as part of this trip. I'd like to know approximately how much you spent on each item on the card (if you spent nothing on an item please mark zero) (SHOW CARD 2) READ THE ITEMS AND NOTE THE AMOUNTS STATED BY THE RESPONDENT.

Total expenditure

a. Fuel for the journey (car) R$_______________

b. Tackle and fishing gear R$ _______________

c. Air tickets (per person) R$ _______________

d. Bait and ice R$ _______________

e. Service of a guide R$ ____________
f. Hire of boat and/or motor R$ ____________
g. Gasoline and oil for the motor R$ ____________
h. Food and drink R$ ____________
i. Payment of other serives including boat-hotel R$ ____________
j. Others ____________________ R$ ____________

interviewer to note:

1. If the respondent indicates quantities (fuel, bait, etc) note these without trying to ascertain values
2. If the respondent has difficulty estimating individual expenditures then request the total expenditure R$ ________________

19. A) Now I'm going to show you a card with some of the species of fish that you or your group may have caught on this trip (**Show Card 3**). Please list the total number and approximate weight for each species. Do not include any fish you have bought.

	Caught Yes	No	Númber	Total weight (Kg)
Pintado/cachara	______	______	______	______
Dourado	______	______	______	______
Jau	______	______	______	______
Pacu	______	______	______	______
Curimbatá	______	______	______	______
Piranha	______	______	______	______
Tucunaré	______	______	______	______
Piraputanga	______	______	______	______
Barbado	______	______	______	______
Other	______	______	______	______

B) Did you buy any fish to take away? 1. /__/ Yes. How many kilos?______ kg. 2. /__/ No.

READ THE FOLLOWING ALOUD AND CLEARLY:

Now I'd like to present some information related to possible environmental changes in the Pantanal.

GIVE THE RESPONDENT THE FOLLOWING INFORMATION:

Environmental changes mainly in the adjacent plateau have negative influences on Pantanal river quality. River quality directly influences the abundance of plants and animals living in and depending on riverine habitats.

The main causes of environmental change in the Pantanal (or surrounding plateau) are:

1. Cattle ranching, which causes river sedimentation (when areas are cleared for pasture) and the discharge of agrochemicals in rivers.
2. Gold mining, which also leads to sedimentation and the discharge of mercury as a result of production processes.

Inside the Pantanal itself, there are other environmental changes resulting from deforestation (for the introduction of pastures, arable crops, and for civil constructions such as dykes, dams and roads)

Now I'm going to show you a card detailing the form of damages that have occured in some areas of the Pantanal as a result of these environmental changes and the damage that might be expected in future if nothing is done to contain further change. I'd like you to read carefully the description provided **(SHOW SCENARIO CARD AND GIVE THE RESPONDENT TIME TO READ THE DETAILS** - LEAVE THE CARD IN FRONT OF THE RESPONDENT.)

AFTER THE RESPONDENT HAS FINISHED READING, CONTINUE:

Damages resulting from these alterations can currently be put at **Stage B.** In the absence of appropriate control, damages of the scale represented in **Stage C** will be reached by 2010. With the use of pollution control technologies to combat these impacts, scientists hope to prevent the degradation to **Stage C**.

The areas where these changes are having the greatest impact are shown on (circled) on this map (SHOW MAP OF PANTANAL) . However, similar impacts may occur in many other rivers of the Pantanal (LEAVE MAP IN FRONT OF THE RESPONDENT).

20. Were you aware of the problems of mercury pollution and sedimentation in the Pantanal?

1. /_/ Yes 2. /_/ No

If yes, where did you learn about these problems? ________________________________

____

CONTINUE WITH THE FOLLOWING INFORMATION (READ ALOUD AND CLEARLY):

The implementation of pollution control measures will be quite costly. In Brazil, a certain level of water quality is currently assured by expenditure from tax revenues. Users like yourself also contribute by means of payments for the use of facilites (public and private) in recreation areas.

Nevertheless, in some areas of the Pantanal, the payments from these sources are insufficient to garantee water quality standards and it is necessary to raise further revenues to prevent further degradation.

An important factor in motivating the governement to allocate further resources to water quality is the assessment of the value visitors place on the Pantanal environment and on the leisure opportunities made available in a clean environment.

To have an idea of this value, we are asking people a few questions about how much they would be willing to pay to maintain water quality in the Pantanal.

Responding to this type of question please keep in mind the following considerations:

1. Any proceeds arising from such a payment would be used **exclusively for the maintenance of water qulity in the Pantanal.**
2. With sufficient receipts it should be possible to maintain water quality conditions (and therefore associated leisure opportunities) at their current level **(Stage B)**.
3. If sufficient resources are not forthcoming via these payments, water quality will almost certainly reach **Stage C.**
4. You should base your response on the types of leisure activities which you currently undertake and those that you contemplate in future.
5. When you state an amount, recall that the same amount will no longer be available as part of your family budget (and will therefore be unavailable for other activities).
6. All river users (including boat operators) will pay the same amount by means of different payment methods. These payments only relate to leisure activities which take place in the Pantanal areas of Mato Grosso do Sul state.

Have you fully understood these consideration? Would you like me to repeat any?

21. Bearing in mind these considerations, would you be willing to pay a sum of money for the maintenance of water quality in Pantanal rivers?
    1. /__/ Yes (Continue with the text below)
    2. /__/ No (Go to Question 26)
    3. /__/ Don't know (Continue with the text below)

Suppose there were a price increase in the fishing licence (which currently costs R$ 34,00) exclusively for the purpose of raising revenue for investment in water treatment. I'd like to know what is the maximum amount of money you would be willing to pay for such a treatment system.

22. What is the maximum licence price increase you would be willing to pay? Please state whichever sum you think appropriate.
    RECORD THE AMOUNT R$ _______________
    (If the stated value was R$ = 0, go to Question 26).

23. If this amount were not sufficient to maintain water quality in the Pantanal, would you pay more (how much more)?
    1. /__/ Yes (Go to Question 24)
    2. /__/ No (Go to Question 25)

3. /__/ Don't know (Go to Question 25)

24. What is the maximum amount additional (in other words, in addition to the amount you previously stated in Q.22) that you would be willing to pay? R$ ____________________

25. What exactly do you mean when you state that you are willing to pay R$ ______ (VALUE GIVEN IN QUESTION 22)? Is this value you stated your true willingness to pay?

____
____

(Go to Question 27)

26. If the value was R$ = 0, why?
   1. /__/ I do not have the money to pay more although I would like to.
   2. /__/ The value for me is zero. Why? ________________
   3. /__/ I don't care about water pollution.
   4. /__/ I can't state a value. Why? ________________
   5. /__/ I think this is the responsibility of others: government, etc.
   6. /__/ I already pay enough taxes.
   7. /__/ No response.

Finally I'd like to take a few details which will allow us to characterise your household. This information is necessary for us to make sure that our research has covered a representitive sample of the population.

27. Now I'm going to show a card describing various income brackets. Could you please tell me which of the groups best describes your total monthly family income. SHOW CARD 5

INFORM THE RESPONDENT THAT THIS INFORMATION IS CONFIDENTIAL AND RECORD THE GROUP: _____

28. What year were you born? _________
29. What level of formal education have you reached? (**Show Card 6**).

   1. /__/ Never went to school
   2. /__/ Some elementary grades
   3. /__/ Primary education
   4. /__/ High school
   5. /__/ Technical school
   6. /__/ Graduate (university)
   7. /__/ Post-graduate

   Profession: ________________________

30. Finally, are you a member of any nature conservation group?
   1. /__/ Yes. 2. /__/No

If yes, which one(s)? ______________________________

THANK YOU FOR YOUR HELP AND ATTENTION!

Time at end of interview: ___ : ___ .

CARD 1

From the following list, please chose the *main* reason for your visit here today. Select only one alternative.

Possibility of catching large fish	01
Possibility of catching a lot of fish of any size	02
Possibility of catching different species of fish	03
Proximity and accessibility from where you live	04
Proximity in relation to alternative fishing sites	05
Possibility of seeing other animals	06
Environmental quality (natural beauty and absence of pollution)	07
Other (please specify) ____________________	08

CARD 2

Approxiately how much did you and your group spend on each of the following items as part of this trip? (If you didn't spend anything on a particular category, please state zero).

Total expenditure

a. Fuel for travel (car)	R$________
b. Fishing tackle	R$________
c. Air tickets (per person)	R$________
d. Bait and ice	R$________
e. Services of a fishing guide	R$________
f. Boat or motor rental	R$________
g. Gasoline for motor boat	R$________
h. Food and drink	R$________
i. Total cost of boat-hotel stay	R$________
j. Other________________________________	R$________

CARD 3

Using the following table, please list the total number (or your best estimate of the weight) for each species of fish caught by yourself or your group during this trip

	Caught		Number	Total Weight
	Yes	No		Kg
Pintado/cachara	___	___	______	______
Dourado	___	___	______	______
Jau	___	___	______	______
Pacu	___	___	______	______
Curimbatá	___	___	______	______
Piranha	___	___	______	______
Tucunaré	___	___	______	______
Piraputanga	___	___	______	______
Outros	___	___	______	______

CARD 4

Please select the amount on the card which approximates your willingness to pay or state any other amount.

Em R$:

0

0,50	1,00	3,00	5,00
10,00	20,00	40,00	60,00
80,00	100,00	300,00	500,00
700,00	1000,00	2500,00	4000,00

CARD 5

Please indicate which of the following groups best approximates your total monthly family income from all sources (including state benefit/pensions/ investment incomes of all kinds) after tax.

Em R$

a. Below 1.000,00
b. From 1.001,00 to 1.500,00
c. From 1.501,00 to 2.000,00
d. From 2.001,00 to 2.500,00
e. From 2.501,00 to 3.000,00
f. From 3.001,00 to 3.500,00
g. From 3.501,00 to 4.000,00
h. From 4.001,00 to 4.500,00
i. From 4.501,00 to 5.000,00
j. From 5.001,00 to 5.500,00
l. From 5.501,00 to 6.000,00
m. From 6.001,00 to 6.500,00
n. From 6.501,00 to 7.000,00
o. From 7.001,00 to 7.500,00
p. From 7.501,00 to 8.000,00
q. From 8.001,00 to 8.500,00
r. From 8.501,00 to 9.000,00
s. From 9.001,00 to 9.500,00
t. From 9.501,00 to 10.000,00
u. Above 10.000,00

CARD 6

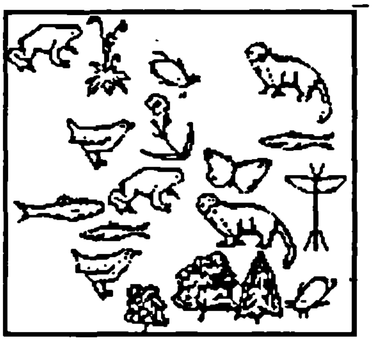
Please indicate which of the categories below best describes your level of formal education.

1. /__/ Never went to school
2. /__/ Some elementary grades
3. /__/ Primary education
4. /__/ High school
5. /__/ Technical school
6. /__/ Graduate (university)
7. /__/ Post-graduate

Environmental Conditions in the Pantanal

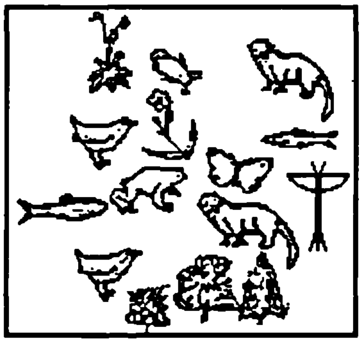
STAGE A - 1970

The richness of flora and fauna indicate a clean and healthy ecosystem without significant damage or perturbation.



STAGE B - TODAY

Environmental change resulting from human activity gives rise to alterations in water quality in Pantanal rivers. This endangers the equilibrium of the ecosystem by reducing the populations of various species of plants and animals.



STAGE C - 2010

The negative impacts of environmental degradation lead to a rapid and continuous decline in the populations of the more sensitive species which are now threatened with extinction.

APPENDIX 3

NOAA PANEL GUIDELINES[41]

General Guidelines

Sample Type and Size: Probability sampling is essential for a survey used for damage assessment.[42] The choice of sample specific design and size is a difficult, technical question that require the guidance of a professional sampling statistician.

Minimize Nonresponses: High nonresponse rates would make the survey results unreliable.

Personal Interview: The Panel believes it unlikely that reliable estimates of values could be elicited with mail surveys. Face-to-face interviews are usually preferable, although telephone interviews have some advantages in terms of cost and centralized supervision.

Pretesting for Interview Effects: An important respect in which CV surveys differ from actual referenda is the presence of an interviewer (except in the case of mail surveys). It is possible that interviewers contribute to "social desirability" bias, since preserving the environment is widely viewed as something positive. In order to test this possibility, major CV studies should incorporate experiments that assess interviewers effects.

Reporting: Every report of a CV study should make clear the definition of the population sampled, the sampling frame used, the sample size, the overall sample non-response rate and its components (e.g. refusals), and item non-response on all important questions. the report should also reproduce the exact wording and sequence of the questionnaire and of other communications to respondents (e.g., advance letters). All data from the study should be archived and made available to interested parties (see Carson et al. (1992), for date, however, the reports has not been available publicly and the data have not been archived for open use by other scholars).

[41] Federal Register (1993).

[42] This footnote comes directly from the Panel report. "This need not preclude use of less adequate samples, including quota or even conveniences samples, for preliminary testing of specific experimental variations, so long as order or magnitude differences rather than univariate results are the focus. Even then, obvious sources of bias should be avoided (e.g., college students are probably too different in age and education from the heterogeneous adult population to provide a trustworthy basis for wider generalization)". *Id.* at 4611.

Careful Pretesting of a CV Questionnaire: Respondents in a CV survey are ordinarily presented with a good deal of new and often technical information, well beyond what is typical in most surveys. This require very careful pilot work and pretesting, plus evidence from the final survey that respondents understood and accepted the main description and questioning reasonably well.

Guidelines for Value Elicitation Surveys

Conservative Design: Generally, when aspects of the survey design and the analysis of the responses are ambiguous, the option that tends to underestimate willingness to pay is preferred. An conservative design increases the reliability of the estimate by eliminating extreme responses that can enlarge estimated values wildly and implausibly.

Elicitation Format: The willingness to pay format should be used instead of the compensation required because the former is the conservative choice.

Referendum Format: The valuation question should be posed as a vote on a referendum.

Accurate Description of the Program or Policy: Adequate information must be provided to respondents about the environmental program that is offered. It must be defined in a way that is relevant to damage assessment.

Pretesting of Photographs: The effects of photographs on subjects must be carefully explored.

Reminder of Undamaged Substitute Commodities: Respondents must be reminded of substitute commodities, such as other comparable natural resources or the future state of the same natural resource. This reminder should be introduced forcefully and directly prior to the main valuation question to assure that respondents have the alternatives clearly in mind.

Adequate Time Lapse from the Accident: The survey must be conducted at a time sufficiently distant from the date of the environmental insult that respondents regard the scenario of complete restoration as plausible. Questions should be included to determine the state of subjects's beliefs restoration probabilities.

Temporal Averaging: Time dependent measurement noise should be reduced by averaging across independently drawn samples taken at different points in time. A clear an substantial time trend in the responses would cast doubt on the "reliability" of the finding.

"No-answer" Option: A "no-answer" option should be explicitly allowed in addition to the "yes" and "no" vote options on the main valuation (referendum) question. Respondents who choose the "no-answer" option should be asked nondirectively to explain their choice. Answers should be carefully coded to show the types of response, for example: (i) rough indifference between a yes and a no vote; (ii) inability to make a decision without more time or more information; (iii) preference for some other mechanism for making this decision; and (iv) bored by this survey and anxious to end it as quickly as possible.

Yes/no Follow-ups: Yes and no responses should be followed up by the open-ended question: "Why did you vote yes/no ?" Answers should be carefully coded to show the types of responses, for example: (i) It is (or isn't) worth it; (ii) Don't know; or (iii) The oil companies should pay.

Cross-tabulations: The survey should include a variety of other questions that help to interpret the responses to the primary valuation question. The final report should include summaries of willingness to pay broken down by these categories. Among the items that would be helpful in interpreting the responses are:

Income
Prior Knowledge of the Site
Prior Interest in the Site (Visitation Rates)
Attitudes Toward the Environment
Attitudes Toward Big Business
Distance to the Site
Understanding of the Task
Belief in the Scenarios
Ability/Willingness to Perform the Task

Checks on Understanding and Acceptance: The above guidelines must be satisfied without making the instrument so complex that it poses tasks that are beyond the ability or interest level of many participants.

Goals for Value Elicitation Surveys

Alternative Expenditure Possibilities: Respondents must be reminded that their willingness to pay for the environmental program in question would reduce their expenditures for private goods or other public goods. This reminder should be more than perfunctory, but less than overwhelming. The goal is to include respondents to keep in mind other likely expenditures, including those on other environmental goods, when evaluating the main scenario.

Deflection of Transaction Value: The survey should be designed to deflect the general "warm-glow" of giving or the dislike of "big business" away from the specific environmental program that is being evaluated. It is possible that the referendum format limits the "warm glow" effect, but until this is clear the survey design should explicitly address this problem.

Steady State or Interim Losses: It should be made apparent that respondents can distinguish interim from steady-state losses.

Present Value Calculations of Interim Losses: It should be demonstrated that, in revealing values, respondents are adequately sensitive to the timing of the restoration process.

Burden of Proof: Until such time as there is a set of reliable reference surveys, the burden of proof of reliability must rest on the surveys designers. They must show through pretesting or other experiments that their survey does not suffer from the problems that these guidelines are intended to avoid. Specifically, if a CV survey suffered from any of following maladies, we would judge its findings "unreliable":

- A high nonresponse rate to the entire survey instrument or to the valuation question.
- Inadequate responsiveness to the scope of the environmental insult.
- Lack of understanding of the task by the respondents.
- Lack of belief in the full restoration scenario.
- "Yes" or "no" votes on the hypothetical referendum that are not followed up or explained by making reference to the cost and/ or the value of the program.

Chapter 5

Valuing Biodiversity: Measuring the User Surplus of Kenyan Protected Areas

5.1 Introduction

There is a growing literature on the economics of wildlife and protected areas (Swanson and Barbier 1992; Shah 1995). Main issues include management costs and cost-benefit assessment (Willis 1989), opportunity costs of conservation (Norton-Griffiths and Southey 1995), and optimal pricing policy (Clarke and Ng 1993). Among the economic arguments for conservation few have been more emotive than the debate about direct use values (see for example Dobson and Poole 1992). In several countries bans and restrictions on direct uses hinder the economic case and, as described in the modified Clark model reviewed in chapter one, remove value from biological resources. In such cases the magnitude and capture of non market value is a salient issue. Nowhere is this question more urgent than in developing countries such as Kenya.

This chapter addresses the non-market dimension to conservation with a second application of the contingent valuation method, and a brief review of the role of the travel cost alternative. The initial aim of the study was to offer an input into game park pricing policy which provides the most direct method for resource owners to capture rent for distribution. The basic premise for conducting the analysis was that park entry fees in Kenya were (at the time the study was conducted)[1] peculiarly low (relative to overall travel costs) and that little was known about visitor price sensitivity. The plan of the chapter is as follows. First, some background to the study is provided and the rationale for the use of valuation methods for guiding pricing. Second, an application will provide the basis for informing pricing decisions in Kenyan parks.

Many of the methodological and empirical problems with CV have been addressed in the previous two chapters. It will be clear from the results that this study provided many lessons which have guided the experience of the previous chapter. Some of these will be discussed along with some of the recent findings on response motives. Some general conclusion on the use of valuation methods will also be offered.

5.2 Kenyan Protected Areas

The extent of Kenyan tourism and its central role to Kenyan development are summarised in Moran

[1] July-August 1993.

(1994). Covering 8% of the country's land area (Figure 1). - including areas of considerable agricultural potential - the domestic resource costs of maintaining a system of parks and reserves are high. Kenya Wildlife Service (KWS) is the parastatal agency with a difficult job to do. As much as 90% of KWS income derives from gate receipts, which in 1991 amounted to the equivalent of around US$8 million from 22 National Parks and Reserves (KWS accounts 1992). Significantly, KWS estimates receipt of as little as 3% of all tourism expenditures (KWS 1990); a meagre return. To put this in perspective, the current day permit at the time of the survey was pegged at US$14, which represents less than 1% of a standard 2 week safari-beach package from the U.K. In a competitive tourism market KWS pricing decisions are complicated by the role of intermediaries in the package holiday business. Bearing in mind the importance of the sector, it might be expected that the protected area network would be accorded some degree of security with sufficient funding to safeguard current and potential economic benefits. Yet park use is haphazard, and there is frequently little coincidence between those who benefit and those who pay for the continued existence of such areas. Parks and reserves are rarely self-sufficient ecosystems. The so-called dispersal areas and wildlife corridors across which wildlife forage and migrate often coincide with prime agricultural resources. The problem is acknowledged to be particularly acute around several popular tourist circuits such as Tsavo and Aberdares (KWS 1990), bringing wildlife into direct conflict with private farmers, pastoralists and communally owned group ranches. Range fencing is used only to a limited extent. Co-operation with adjacent landowners is therefore an on-going theme. Revenue sharing, which exists alongside an unsuccessful crop damage compensation scheme is part of KWS's advisory role outside parks and reserves. Payments are designed to compensate for attenuated land use rights of communities adjacent to parks and derive directly from visitor fees. The effectiveness of both schemes has been limited partly by funding constraints and partly by corruption. This experience has diminished many communities' stake and perception of wildlife and there is considerable cynicism about government priorities which appear to favour animals over people. The picture that emerges, and one aptly summarised by Wells (1992), is of the distribution of wildlife benefits being spatially skewed nationally (in favour of hotel and tour operators), and globally (foreign tourists). In other words, those in closest proximity bear disproportionate costs of conservation.

In recent years Kenya has been at the forefront of a largely symbolic anti-poaching campaign involving a shoot-to-kill policy and limits on trophy hunting. The latter restriction puts Kenya at a disadvantage relative to neighbouring Tanzania and confines returns to those derived from non-consumptive uses such as tourism. Furthermore, growing economic and demographic pressures (Table

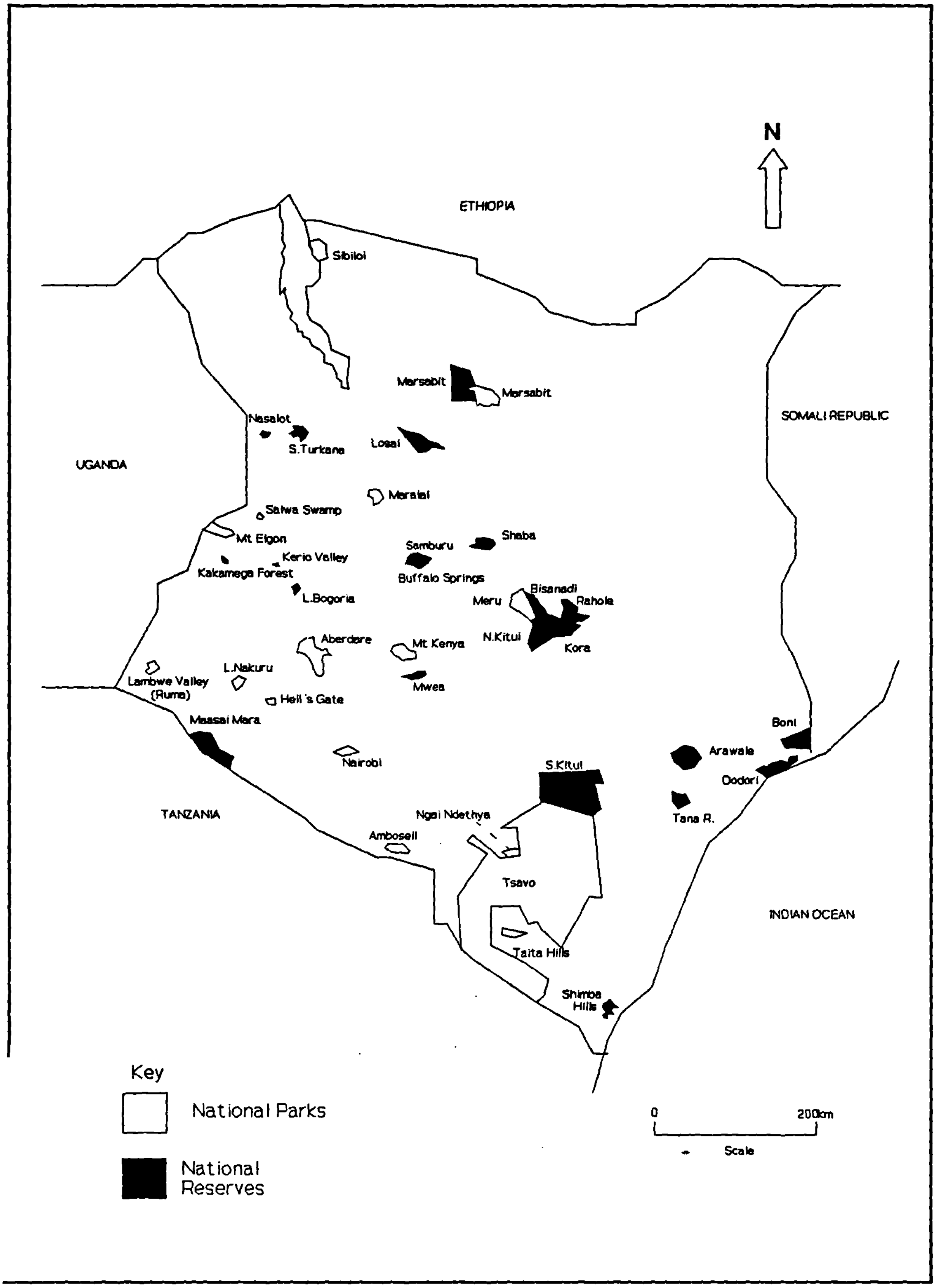


Figure 1: Kenyan National Parks and Reserves

1) now threaten to swamp protected areas. Alarming population growth is a real concern, while the promotion of settled husbandry and subdivision of communal properties into private holdings - both (mistakenly) regarded as indices of progress - complicate matters. The latter in particular, threatens the fragmentation and irreversible conversion of dispersal habitat. To counter such threats, KWS seeks to encourage direct participation in wildlife management by group ranches and private landowners. As yet, this is largely restricted to the promotion of tourist-related enterprise aimed at making this a financially attractive alternative to agricultural conversion. The financial viability of complementary game ranching plans is also the subject of current interest with around 16 successful operations and more than 50 licensed game farms (Byrne *et al.* 1993)[2].

These pressures re-emphasise the implicit subsidy currently paid by Kenyans to support conservation for the benefit of the world at large. As Norton-Griffiths and Southey (1995) show, the financial returns to game parks relative to wheat can provide a cogent case for the conversion of some protected areas. Demonstrating and appropriating full economic benefits of wildlife conservation has become a central tenet of wildlife economics. The maxim is most relevant where demographic pressures and external economic constraints mean that resources tied up in conservation are associated with an ever increasing opportunity cost. In the Kenyan case the economic comparison is particularly challenging and dependant on the capture of elusive global values for which formal markets have yet to fully develop It is suggested that an alternative and more equitable revenue source lies in significant untapped willingness to pay above existing fees paid by foreign visitors. Tourists may be the most accessible vehicle for realising the North-South transfers reflecting (and giving substance to) the "existence value" manifested in the ivory debate of the late 1980s.

It would seem reasonable to extract revenue from those most willing to pay and this proposition is put to the test using a valuation survey of foreign tourists. The issue of subsequent revenue disbursement has recently been the battleground of Kenyan politics. Current concern that some parks and reserves may be nearing their tourist threshold provides an additional rationale for the use of pricing as a rationing device. The long term ecological consequences of high visitor volumes remains a matter of debate. The findings of a survey conducted in the summer of 1993 lend some support to the view that Kenya is underpricing its most valuable national assets and foregoing considerable accessible economic benefits which might be ploughed back into management and revenue sharing

[2] The management distinction between farming and ranching is slight. Farming implies the domestication of wildlife whereas ranching simply implies the manipulation of herd structures towards a "best" use. Commonly used species include Gazelles, Wildebeest and Hartebeest, Impala and Oryx.

schemes (KWS 1990).

5.3 Contingent Valuation

Little is known about the extent of user and non-user consumer surplus associated with Kenyan parks and these can only be elicited by revealed preference (travel cost) or expressed preference (contingent valuation) techniques. This chapter reports data collection and estimation of mean WTP of a sample of foreign visitors to the Kenyan protected area network. The derived consumer surplus accrues mainly from the non-consumptive use by a sample of foreign visitors, and is contingent on the maintenance of the network in its current condition in the face of creeping degradation.

Table 1: Population Growth in Kenya

Census Year	Total Population
1948	5.4m
1962	8.6m
1969	10.9m
1979	15.3m
1989	22.0[1]m

Source: Central Bureau of Statistics, *Kenya Statistical Abstract*, various years. [1]Estimate

Surveying (by self-administered questionnaire) took place over 1 month at several sites selected to maximise the response rate[3] from foreign visitors. Sites were located both inside and outside protected areas, and it is anticipated that responses therefore pick up consumer surplus associated with use (where a respondent is actually in a park) and non use where a respondent is outside a park anticipating a visit. Prior to receiving a questionnaire, respondents were screened on their resident status (resident versus non-resident), reasons for visiting Kenya and their willingness to participate individually in a survey on their own experience of Kenyan parks. The screening process helped reduce non- response rates (incomplete questionnaires), although a statement of time necessary to complete the questionnaire lead to a high level of refusals to participate. The self-administered format is not ideal, although it has been used in several widely cited U.S studies (eg Desvousges *et al* 1992). The format is now widely recognized as relatively unreliable for allowing respondents to thoroughly address complex issue (Schuman 1995), and can suffer many of the same restrictions as mail surveys.

[3] Survey locations listed in Annex 1.

5.4 Questionnaire Design

Annex 1 contains one version of the four basic questionnaire versions randomly assigned (with French and German translations freely available). All began by eliciting basic preference information in a manner commonly employed to maximise responses for self-administered postal surveys (Dillman 1978). Next, country of origin and component travel cost information was gathered, plus questions on days spent in parks, parks visited, length of safari and days prior to questioning. Respondents were then asked to consider the costs of park management and the constraints binding on conservation decisions. The option of higher entrance fees was then suggested as a possible solution to finance conservation with an implication that quality would decline otherwise. Respondents were made aware that they had the option of alternative vacation choices and of the existence of competing game viewing alternatives which may or may not raise prices.

Survey versions differed in the contingent valuation question posed with 3 versions attempting to elicit dichotomous choice and open-ended responses (Table 2). The dichotomous choice format was double bounded with higher or lower follow-up offers in response to an initial offer. In framing WTP questions, one difficult issue concerned choice of payment vehicle. Numerous studies have identified a vehicle bias where responses are shown to represent protests against the mechanism of payment rather than a refusal to value the good on offer (Mitchell and Carson, 1989). Typically this is most acute in the case of tax increases or new fees levied to finance hitherto free goods. In the preamble to the CV question fees were suggested, and initial probing revealed that most visitors found this an equitable way for foreigners to contribute to Kenyan conservation[4]. However a pre-test revealed that few visitors had any idea about the current fee structure. Prompted to specify their WTP in any format, many hesitated, suspecting (incorrectly) excessively high fees inclusive in the cost of their overall package. The high number of package tour visitors therefore ruled out asking WTP questions based on hypothetical fee changes. The final question format used, framed dichotomous WTP in terms of a percentage increase on the individual's overall tour cost. Most visitors had a clear idea of their overall trip cost and were not averse to expressing their WTP in these terms.

Finally, all questionnaire versions asked for basic socio-economic information; income, age, sex, member of conservation group, education, as well as information on how respondents thought higher fees should be charged and (in the case of the open-ended format), reasons to validate zero bids[5].

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[4] These discussions also provoked many unprompted comments regarding corruption and the ultimate beneficiaries of fee revenues.

[5] To separate true zero value statements from protest bids.

Table 2: Survey formats

Survey Category	Travel Cost	Dichotomous Choice WTP bidding format		Open-ended WTP
		1st	2nd	
A1	+	WTP 10% ?	YES → WTP 15% YES NO NO → WTP 5% YES NO	+
A2	+	WTP 15% ?	YES → WTP 20% YES NO NO → WTP 10% YES NO	+
A3	+	WTP 5% ?	YES → WTP 10% YES NO NO → WTP 2% YES NO	+
B	+			+

5.5 Data analysis

Analysis concentrates on the DC format, and a total of 311 usable responses were available from respondents were presented with an initial take it or leave it percentage 5%, 10% or 15% increase (in overall cost). Converted to an absolute amount and divided by the stated number of days on safari, provided a continuous dollar bid range which may be interpreted as a daily WTP fee equivalent[6].

[6]Recalling the findings from chapter four, this bid vector is far from ideal. This is because the vector presents a unique bid to every individual. The risk is that the resulting proportions may be all zero or one anywhere in the bid range. The survey design for this study was outside the control of

Rejection or acceptance to pay this sum (coded 0 or 1 respectively), provided a binary dependent variable to be modelled in respect of the bid amount plus other explanatory variables (Table 3).The general format for parametric dichotomous choice evaluation of WTP was outlined in the two previous chapters. The unconventional bid design provided little definitive distributional clues and so a logit regression was taken as a starting point for the analysis, Table 4.

The first thing to notice about the regression is the coefficient on bid which is insignificant and of a magnitude which suggests a worrying lack of price sensitivity. This is confirmed by the plotted logit function figure 2, which is somewhat flat, suggesting that the distributional assumption has little effect on 'dampening down' the tail, which is 'fat' in both positive and negative domains. Hanemann (1984) suggests that slope value between 0 and -1 lead to potentially infinite means. The flat function is precisely the finding which in the study by the Australian Resources Assessment Commission on the South East Forests in Australia (Blamey and Common 1993).

It is worth conditioning responses on other variables to see if anything else is explaining the response pattern. Table 5 presents maximum likelihood parameter estimates and asymptotic t-values of a model selected for its predictive power. Values represent estimated means derived by regressing the observed WTP responses against the specified offer price for maintaining parks and additional independent variables. The estimated model is significant, with a likelihood ratio test of the hypothesis that the 14 coefficients are zero based on a chi-squared value of 35.35 with 14 degrees of freedom[7]. The likelihood ratio index (analog to R-squared in OLS) is low at 0.09, although the statistic cannot strictly be compared with R-squared of a classical regression (see Greene 1993 pp 653). More weight can be given to 73% of correct predictions although this is no great improvement on the model with the bid variable alone.

Most variables have the expected signs although the level of significance of several variables is disappointing and apparent conditioning adds little to the model other than confirm some expectations. Accordingly, The probability of saying yes falls with the bid amount presented to respondents, increases as income rises and is lower among older age groups. Dummy variables P1 to P6 and PRIV have been included to test conflicting hypotheses regarding WTP according to parks visited.

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the author.

[7] critical value of 23.69 chi-squared 14 at 5%

Table 3: Definition of variables tested in logit function modelling probability of a "no" response to park maintenance CVM question

Variable	Definition
X1	Offer sum in WTP question
INC	Annual family income (all sources)
AGE	Age in years
P1-P13 (see below)	Dummy variable for park visited (1 if visited 0 otherwise)
PRIV	Dummy variable for private reserve (1 if visited 0 otherwise)
EDUC	Level of education 1 lowest 4 highest
PASTSDUM	Dummy variable for past safari (1 yes 0 otherwise)
PLANSDUM	Dummy variable for future safari (1 plan 0 otherwise)
LOCDUM	Dummy variable for respondent location (1 inside park 0 elsewhere)
COMPDUM	Dummy variable for respondent on organized tour (1 yes, 0 otherwise)
CON	Dummy member of conservation group (1 yes, 0 otherwise)
TV	Dummy for viewing wildlife programs (1 yes, 0 otherwise)
FRDUM	French questionnaire (1 French, 0 otherwise)
GERDUM	German questionnaire (1 German, 0 otherwise)
ENGDUM	English questionnaire (1 English, 0 otherwise)
EXP	Visitor experience rating (1 lowest 4 highest)

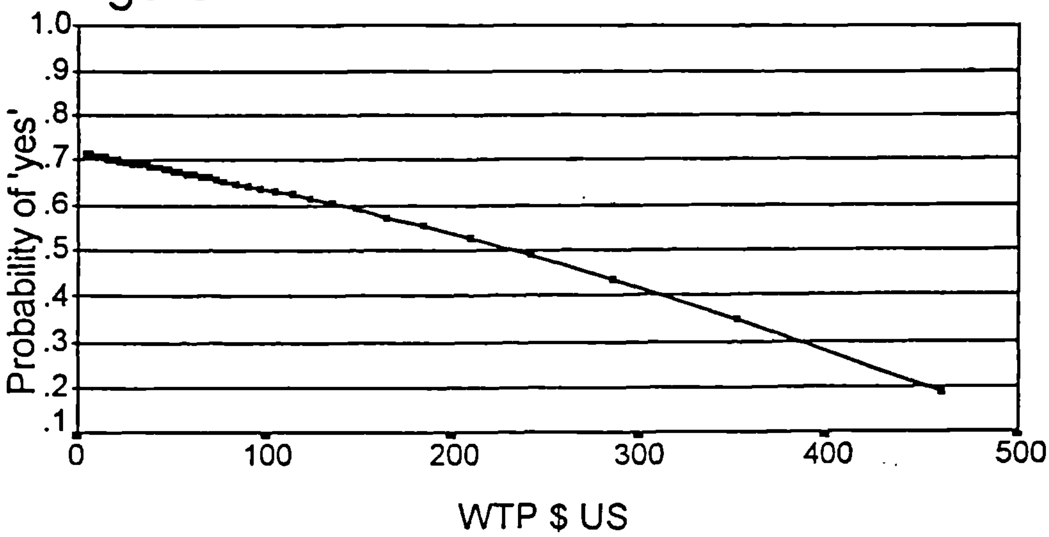
Park and Reserve dummies: P1 Nairobi, P2 Amboseli, P3 Maasai Mara
P4 Tsavo, P5 Aberdare, P6 Lake Nakuru, P7 Mt. Kenya, P8 East Turkana
P9 Marsabit, P10 Sibiloi, P11 Shimba Hills, P12 Samburu, P13 Meru
P14 Other (except private).

Table 4 Logit model (first bid)

Logit regression, (1/0 dependent variable)		
Variable	Coefficient	t-stat.
Constant	0.92553	4.924
Bid	-0.00338	-1.554

n=311
Log-Likelihood = -177.64
Restricted Log-Likelihood (slopes = 0) = -184.33
McFadden's R^2 = 0.04
% correct predictions 72%
y=1: 224, y=0:87

figure 2



Dichotomous choice (single response)

Table 5: Parameter estimates for logit function modelling probability of a "no" response to park preservation CVM question

Variable	Parameter estimate	t-ratio
Constant	0.0976	0.141
X1 (Bid)	- 0.0047	-1.778(10)
INC	0.0000585	2.006(1)
AGE	-0.0162	-1.489
P1	0.0082	0.033
P2	0.2599	0.832
P3	0.539	1.298
P4	0.0042	0.011
P5	0.468	1.61
P6	-0.808	-2.43(1)
PRIV	-0.707	-1.65(10)
EDUC	0.231	2.05(5)
PASTSDUM	-0.045	-0.162
PLANSDUM	0.457	1.6(10)

n = 311
Log-likelihood = -166.65
Restricted LL (Slopes = 0) = -184.33
Chi^2 (14) = 35.35
Likelihood Ratio Index (fit statistic) = 0.09
Significance level 1%, 5%, 10%
% correct predictions = 73%

One testable hypothesis on the congestion disamenity in the mega-parks (particularly P2, P3, P4) appears to be unfounded. Positive signs (albeit attached to insignificant coefficients[8]) indicate that a visit to these parks in fact increases the probability of WTP, probably due to the increased incidence of game. The negative signs on the significant coefficients for P6 and PRIV require some clarification. Respondents who visited private reserves were more likely to have paid higher rates and know what they paid and therefore object to a proposal of further payment. The negative coefficient on P6 is more surprising, and leaves room for testable hypotheses on the characteristics of visitors to Lake Nakuru (overlanders etc). The positive sign on future visiting intentions PLANSDUM seems to suggest that visitors are prepared to pay for the option of visiting parks in their current state. As expected, probability of a yes increases with education level.

5.6 Mean Estimation

As before the mean is given by the integral or formulaic equivalent of the expression:

$$Mean \quad WTP \; = \int_{LowBID}^{HighBID} (1 + e^{0.9255 \,-\, 0.00338\, BID})^{-1} \, d\, BID \tag{1}$$

The poor performance of the basic logit model for the single response, provides no reason to pursue the double-bounded which if anything will be more prone to yea-saying (see below).

Accordingly a restricted mean estimate produced a willingness to pay value of \$US 406.6 (95%, 201.73 - 6727.98) Recalling that these values are additional to park entry, both the point estimate and the confidence interval do not pass a basic credibility test. An alternative test of plausibility is to check the predicted WTP values corresponding to response percentiles of interest. Thus rearranging the expression for the predicted function

$$\log\frac{P}{1-P} \; = \; 0.925 \; - \; 0.0034BID \tag{2}$$

for a given probability $P = 0.05$, shows that 95% of the population will refuse to pay a value of \$1138, which seems an unreasonably high amount to pay for park entry. Again the difference in this value and the upper value of the confidence interval gives some indication of the length of the tail of

[8] A likelihood ratio test restricting variables P1-P5 to zero, which is chi-squared 5, compared to a critical value 4.01 fails to reject the null that these parameters are significantly different from zero

the distribution.

At this point a choice must be made about the validity of using this data for providing guidance on true population willingness to pay. Courses of action are either to rescue some information on which to base an aggregate consumer surplus estimate or to assume that the response pattern is not significantly determined by the appropriate variables suggesting that in fact the survey was poorly designed or administered. Having administered (but unfortunately not designed) the survey, the author is inclined concur with the latter view[9]. This gives some significance to the revealed value data that was collected in the survey (see below). To rescue a credible mean value implies the use of the median or truncation. Figure 2 shows the median WTP lying around $210 which is still excessive. Truncating the function introduces difficulties, not least because the decision is arbitrary. There have been several recent contributions dealing with the problem of fat tails at upper and lower (negative) WTP values (see Ready and Hu 1995; Kerr 1996). Basically these deal with forms of scaled or 'pinched' normalised distributions, which assume that the only problem was of a badly selected bid range as opposed to a fundamentally poor data set. With a particularly poor set such as this, the only way to rescue information is to specify a reasonable upper value and to simply calculate the integral of the area between the lower and upper value. Thus the mean would be given by

$$\begin{aligned} \textit{Mean WTP} = & \left[\frac{1}{0.00338} \log(1 + e^{0.9255 - 0.00338\,BID})\right]^{Upper} - \\ & \left[\frac{1}{0.00338} \log(1 + e^{0.9255 - 0.00338\,BID})\right]^{Lower} \end{aligned} \tag{3}$$

Choosing a reasonable maximum daily park increase of $200 and a minimum of $2 produced a truncated mean of $66.94 which is not an inconceivable increase. Because of the assumptions necessary to derive this welfare figure, its reliability for subsequent use is in doubt.

5.7 Response motives

Chapter 2 noted that Blamey and Common (1993) ascribed the flat response function to the dichotomy between respondent motives as citizens or private consumers (Sagoff 1988). It is possible to speculate further about the causes of the response pattern observed in this study. Without plotting the response

[9]Based on purely on first-hand observation of respondent behaviour and attitudes.

proportions it is clear that the predicted model does not show the expected tendency for the plotted proportions of yes responses to fall with the WTP amounts. Looking at the pattern of yes and no responses shows that the yes comprise 72%. The question to address is the cause of the propensity for response patterns of either sign to be located at high and low values. More specifically, what is behind apparent yea-saying behaviour causing respondents to accept values greater than their true WTP?

A typical argument to support yea-saying is that respondents are simply adopting the proffered amount as a value cue for a mental short-cut whenever they are unsure about their valuation of a well defined good. Similar responses may occur in the event that the good is not well defined by the survey and leaves the respondent pondering a whole universe of environmental issues on which to cause-dump. Ironically the unconventional questionnaire format requiring respondents to calculate and then consider a specific WTP value, may have sewn a degree of detachment from the actual value as a 'correct' cue price. However the second argument related to the extent of the scenario cannot be dismissed. In particular the suspicion must be that the good (park condition), was not adequately described to prevent some form of embedding or default assumptions about the extent of the good (Fischoff and Furby 1988). This might arise as a result of different visiting patterns amongst the respondents (obviously influencing what they saw in the parks) as well as their limited knowledge of the extent of the Kenya national park network.

Furthermore, the exact question wording requires some consideration of retrospective utility. Kahneman (1994) has disputed the reliability of what he calls peoples' 'evaluative memories' of their own preferences. In this case the implication is that asking individuals to time travel and reconsider a past purchase decision with a new price and other things being equal is problematic. In a wider sense Kahneman's view that people do not accurately recall their preferences challenges economic rationality. Stable preference learning is basically compromised. In so far as events can be shown to leave any trace that can be mentally compartmentalised, then only the peak and end experiences matter. Thus if the purchase decision is one part of a whole event which includes whatever respondents happened to be doing (or had just done) when they were surveyed, then the 'peak' of a recent game drive is likely to cloud a distant WTP decision. The result is that people probably consider the price offered on a set of merits which are other than the ones that the survey was designed to elicit. Consideration of what these are resurrects speculation about the legitimacy of certain motives and the related critiques of the warm-glow hypothesis mentioned in previous chapters. In essence no WTP motives can be excluded although one can speculate that African wildlife may motivate stronger symbolic motives which augment any other default assumption arising from the incomplete or biased scenario. Mitchell and Carson (1989 p250) note that a 'symbolic bias' is most likely to be a problem when the issue under investigation is controversial and/or stimulates strong emotional feelings which "might make it difficult for respondents to focus on the valuation-relevant aspects of the scenario" (eg the level of provision). Such behaviour seems to be reinforced by the findings of Schkade and Payne (1994 p100) in their verbal protocol reanalysis of the cognitively-taxing oil pot/bird kill questionnaire developed by Desvousges *et al* (1992). They find that 23% of respondents "suggested a desire to signal concern for larger or more inclusive issues, such as preserving the environment or leaving the planet for their progeny". In both cases, the level of provision of the actual good on offer may be of little importance.

The conclusion is that the data generated by the CV part of this survey demonstrate some of these aspects and is therefore unreliable as the basis for measuring consumer surplus. In these circumstances the only alternative is to use the revealed preference travel cost information.

5.8 The travel cost model

The theory and practice of the travel cost method is well developed (see Freeman 1994). The only difference here is the international dimension which, to the author's knowledge, has been addressed in only three studies (Brown and Henry 1989; Maille and Mendelsohn 1993, and by the author in Adger *et al* 1995). This dimension accentuates several well-known problems involved in using the method. For example, the valuation of time and the likelihood of multiple destination visits (Mendelsohn *et al* 1992).

The crux of the idea is the following. People who live in different cities and towns bear different travel costs when they visit a particular park. Those who come from far away bear a high travel cost compared to those who live close by. Therefore, the rate of participation by area of origin-city, region, country-should vary. This is a demand relation for a park. Roughly speaking then, information about (i) the origin of visitors to parks in Kenya, (ii) their rates of visitation and (iii) their "travel" costs are the fundamental ingredients necessary for a partial estimate of the economic value of parks. With a travel cost demand curve in hand, it is then possible to estimate what price should be charged to maximize revenues from those who visit game parks. Non-use values are disregarded and hence the approach is somewhat inferior to CV.

TABLE 6

Travel cost data:
air cost

Country	Median	Mean	Sample
United States	2,200	2,569	72
Canada	1,350	1.636	18
United Kingdom	853	1,044	50
Germany	817	779	27
France	831	775	33
Switzerland	1,445	1,516	3
Scandinavia	931	953	7
Australia/New Zealand	1,635	1,627	8

TABLE 7

TRAVEL COST DATA:
LAND COST PER SAFARI DAY
(Overall Median = 116, Mean = 182)

Country	Median	Mean	Sample
United States	221	313	71
Canada	87	126	17
United Kingdom	98	104	51
Germany	83	83	29
France	166	164	34
Switzerland	138	134	1
Scandinavia	61	80	6
Australia/New Zealand	241	270	8

Travel cost data

Respondents reported separately the cost of airfare and the safari or land cost in the currency of their choice[10]. (Quest. 15 -18). See Tables 6 and 7. They also reported the number of days on safari, Table 8. Recall that the driving force of the travel cost technique is different rates of participation associated with different "prices." Individual data are inappropriate because each observation is a trip. We aggregate up to the country level. The nine countries in our sample constituted 78 percent of the lodge nights in 1989.[11] Therefore the observation of interest is safari trips per country. These data do not exist for Kenya. The latest Economic Survey for Kenya available (1990) reports the lodge nights by country. A country's lodge nights divided by sample mean estimates of safari length for each country provides the estimate of visitors or trips per country (Table 8).

Trips per capita for the sample countries, computed using population data, are combined with median airfare cost for each country to estimate a travel cost demand relation using a weighted least squares linear regression procedure. Figure 4 illustrates the econometric results in Table 9. Three observations are in order before describing the results.

TABLE 8

TRAVEL COST DATA:

Country	Population (000,000's)	'89 Lodge Nights (000's)	Safari Days Mean	Visitors= Nights/Days	Air Cost Median
United States	254.5	146.6	8.7	16,897	2,200
Canada	27.3	11.5	11.1	1,032	1,350
United Kingdom	57.8	96.8	7.8	12,410	853
Germany	68.4	104.6	11.1	9,451	817
France	57.3	62.0	8.75	7,085	831
Switzerland	6.8	49.6	15.	3,306	1,445
Scandinavia	23.2	9.2	6.	1,533	931
Australia/New Zealand	20.6	8.1	11.2	725	1,635

First median airfare costs are used for each country to avoid giving undue weight to those flying first class. The travel cost procedure attributes all the utility obtained from the trip expenditures to the safari and none to the travel. Why then do otherwise equal individuals (by assumption) choose different classes of air service? Only if the extra cost a first class fare provides no utility can one

[10] The exchange rates were an average for October 1993.

[11] Central Bureau of Statistics, 1990.

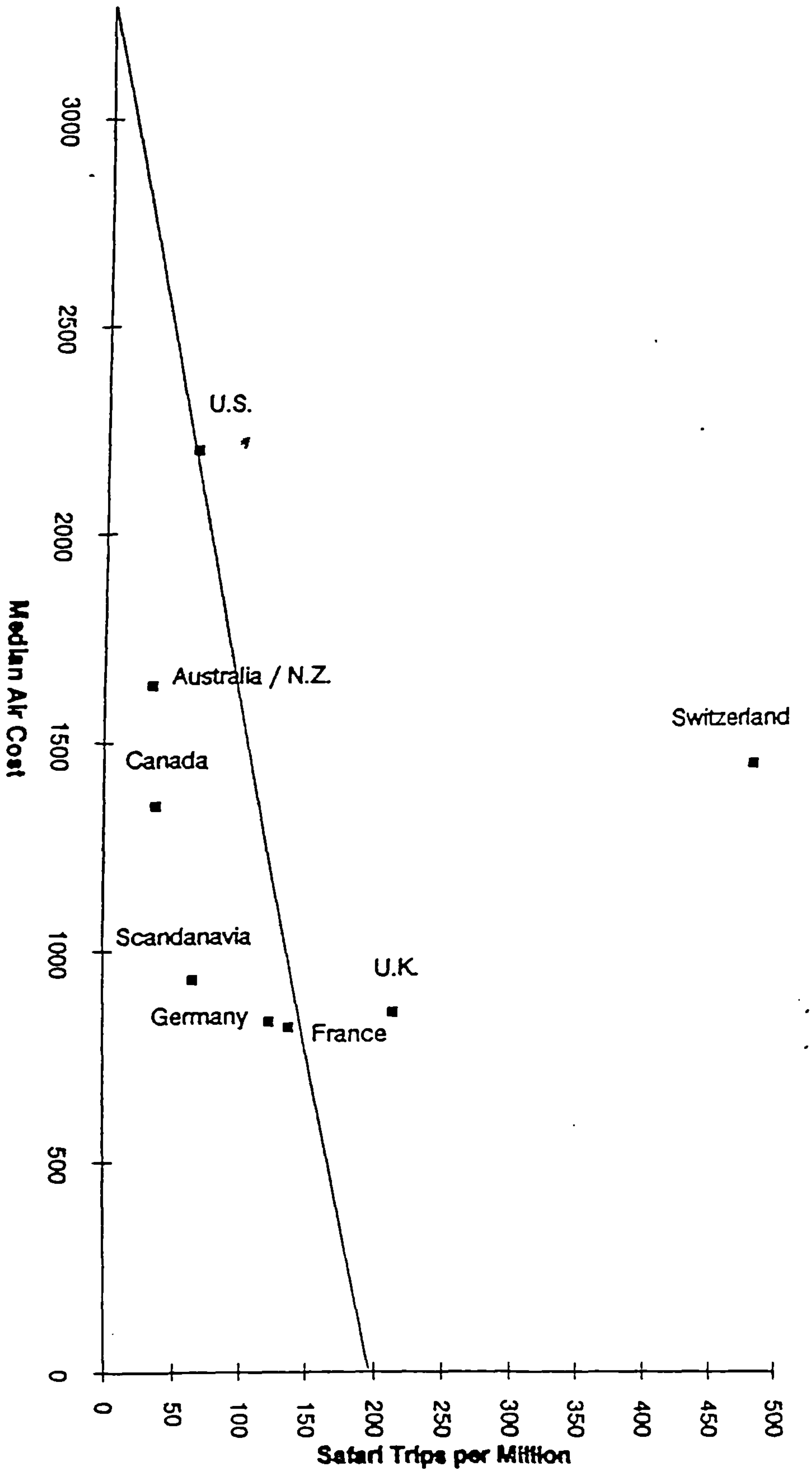


FIGURE 3

TRAVEL COST DEMAND AND
ACTUAL VISITATION RATE

arguably give the same weight to first class fares as is given to Apex fares. Second, the cost of a safari has been omitted from the estimation procedure. We are in effect, estimating a consumers' surplus function for a safari of average quality which, in fact, sells for about the same price to everyone regardless of country of origin. Safari prices, in fact, are common to all. As such, these prices provide no useful information about the demand for a safari. Moreover, safari costs are also omitted because they are and theoretically should be strongly correlated with the cost of airfare. The econometric consequence of including land costs in the travel price is to bias the consumer's estimates upward. The demand for safari quality is however considered in a subsequent section.

Second, it is well to emphasize, that quite apart from the above qualifications, Figure 4 is not a consumers' surplus relation for a safari, only for those who bear no travel cost. To get a consumers' surplus function for safaris in general, one needs to aggregate the function for each country across countries. Thus to compute consumers' surplus (CS) for safaris taken by all, let the estimated function be

$$(1) \qquad X_i = b_0 + b_1 \ P_i = f\ (P_i),$$

where P_i = median airfare cost from country i; X_i = visitation rate (per unit population) for country i; N_i = population for country i.

Then the consumers' surplus for safari's is

$$(2) \qquad CS = \sum N_i \int_{P_i}^{P} f\ (t)\ dt,$$

where P is the maximum median air cost ($-b_0/b_1$) and t is the variable (cost) of integration.

Third, a weighted least squares routine is adopted to adjust for heteroscedasticity. Since the population per country varies from 7 million to more than 250 million and our sample iss trongly correlated with population, each observation was weighted by for each country (Maddala, 1979).

Fourth, it is apparent from Figure 3 that the observation for Switzerland is unusual. It is the average of 3 individual respondents. The representative travel cost for Switzerland is $1,445, 1.75 times the median for Germany and France. It could be argued that this is an outlier observation and should be dropped or Switzerland could be attributed the European mean or median cost. The regression results above represent actual data for Switzerland. Attributing the European mean cost of airfare to Switzerland reduces the estimated consumers' surplus per day by less than 10 percent.

The effect of functional form on consumer surplus estimates is similar to the issue of distribution selection in CV and is well documented (Ziemer *et al* 1980; Adamowicz *et al* 1989). The regression results for three functional forms of the travel cost-participation rate are reported in Table 9. Statistical measures such as log-likelihood ratios do not single out a superior equation. We adopt the linear form on grounds of simplicity, noting that in doing so, that functional form yields the lowest consumer's surplus. It is difficult to discriminate between alternative functional forms because, there is not much variation in travel cost. The regressions reported in Table 9 are represented in Figure 4. There is not much difference in the three functions over the range $800 - $2400 which includes all the average airfare costs (Table 7). The intercept terms are highly statistically significant. Happily, the slope terms are negative and statistically significant at better than 10 percent, using the one-sided t-test. The R-squared for each regression is below 0.33 which is not surprising given the nature of the cross-section data and sample size.

5.9 Consumer's Surplus From Travel Cost Method

Using (2) and data in the above-mentioned tables, the estimated average consumer's surplus in the linear regression is $77 per day. Consumer's surplus for the linear-log regression is $105 and when the dependent variable is in logs, consumer's surplus is $134 per person per day.

Some respondents visited other countries and some enjoyed shopping and other activities in Kenya not directly related to safaris. There are at least two ways to adjust consumer's surplus to handle the fact that a safari can be one element of a bundled good, a vacation trip.

First, one can create several goods; for example, a safari trip, a safari trip plus one stopover outside of Africa; a safari trip, one stopover outside of Africa and one or more other countries in Africa. Then estimate a set of simultaneous travel cost demand equations for these three goods. Available data are inadequate for this task. The alternative is simply to ask respondents to distribute their total

TABLE 9 TRAVEL COST FUNCTIONAL FORMS				
	Constant	Air Cost	R^2	WTP/Day
Linear q=a+b*p	196.0 (61.5	-0.0598 (0.0361)	0.215	$77
Linear q=a+b*In(p)	722. (321	-85.4 (44)	0.216 -	$105 -
Log-Linear q=exp(a+b*p)	5.50 (0.44)	-0.05E-04 (3.54E-4)	0.216 -	$134 -
Safari Share of Air Cost (Linear)	178. (63.9)	-0.071 (.0548)	0.106 -	$93 -

pleasure of their trip over its attributes (Q14) which yielded:

67	Safari
21	Other Aspects of Kenya
12	Features of the Trip Outside Kenya
100	

Respondents, on average, attributed 67 percent of their total pleasure to the safari. The reported consumer's surplus estimates of $77 per safari day is obtained by estimating the consumer's surplus for each country, weighting it by the fraction of total value attributed to safaris for each country, then aggregating over the countries. There are two other ways to estimate consumers' surplus. One is to compute the median airfare for each country in the sample and weight each country's median airfare by its percent of pleasure associated with the safari (see Table 10). In this case average consumer's surplus per day is $93. Alternatively, when consumer's surplus for each country is computed from the underlying regression, then weighted by the sample country's average percent of pleasure, the overall average consumer's surplus per day is $78. In sum, estimates of CS vary from $77 to $134 per day, depending on the function and the rule adopted to estimate the safari value's share of the total value of the joint product produced by a journey. What is the best value? The Box-Cox test for

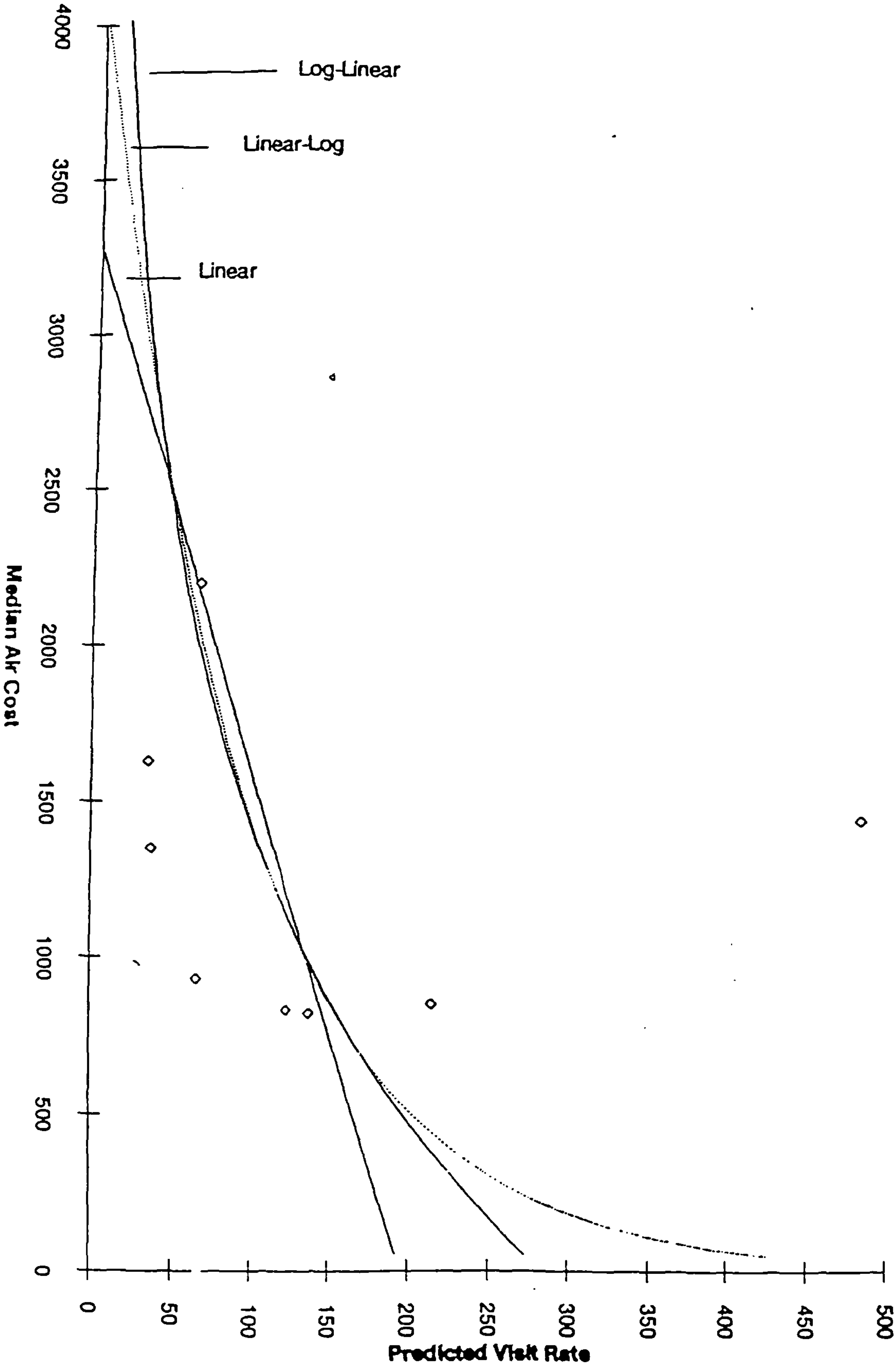


FIGURE 4

ESTIMATED AIR COST AND VISITATION RATE

functional form is indecisive[12] so we choose the lower estimate of $77 per person from the linear specification to be "conservative," but this is not a very compelling criterion.

TABLE 10

SHARE OF PLEASURE ATTRIBUTED TO THE SAFARI

Country	Share
United States	69
Canada	65
United Kingdom	70
Germany	57
France	75
Switzerland	96
Scandinavia	76
Australia/New Zealand	55

A further testable proposition with this information relates to the demand for different quality safaris according to distance. The hypothesis is based on the observation of lower *relative* prices for people originating from greater distance leading to the consumption of more costly safaris. That is:

$$\frac{P_H}{P_L} > \frac{P_H + c}{P_L + c},$$

where P_H and P_L are the price of the high and low priced good respectively and c is the transport *cost* per unit. For an extensive discussion of this topic see Silberberg (1990).

Silberberg (1990) demonstrates that this prediction requires that the cross price elasticity of demand of safaris with all other goods should be the same, regardless of safari quality, an assumption which seems innocuous to us. This is a novel setting in which to test the proposition. Big tour operators such as United, Abercrombie and Kent, and others, print up a brochure describing the menu of safaris offered and the price of each and distribute them worldwide. While one or more tour operators may have a substantial market share in a particular country, there is healthy competition with many suppliers in a given country and more than one hundred tour operators overall. In short, while quality

[12] The estimate of $\lambda = 0.715$ with a standard error of 0.543. A statistically significant value of λ not different from 1 implies a linear specification; λ not significantly different from 1 implies a long-linear specification.

varies, the price of a given quality safari package or the land cost, is the same for everyone in a given country.Since transportation cost varies by origin, we expect that those who reside further away should purchase relatively higher quality safari packages, that is:

$$\text{Land Cost}_j = f(\text{Airfare Cost}\, j\ldots),\ f' > 0$$

The econometric results of estimating this in a variety of ways are illustrated in Table 11.

TABLE 11

The relative price hypothesis

Dep. Variable	Constant	Air Cost	Income
Mean Land Cost	-626. (447)	1.99 (.522)	-11.9 (13.9)
Median Land Cost	-519 (123)	1.64 (.145)	-1.75 (1.52)
%High Cost Mean	0.107 (.057)	1.89E-04 (0.67E-04)	4.17E-03 (1.78E-03)
%High Cost Median	-0.164 (.0697)	3.68E-04 (0.82E-04)	1.12E-03 (5.6E-03)

The independent variable in the first two regression is median airfare. The other independent variable is median or mean income in the first and second regression respectively. Median airfare is highly significant, at about 7 percent or better, while neither median or mean income (in unreported regressions) is significant. We would expect the quality of safaris to be income responsive and are a little surprised by these results. The R-squared statistic is very high, greater than 0.85 in all the regressions. For the third and fourth regression, the dependent variable is the fraction of visitors in each country whose land cost is above the sample median or sample mean cost per day. The sense of this is that as airfare costs increase, the fraction of high to median quality trips should increase. The results of the linear, weighted lease squares regression (above) show just that. Moreover, the significance level of median airfare is better than 5 percent.[13] Happily median income is now statistically significant at better than the 10 percent level.

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[13] The weight is the square root of the population observations from each country.

5.10 Price Setting with Travel Cost Estimates

The suspect nature of the contingent valuation data set restricts pricing considerations to the travel cost information and two pricing policy options are considered: What is the price to charge per day which maximizes park revenues from entrance fees? Charging on a per day basis allows visitors to reduce cost by choosing less days. This results in lower revenues if demand is elastic. This substitution effect of a price increase can be avoided by charging a single price. Therefore, what is the lump sum charge which maximizes revenue?

The revenue maximizing entrance fee can be derived from the estimated demand curve. The visitation Q_i for any country i is $n_i*(\alpha+\beta P_i)$ where n_i is the population, P_i is the airfare and $(\alpha+\beta P_i)$ is the estimated demand relation. If the government adds an entry fee of F, the visitor-days for a country will be $Q_i[F]=n_i*(\alpha+\beta(P_i+F))$. The corresponding revenue will be $Q_i[F]*(F+E)$ where E is the existing entrance fee, about \$14 for each day. Total revenues (R) across all countries will then be $R=\sum Q_i[F]*(F+E)+\sum n_i*(\alpha+\beta(P_i+F))*(F+E)$. To find the revenue maximizing fee F, the first order condition is:

$$\sum n_i*(\alpha+\beta(P_i+2*F+E))=0.$$

Rearranging yields:

$$F=-(\alpha/\beta+E+\sum n_iP_i/En_i)/2.$$

This is a lump sum, not a daily fee. To find a corresponding per day fee, divide F by the average number of days on safari. This approach fails to capture the possible response to a higher fee of shorter trips, but it is not clear which way that would shift the optimal fee. The reduction in days from a small fee increase may be underestimated and the reduction in days by the average number of days on safari. Hopefully, these effects cancel out. The estimated revenue maximizing fee F is \$790. On a per diem basis this works out to \$93 for an average of 8.5 days per visitor. These fee are in addition to the current \$14 per day fee. Charging these fees would increase revenue for 415 percent.

Although it is not good practice to compare the welfare estimates of these two studies (particularly given the caveats on the reliability of the CV data) used here and those usually attached to even the most carefully designed studies, many authors do (eg Loomis *et al* 1991).

5.11 Discussion.

A conservative estimate of the consumer surplus associated with non-consumptive use of Kenyan park resources can be derived by multiplying the favoured WTP per day estimate by the typical number of days spent in parks by visitors and then aggregating over some proportion of foreign arrivals in Kenya (assuming that not all arrivals visit parks). Table 13 uses the travel cost data and recorded holiday arrivals for 1990 and 1992[14] to give a range between $49 and $482 million per annum (equivalent to between $16 - $157 per hectare), depending on assumption about the number of days spent in parks. Under the best visitation scenario, this surplus more than doubles best (comparative static) estimates of opportunity costs of $203 million per annum (Norton-Griffiths and Southey 1993), and thereby provides an economic justification for current resource use. Note that 1992 figures record a downturn in visitor numbers, although recent currency devaluations can be expected to reverse this trend. This aggregate measure of surplus is *additional* to recorded net financial revenues and the consumer surplus attributable to *resident* Kenyan park users. Furthermore, it does not account for the extent of other indirect use values such as watershed protection nor any direct attempt to elicit existence values from a wider population of non-users.

What proportion of this surplus might park managers be able to capture through fees? In an ideal world the demand curve which bounds this estimate of consumer surplus (figure 4), would be determined solely by consumer preferences based on market information which includes knowledge of substitute goods. That is, the expressed surplus of visitors to Kenya would reflect their own appraisal of the utility derived from park use, and, subject to the elasticity (slope) of this curve, some proportion of this surplus could always be captured before demand is driven to zero. Determination of the elasticity of demand for park tourism is however complicated by the presence of intermediaries between sellers and many of the ultimate consumers. This causes two problems. Firstly, the actual price of the good on offer becomes less transparent to consumers who pay an overall price for a package of which it is a constituent part. As was shown above this complicates the payment vehicle choice in the use of CV to elicit park-related preferences. Secondly, intermediaries complicate the measurement of a tourist cross-price elasticity of demand for wildlife sites, replacing it of an operator assessment normally determined by profit margins and the extent to which operations are sunk into one country as opposed to a competitor.

[14] Note that some proportion of individuals who state their purpose of visit as 'business' or 'other reasons' can be expected to visit parks. Use of holiday arrivals therefore understates true visitor numbers.

Of course, an assumption of operators being aware of extant surplus on the part of visitors or even sharing similar preferences might alter reactions to unilateral fee increases. In this case visitation rates could be maintained, albeit at higher prices (although probably some surplus going to operators).

Table 12: Total consumer surplus of non-consumptive use and fee capture scenarios

		1990	1992
Total visitors '000		814.4	698.6
Holiday visitors '000		695.6	588.1
Days in parks	(1)	1.1	1.1
	(2)	9	9
Total consumer surplus 1993			
	(1)	\$58.9m	\$49.8m
	(2)	\$482m	\$407m

Mean days spent in parks (1) Southey (1992) estimate (2) This survey n= 469

Ultimately, the prevailing market structure of East African tourism and an asymmetry of information between visitors on the one hand and operators and pricing managers on the other, suggests that the conventional demand curve is likely to be discontinuous at some price, limiting revenue capture to a fraction of total consumer surplus. A 'kinked-demand' model (see Koutsoyiannis 1979), can be rationalised by the largely non-collusive oligopolistic nature of African 'wildlife supply'. Suppliers will expect competitor countries to match a price cut, which will increase total demand but leave market shares unchanged. A price increase will however, not be followed. Diagrammatically (figure 5), this gives rise to the relatively price elastic section *yz* of the demand curve beyond notional price P_o which bounds Kenya's potentially capturable surplus *yzt*. A collusive pricing strategy with close rivals is one way to maximize returns from a capturable surplus *ytx*, bounded by a market demand curve *wyx* (Figure 4).

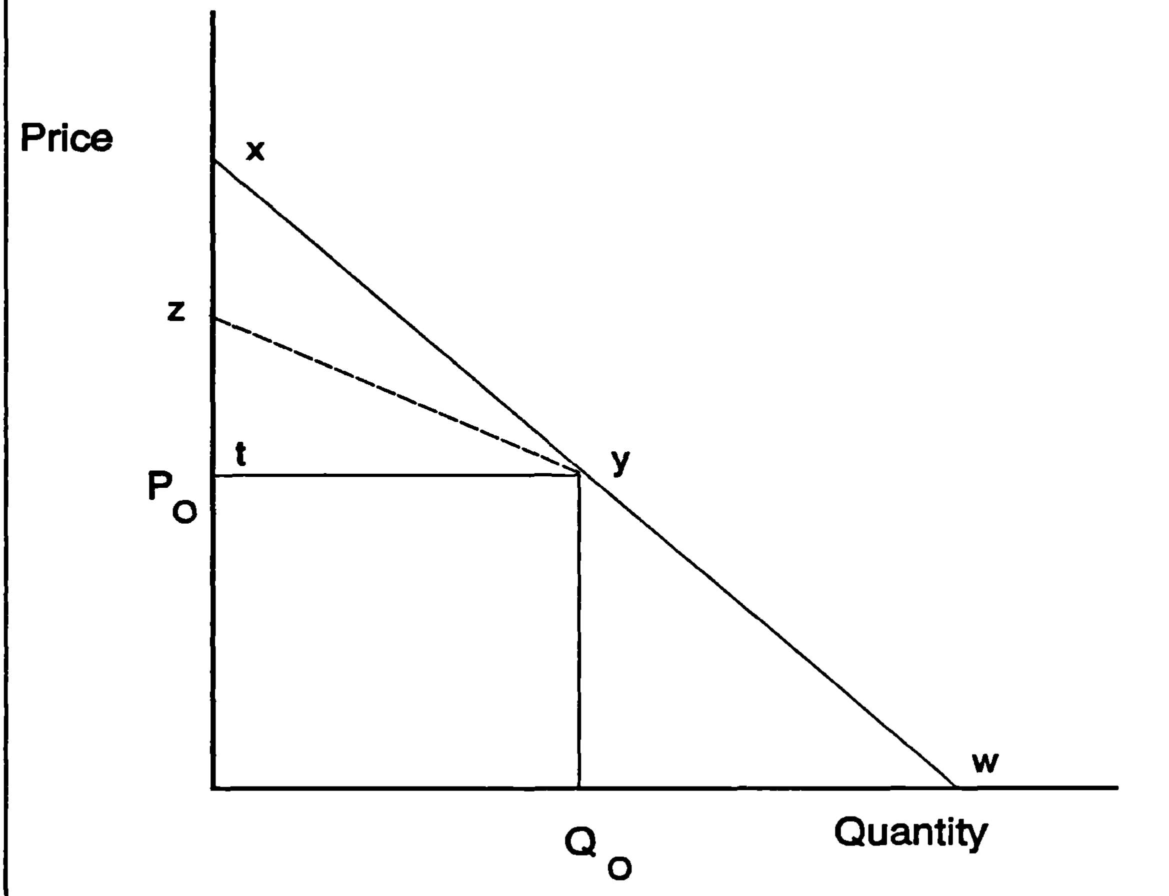
With due regard for competitor pricing, and operator reaction, this study suggests that KWS might individually experiment with a margin between the current fee of \$15 and \$85. Noting the likely sensitivity the price elasticity estimates from the limited travel cost data, we can only speculate about the effects on visitation. However, assuming an attempt to capture 50% of identified consumer surplus implies a fee of approximately $50 per day and the potential capture of between $23 million and $225 million per annum. A more modest fee increase to $30 implies a 20% capture rate ($9.3 and 90 million per annum), while parity with current Tanzanian fees of $20 implies capture of around 7%, $3.23 - $31.3 million, an interval encompassing recurrent expenditure projections of US$30 million by 1996 (KWS 1990). All scenarios naturally depend on efficacy of fee collection methods.

Calculation of aggregate consumer surplus is not merely of interest for fee purposes. What KWS may not be able to capture through established instruments may become capturable through other mechanisms designed to compensate for missing *global* markets (chapter six). The Convention on Biological Diversity makes reference to a commitment on the part of developed countries to finance the " agreed full incremental cost" incurred by developing countries in complying with the convention (art. 20, UNEP 1992). This implies that a global market might be created through which developed countries may compensate developing countries according to the net costs incurred in increasing conservation effort relative to some baseline, and which is deemed to provide global benefits. In other words, the identification of a uncaptured global benefit provides the rationale for a transfer to Kenya to finance projects where domestic costs outweigh domestic benefits. From a global perspective, knowing how great these benefits are, provides one efficiency indicator for ranking transfers according to the ratio of global benefits to incremental costs (net costs). Such a process has already begun under the auspices of the Global Environment Facility.

5.12 Conclusion

It is important to get the pricing policy right for parks and reserves in developing economies. These habitats store precious biodiversity for the enjoyment of all but certainly for the benefit of the relatively rich visitors from North America, Europe, the United Kingdom and elsewhere. The stakes are large and in some cased irreversible. The Kenyan park case is an interesting one as Kenyan park use is somewhat under-regulated relative to other safari suppliers. It is absurd that visitors can pay as little as $14 and to harass wildlife in Kenya while a substitute like Zimbabwe may charge double, restrict park use and suffer no adverse impact on demand and still have a cachet lacking in Kenya. It is, however, a vexing problem even if we focus on use value alone. Going on a safari is one of the joint products of a vacation to one or more foreign countries. Maybe this can be sorted out in principle by analysing a set of demand functions for nested goods; a safari, a safari and a stopover in London, a safari to Kenya and one to Tanzania. One might be courageous enough to tackle this from an hedonic analysis framework, carrying through the analysis to estimate the demand function

Figure 5 The demand for wildlife viewing



for a set of characteristics which constitute a safari. The empirical challenge is daunting both in terms of the number of necessary observations and the quality and quantity of information required from each participant.

In the meantime one adopts pragmatic measures such as asking respondents to allocate total satisfaction over the constituent parts of the joint product and experiments with alternative valuation methods, as we have done. In this pragmatic manner, improved econometric tools are brought to bear on the problem and familiar pitfalls of contingent valuation methods are discovered

this study has attempted to quantify the benefit associated with the non-consumptive use of Kenyan parks and reserves. The central estimate of consumer surplus of \$178.6m[15] million per annum demonstrates the magnitude of benefit provision by Kenyan conservation and some proportion of revenue foregone at current pricing rates. This surplus represents only one category of total economic value but is itself sufficient to overturn approximate estimates of the opportunity cost.

Growing pressures on protected areas hasten the need to emphasise the economic case for conservation versus competing resource uses, perceived to provide more 'tangible' returns. This perception is most critical for communities in closest proximity to parks often barred from direct participation in tourism, yet tolerating marauding wildlife and tourists. In the absence of workable compensation, this tolerance of conservation and acceptance of an inferior resource rent amounts to a subsidy to the tourist industry, and, as demonstrated here, to foreign visitors. Experience elsewhere has demonstrated the necessity of instilling a perception of wildlife as a public resource, capable of generating returns to all that live with it, and not just for private gain. Raising local stakes in, and even dependence on, park existence begins this process.

Compensation pre-supposes capture of currently skewed benefits, although current market structure may limit the extent to which current instruments (ie fees) can be employed. Some initial capture is feasible with increases, and possible differential pricing addressing additional issues of congestion and load redistribution to remoter parks and emerging private reserves. Alternative instruments may correct for the absence of a market for a global WTP for conservation. The issue of determining Kenya's incremental cost - that is, cost arising from conservation it would not ordinarily undertake in the absence of transfers - may be problematic (King 1993).

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[15] (407 - 48.9)/2.

The versatility of both CV and the travel cost methods means they are useful instruments for informing decision making in conservation planning and management. Limitations of the travel cost method for valuing more specific aspects of biodiversity or biological resources are self-evident. In the case of CV the implication is that imply that biodiversity conservation surveys must be framed in a fashion which is both comprehensible and familiar to respondents and with due regard to statistical design protocol. To date, applications have concentrated on parks, celebrated ecosystems and species. From the conservationist's perspective, this focus can be rationalised by the frequently inseparable nature of the subject good from its biosphere and supporting species links. In other words, the 'purchase' of the good offered in a CV exercise can often imply the (albeit unintentional) purchase of a complementary bundle of biodiversity. A concentration on subject 'umbrella' or 'flagship' species and ecosystems (Noss *et al.* 1992), is therefore consistent with a much broader view of biodiversity. With careful survey design and a scientific input, subject goods can be broadened to encompass vital links and a multitude of optional and quasi-optional biological value. Clearly, the maxim of 'more is better than less', is one view of how CV applications may progress.

Valuation of familiar units of conservation estate need not preclude the use of the methodology at a more disaggregated species or ecosystem level. At some point however, respondent information constraints become binding, and the fundamental issue of information provision must be addressed. Few experimental studies have investigated this information threshold in the context of biological resources. Those that have, unsurprisingly find that responses are highly conditioned on the amount and content of information provided, Hanley Spash and Walker (1995), DeKay and McClelland (1995). This finding raises the interesting and controversial issue as to whether CV respondents should be given any information at all. In the context of the NOAA panel review of submitted evidence, this remains an outstanding issue. More specifically, should uninformed respondents be informed and how does this process affect their eventual responses to WTP questions? If not, should the responses of uninformed respondents count, and what does this imply about the range of subjects suitable for CV studies? Respondent familiarity with African wildlife presented no such constraint in the current study. If anything the problem was that they were over-informed but in the wrong way. Information acquisition and processing is an ongoing process. There are no fixed rules about preferences formation and it seems that the coherent recall of previous preferences is limited. Yet the for the general use of even the most flexible valuation currently available the basic problem is a lack of information. Not only is it largely unavailable, but whether it could be presented in any meaningful way. This rather pessimistic conclusion should not detract from a vital role that valuation methods can play in resource allocation. Broader conclusions on this subject are reserved for the concluding chapter.

Post Scriptum

Fee increases and price differentiation were adopted by Kenya Wildlife Service in late 1995 (see Elliot 1996). Whether these were informed by the current survey is unknown. Moreover the difficulties of sustaining wildlife on the basis of non consumptive use are now apparent. KWS is currently undertaking a policy review with the aim of reintroducing sustainable use policies.

7/6/93

Tourist Survey
Version A.1.1

KENYA WILDLIFE SURVEY

KENYA PARKS AND RESERVES STUDY
A VISITOR SURVEY

Dear Visitor:

Kenya is one of several east African countries studying how to provide and pay for wildlife parks and reserves for the enjoyment of people from around the world. The answers you and others give in the survey are important. They will inform managers about the value of parks and reserves so that they can plan accordingly. Survey responses will be treated with confidentiality. No individual responses will be released, only grouped data will be made available. Your cooperation in answering these questions and returning the survey is much appreciated.

If you are not a resident, please fill out the survey now, put it in an envelope and we will collect it from you.

Answer all the questions as best as you can. It will only take about twenty minutes. If you would like a summary of the results, please check the box on the last page of the survey and we will send them to you.

Date: Lodge/Airport:

INTRODUCTION

The first five questions address how important wildlife and environmental matters are for you and how they compare with other problems in the world today.

Q1. We are interested in your views about protection of wildlife and their habitat by public authorities around the world. How would you rate your concern about this topic? (Circle the number of the most appropriate response).

Not at all Concerned						Greatly Concerned
1	2	3	4	5	6	7

Q2. How adequately do you believe Kenya is now protecting its wildlife and their habitat based on what you have read and seen? (Circle the number of the most appropriate response).

Not Very Adequately						Very Adequately
1	2	3	4	5	6	7

Q3. There are many problems in the world and in your own country, none of which can be solved easily or inexpensively. Some of these problems are listed below and for each one indicate whether you think *your country* should spend more, the same, or less money than it is now spending now by circling the most appropriate response:

		Much Less	Somewhat Less	Same Amount	Somewhat More	Much More	Not Sure
A.	GIVING FOREIGN AID TO POOR COUNTRIES FOR FOOD, MEDICINE, DEVELOPMENT	1	2	3	4	5	8
B.	MAKING SURE THERE IS ENOUGH ENERGY FOR HOMES, CARS, AND BUSINESSES	1	2	3	4	5	8
C.	FIGHTING CRIME	1	2	3	4	5	8
D.	REGULATING POPULATION	1	2	3	4	5	8
E.	IMPROVING PUBLIC EDUCATION	1	2	3	4	5	8
F.	PROTECTING THE ENVIRONMENT	1	2	3	4	5	8

Q4. It is useful to understand how important you feel protecting natural habitat is compared to some of the many other issues also facing citizens in your country. From least important to most important, how do you rate the issues listed below? (Circle number of best response for each issue.)

	Least Important						Most Important
	tt						
IMPROVING PUBLIC ROADS AND HIGHWAYS AT HOME	1	2	3	4	5	6	7
MEDICAL RESEARCH ON LIFE-THREATENING ILLNESSES	1	2	3	4	5	6	7
REDUCING AIR POLLUTION AT HOME	1	2	3	4	5	6	7
PUTTING A SPACE STATION IN ORBIT AROUND THE EARTH	1	2	3	4	5	6	7
PROTECTING NATURAL HABITAT AT HOME	1	2	3	4	5	6	7

Q5. Do you watch television programs about animals and birds in the wild...

1. NEVER ____________________ 1
2. RARELY ____________________ 2
3. SOME OF THE TIME ____________________ 3
4. FREQUENTLY ____________________ 4
5. VERY FREQUENTLY ____________________ 5
6. NOT SURE ____________________ 8

QUESTIONS ABOUT YOUR TRIP:

Q6. Please take a few minutes to explain the purpose of your trip and your most satisfying experience so far.

Q7. Will you visit next or have you visited countries other than Kenya on this vacation? (Circle correct answer):

1. NO

2. YES → How many days is your total trip including days spent in other countries? __________ days.

→ Name the countries: ______________________________.

Q8. Company with whom you booked your trip to Kenya:

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Q9.a About how many hours did it take you to *fly* or drive to Kenya from the place where your trip began?:_______ Hours. Or, how long was your *drive* from the last county to Kenya? _______ Hours.

Q9.b At what *city* did your trip to Africa begin? ___________________________.

Q10. How many days will you stay in Kenya?:_________________ Days.

Q11. How many days will you spend on your Kenya safari in parks or wildlife reserves on this trip?

_________________ Days.

Q12. At this point in my trip I have spent _________ days in the parks or wildlife reserves in Kenya.

Q13.a Check the parks or reserves you have visited or plan to visit: (Circle numbers of all that apply)

1.	NAIROBI	8.	EAST TURKANA
2.	AMBOSELI	9.	MARABIT
3.	MASAI MARI	10.	SIBILOI
4.	TSAVO WEST	11.	SHIMBA HILLS
5.	ABERDARE (INCLUDES TREE TOPS)	12.	SAMBURU
6.	LAKE NAKURU	13.	MERU
7.	MT. KENYA	14.	PRIVATE SANCTUARIES
		15.	OTHER EXAMPLE

Q13.b Please tell us in a sentence or two why you chose this combination of parks:

Q14. People travel to East Africa for many reasons. Thinking about the pleasure and enjoyment you are experiencing (or have experienced) from your visit, what percent of your pleasure would you attribute to each of the following? **(Please make your responses add to 100 Percent)**

	Percent:
SEEING, PHOTOGRAPHING AND LEARNING ABOUT THE WILDLIFE IN PARKS AND RESERVES *IN KENYA*	______
VISITING OTHER PARTS OF KENYA OUTSIDE THE PARKS AND RESERVES SUCH AS NAIROBI OR THE OCEAN RESORTS	______
VISITING PLACES *OUTSIDE OF KENYA* ON THIS TRIP	______
	100%

Q15. The cost of your safari has two parts: airfare and land or other costs. What is the approximate *land* cost per person of your safari experience in Kenya? *DonÕt include airfare* to Kenya.

TOTAL SAFARI LAND COST PER PERSON = ________ IN ____________________

CURRENCY.

Q16. **What is the approximate round-trip airfare cost per person you paid to get to Kenya from where your trip started:**

AIRFARE COST PER PERSON = ________ IN ____________________ CURRENCY.

Q17. If you are visiting several countries and know the added airfare cost of the Kenya portion, enter this cost below: KENYA PORTION OF AIRFARE PER PERSON = _______ IN __________ CURRENCY

Q18. It will help when you answer other questions to have a total cost of your trip to parks and reserves. The total cost per person is just the sum of your responses to questions 15 and 16 or 17. *Just put in total cost per person if you donÕt remember the land or airfare cost individually.*

SAFARI LAND COST = _________

AIRFARE COST = _________

TOTAL COST = _______

IN ________________________ CURRENCY

Q19. How many members of your household are traveling with you _________?

QUESTIONS ABOUT COSTS AND CHARGES:

Kenya is one of several countries in East Africa where there are many parks and reserves which support a rich variety of wildlife. Kenya Wildlife Service recently completed a study which provided estimates of the cost of

- Monitoring And Protecting Wildlife,
- Preserving Habitat At Sustainable Levels And,
- Effectively Administering The Park And Reserve System

- Government expenditures meet the costs not covered by entrance fees.
- Since parks and reserves greatly benefit non-Kenyans, there is increasing pressure to spend government funds on other needed development projects.
- If park revenues do not increase, some land now supporting wildlife will change to settled agriculture and other productive land uses that will provide jobs and income.

If higher entrance fees are charged, some people will be willing to pay the higher costs of a safari that incorporate these fees because the experience is worth it and because helping to maintain biodiversity in this way is useful or for other reasons.

Some people would not be willing to pay the higher costs of a safari because the extra cost makes the safari too expensive and because they believe others should contribute to the maintenance of the parks and reserves or for other reasons.

Q20. Suppose, according to the wildlife management agencys' *preliminary* calculations, a new entrance fee increases the land and air cost of your Kenya safari by 10 percent of the total cost you answered in Question 18. *Thinking back to when you decided to take the safari, would you still choose this safari at the 10 percent higher cost?* (Circle correct answer.) Sample calculations of percents are at bottom of page to help you get an answer.

(IF YES, PLEASE GO TO Q21) → 1. YES
2. NO (IF NO, PLEASE GO TO Q22)

Q21. If the *final* estimates of cost amounted to a 15 percent increase in your total safari, would you choose this safari at the higher cost? (Circle correct number.)

1. YES→ IF YOU ANSWER YES, SKIP TO Q25.
2. NO→ IF YOU ANSWER NO, SKIP TO Q23.

Q22. What if the *final* estimate increased your total safari cost by 5 percent, would you have taken the safari? (Circle correct number.)

1. YES→ IF YOU ANSWER YES SKIP TO Q25.
2. NO→ IF YOU ANSWER NO, SKIP TO Q23.

Handy Table to help with percents. If the total cost of your safari was:

		If Cost is 2500	If Cost is 4400	Your Cost
10%	Drop a zero at end	250̶ = 250	440̶ = 440	______ = ______
5%	Divide 10% answer by 2	250 / 2 = 125	440 / 2 = 220	______ /2 = ______
15%	Add 5% to 10%	125 + 250 = 275	220 + 440 = 660	______ + ______ = ______

Q23. If you decided not to take the same safari at the higher stated cost, what would you do instead? (Circle the best number.

1. CHOOSE A SAFARI AND VACATION ARRANGEMENT WITH THE SAME TOTAL LAND COST YOU PUT IN QUES. 15 BUT HAS LESS DAYS ON SAFARI AND PERHAPS OTHER ACTIVITIES.

   IF YOU SELECT 1, PLEASE SKIP TO Q25

2. TAKE A SAFARI TO A DIFFERENT COUNTRY OR COUNTRIES, *ASSUMING IT DID NOT CHANGE ITS FEES.* IF YOU CIRCLE THIS OPTION, WHICH COUNTRY OR COUNTRIES WOULD YOU CHOOSE?

   ________________________ IS THE COUNTRY OR COUNTRIES I WOULD CHOOSE.

3. OTHER, SUCH AS CHOOSING ANOTHER TYPE OF VACATION. PLEASE DESCRIBE BRIEFLY.

   ____

   ____

   IF YOU WOULD CHOOSE A DIFFERENT VACATION, SKIP TO Q25.

Q24. If the land costs of safaris increase in *all* countries because costs increase everywhere, which option will you now choose? (Circle the most correct number.)

1. TAKE A SAFARI IN KENYA WITH LESS DAYS IN A PARK OR RESERVE WHICH COSTS ABOUT THE SAME AS THE TOTAL COST OF THE PRESENT SAFARI WHICH YOU PUT IN QUESTION 18.

2. TAKE A SAFARI TO A DIFFERENT COUNTRY EVEN IF IT NOW COSTS MORE.

3. TAKE A VACATION WHICH DOES NOT INCLUDE A SAFARI.

4. OTHER. PLEASE DESCRIBE BRIEFLY.

   ____

   ____

Q25. Park managers can charge *uniform* entrance fees for each park and reserve or charge entrance fees in each park *in proportion* to the costs of preservation or in proportion to the availability and uniqueness of animals in the park. Given that revenues must be increased, which principle do you *most* prefer? (Circle the appropriate number.)

1. CHARGE FEES UNIFORMLY.

2. CHARGE FEES IN PROPORTION TO *COSTS.*

3. CHARGE FEES IN PROPORTION TO *AVAILABILITY AND UNIQUENESS OF SPECIES.*

A FINAL QUESTION ABOUT VALUING PARKS/RESERVES:

Q26. Think back to when you decided to take the safari. What *maximum amount* added on to your original cost, would you be willing to pay to keep all the Kenya parks on your itinerary (in their approximate present condition) rather than change your plans?

MAXIMUM ADDITIONAL AMOUNT______________IN___________ CURRENCY.

Q27. *Please circle the primary reason why this is the maximum additional amount that you would be willing to pay.*

a) Further increases in charges would be unfair or unreasonable.

b) Additional charges would not be applied to park conservation, or other useful purposes regarding the park.

c) There are more satisfactory things to spend the extra money on.

d) Another reason. Please specify:

____.

Q28. Park managers would like to know how adequately you think they are providing park and reserve related services. (Circle the number of the most appropriate response)

	Not Very Very Adequately						Adequately	No Opportunity to Observe
PARKS AND RESERVES	1	2	3	4	5	6	7	8
LODGING	1	2	3	4	5	6	7	8
GUIDES	1	2	3	4	5	6	7	8
ROADS	1	2	3	4	5	6	7	8
POLICING	1	2	3	4	5	6	7	8
CONGESTION AROUND GAME	1	2	3	4	5	6	7	8
INFORMATION	1	2	3	4	5	6	7	8
ATTITUDE OF STAFF AT GATE	1	2	3	4	5	6	7	8

FINAL SECTION: Finally, we have a few questions about yourself or family. The answers you give are confidential. We will only report averages from the hundreds of people surveyed.

Q29. How many safaris have you taken in the last ten years? ________________safaris.

Q30. Do you plan to take another safari in the next ten years? 1. YES_____. How MANY___________?

2. NO.

Q31. Country of Residence _____________. City of Residence _____________.

Q32. Next, in what year were you born? __________

Q33. Occupation? ______________________

Q34. What is the last grade of formal education you have completed? (Circle one number.)

1. SOME HIGH SCHOOL OR LESS
2. HIGH SCHOOL GRADUATE OR EQUIVALENT
3. SOME YEARS OF UNIVERSITY OR COLLEGE, BAC OR TECHNICAL SCHOOL
4. DIPLOMA OR DEGREE FROM A UNIVERSITY OR COLLEGE OR TECHNICAL SCHOOL.
5. FORMAL EDUCATION BEYOND THE UNDERGRADUATE DEGREE OR EQUIVALENT.

Q35. Your sex?

1. MALE
2. FEMALE

Q36. Please check which category best describes your family's annual income from all sources. Please be sure to tell us the currency of your answer.

IN ________________ CURRENCY

1.	Less than 25,000	6.	150,001-200,000	11.	400,001-450,000
2.	25,001-50,000	7.	200,001-250,000	12.	450,001-500,000
3.	50,001-75,000	8.	250,001-300,000	13.	500,001-550,000
4.	75,001-100,000	9.	300,001-350,000	14.	550,001-600,000
5.	100,001-150,000	10.	350,001-400,000	15	600,000-1,000,000
				16.	1,000,000-2,000,000
				17.	Over 2.000.000

Q37. **Are you or any member of your family a member of a conservation group? (Circle number of best answer.)**

1. YES
2. NO

Q38. **How do you rate the overall experience you have had in Kenya thus far?**

1. UNSATISFACTORY
2. SATISFACTORY
3. VERY SATISFACTORY
4. EXTREMELY SATISFACTORY

Thank you for your time and cooperation. Any additional thoughts or comments are welcome; please add them here. If you want a copy of the study result, please include your name and address.

r **Yes, I want a summary of the results. My name and address are:**

Chapter 6

Investing in Biodiversity: An Economic Perspective on Global Priority Setting

6.1 Introduction

The previous chapters have reviewed many of the problems related to the valuation of biological resources. The failure to quantify comprehensively the benefits side of the conservation equation complicates the determination of a socially optimal level of investment in biodiversity. This is the basis of the argument that market failure will bring about sub-optimal investment levels, a pattern which holds true at national and global scales and in all countries irrespective of their development status.

This chapter focuses mainly on global market failure and examines the issue of how to prioritize investments on a global scale and thereby move in the direction of a notional first best. Increasing awareness of environmental degradation has heightened concern for the world's biodiversity. Although funding prospects are improving, there is a limited budget for global biodiversity conservation. The total resources allocated to the Global Environment Facility (GEF) the conduit for global environmental good transactions, is limited. Hence some form of ranking device is needed to prioritise investments both at the global level and in terms of allocating national conservation budgets.

From a biological perspective, chapter two noted that regional prioritisation on the basis of species rich hotspots (e.g. Myers 1990) may be inappropriate. Furthermore such lists ignore important socioeconomic factors which should impinge on area selection. Recent attempts to go beyond hotspots (Dinerstein and Wikramanayake 1993; Sisk *et al* 1994) and the application of optimal and heuristic area selection methods (see chapter two), have provoked considerable interest among funding agencies bemused by the apparent intangibilities of biodiversity projects. A criticism of these and earlier studies is that they either omit or do not explicitly address important considerations such as the cost of an intervention, or they do not develop an "absolute" ranking system such as an index. A question increasingly being raised as a matter of urgency by funding agencies and developed parties to the Convention on Biological Diversity, is where is it is best to invest in biodiversity projects to achieve maximum "value for money".

This chapter is in two parts. Initially the problem of global market failure is revisited and biodiversity is briefly contrasted with another global environmental problem, that of climate change. This discussion raises several points relevant to the exercise of determining the different requirements for cost effective investment. Then, taking the perspective of an omnipotent global finance conduit, the issue of global biodiversity investment is addressed. This involves the development of a single biodiversity investment "cost-effectiveness" index which incorporates several relevant socio-economic factors and can be used to rank countries (Moran *et al* 1996). The problem is approached using conventional investment appraisal criteria; accounting for investment cost and biodiversity benefit as maximised through a representative surrogate such as species richness. Biodiversity investments are typically accorded a presumptive exemption from conventional appraisal criteria. Typically this allows appraisal to dispense with issues which are ordinarily central to investment decisions. Unsurprisingly, biodiversity investments are therefore viewed in some circles with suspicion or thought of as wooly, another possible reason why investment may not occur. The cost-effective priority investment index (CEPII) proposed here is derived by combining a basic cost-effectiveness ratio with information describing threat and the probability of a successful intervention. The use of these elements addresses the problem of how to assess the expected returns in conventional project appraisal criteria. The importance of this is emphasised by the findings from the previous two chapters. For region or country-wide assessments which are the scales being considered by many funding organisations, the role that valuation methods can play (beyond being purely demonstrative) is limited. Focusing on the effectiveness of international spending, the CEPII is applied using country data for the Asia-Pacific region, Sub-Saharan Africa and Latin America and the Caribbean. The resulting ranking is compared with those of previous studies and relevant caveats discussed.

6.2 Biodiversity as a global good

As discussed in chapter one, the sub-optimal rate of habitat conversion (and by extension the the negative externalities of biodiversity loss) are related to the public good nature of biodiversity. This complicates the theoretically optimal investment decision determining the amount each country should conserve in its own interests and with regard to the global good.

The optimum is encapsulated in the equalisation of a notional global marginal cost and benefit schedule where the latter is best thought of as the total damage avoided from the loss of biological diversity (widely defined to include the loss of value to anyone for any reason). This approach is obviously fraught with difficulties and previous chapters only dealt in a highly limited way with one, albeit important, part of the question -valuation of the benefits. Furthermore, it is possible to dissaggregate the benefits and costs accruing to individual countries to see that the improvement implied by the move toward an optimum may not be a Pareto improvement. In other words, an unassisted adjustment may imply that particular countries are required to suffer substantial welfare losses (combined opportunity costs and direct and indirect costs) for the benefit of the world in general[1]. Indeed, as discussed in the Kenyan case, the current pattern of global conservation options is characterised by a mismatch of costs and benefits (Wells 1992) frequently to the detriment of poor biodiversity-rich nations. The economic reasoning for this can be summarised using figure 1, which shows the notional global optimum level of conservation defined by hypothetical global marginal cost - the horizontal summation of the marginal costs of an industrial country I, and developing country D - and a global benefit curve which is the vertical summation of benefits to D and I.

To simplify, the case shown here is one implying equal conservation costs between developing and Industrial countries, an abstraction which will be returned to below. On the benefit side, the shape assumed here arises from the standard economic reasoning of a declining marginal utility of money. Furthermore, a comparative advantage in cost-effective scientific exploitation of biological resources in the developed world gives the same resources a higher value in developed countries (the reason why the marginal benefit curve for the industrial country is at all points higher than the developing country).

Left to its own devices, the optimal approach for D would be to conserve where marginal costs and benefits are equalised and similarly for I. The outcome is a level of conservation which may be less than the global optimum. For D, the level will be something like Q_D while for I the relevant level will be Q_I. The sum of these may fall below Q_O, the global optimum. Hence again, the familiar market failure. Interestingly as drawn, this state of affairs is likely to be cost inefficient in the sense that marginal costs of conservation will not be equalised across all countries. This is a first order condition for a cost-effective solution ensuring that the lower cost country conserves most of the cheapest - and most probably the best - biodiversity. This quality element is a matter of some importance and in some sense a differentiating factor between the global environmental problems of biodiversity loss and climate change.

Biodiversity and climate change contrasted

The problem of greenhouse gases emission control has been assessed as a cost-benefit decision in stylized models of optimal abatement (Fankhauser 1995). In the global context, countries can to varying degrees be considered as price takers in the sense that only exceptionally large states can

[1]To the extent that the increment in the global benefits accruing to the adjusting state is generally insufficient to off-set the implied domestic loss.

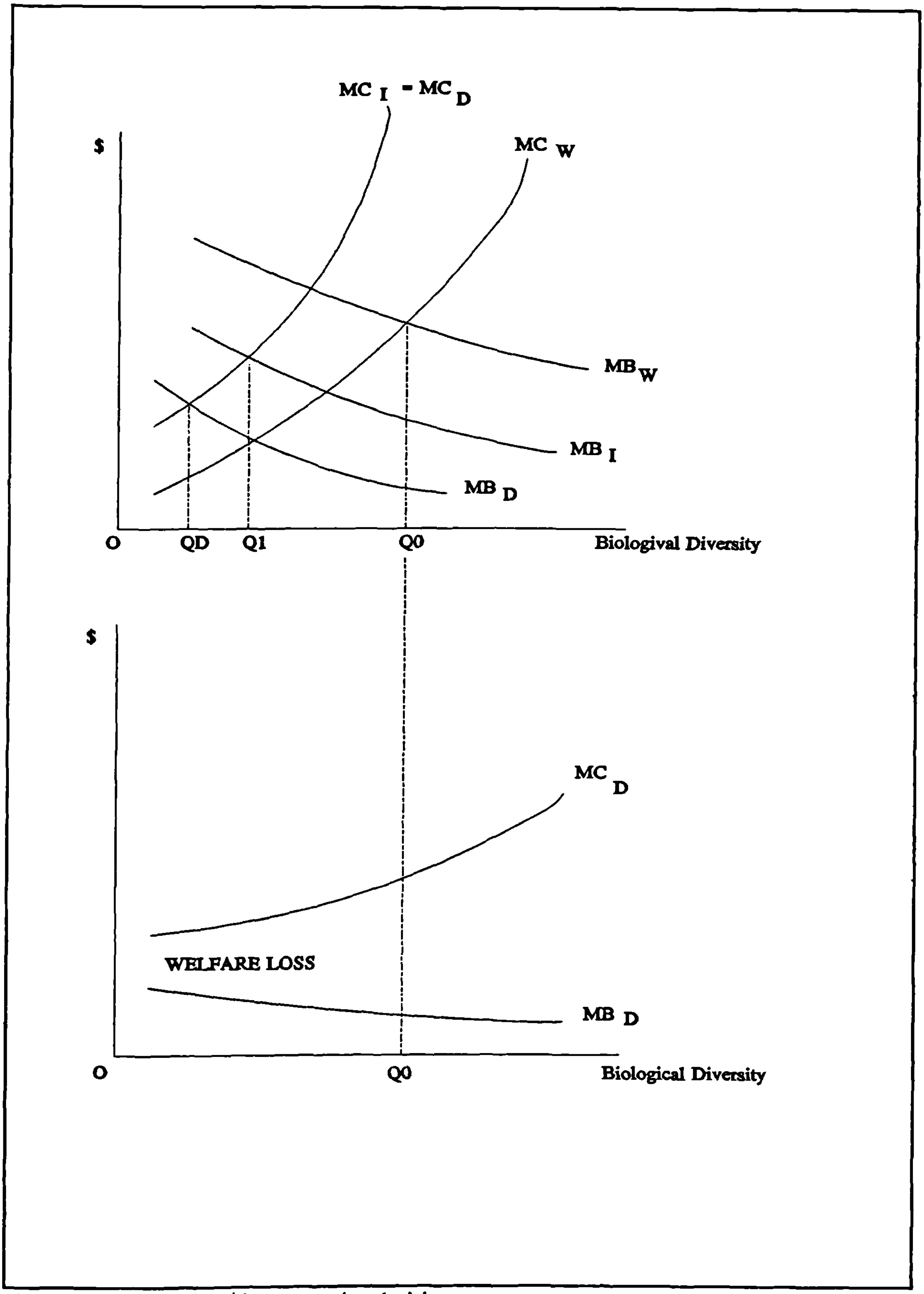

Figure 1: Global and local conservation decisions.

make the climate and the shadow benefit of abatement is the same wherever it happens. At a local level countries can also mitigate and it is typically this strategy which differentiates rich from poor. With biodiversity the same global and national benefits dichotomy appears to hold except that poorer species rich countries can now regard themselves as price makers in terms of the global damage which developed parties cannot independently mitigate[2]. Thus even the smallest states can potentially dictate global damages if they possess some treasured endangered species. This asymmetry is attenuated by two factors which determine the national decision. The first, which has already been discussed at several points, is the extent of any domestic benefit accruing irrespective of the global significance of the host country's biodiversity. Second is the availability and mobilisation of global funds to compensate developing countries for doing what developed countries want. In essence, the idea here is the mitigation of damage costs *in situ*. This can also address the cost inefficiency issue which arises because lower cost diversity opportunities are available in D rather than I (that is 'I' will undertake conservation up to a point where $MC_I > MC_D$). Mutually beneficial gains from trade are therefore to be had by direct transfer payments or by the deployment of instruments that jointly implement the global optimum[3] by buying cheaper conservation services in developing countries. In figure 1 this amounts to making good some of the welfare loss implied by the optimum Q_O, incurred by the developing party which is somehow obliged to meet its contribution independently.

However, that these gains do not occur spontaneously is mainly due to the public nature of biodiversity and individual developed country incentives to free-ride on the benefits side payments. Furthermore, the peculiarly cross-border nature of such deals raises a whole raft of principle agent and moral hazard problems as well as basic sovereignty issues which currently limit the role of economic instruments. It is into this void that the Global Environment Facility has been placed as a as an interim financing conduit for the global conventions through which the global optimum may be enacted. The existence of such a body is the point of departure for this chapter and some background on its limited mandate illuminates some of the institutional restrictions which unfortunately add to all those previously covered.

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[2]Countries can cope to the extent that they can exploit existing *ex situ* stocks and conserve whatever they happen to possess *in situ*. However the key element in the obvious desire not to make do with these resources is the fact that some countries are product differentiated hot-spots (see for example Caldecott *et al* 1994).

[3]This process of joint implementation is commonly explored in the context of transboundary pollutants such as carbon dioxide and sulphur emissions.

6.3 The Global Environment Facility: raison d'etre and brief history

The GEF is a young institution and given the nature of the problems with which it grapples its growing pains have hardly been surprising[4]. In its first operational phase it will spend around $2 billion over three years (1994-97)[5]. Only part of this will be spent on biodiversity- hence the need to consider priorities.

The purpose of the GEF is to invest in the developing world in order to capture the global environmental value of investments, policies and capacity building. Its remit is to do this in the context of biodiversity, global warming, international waters and the ozone layer. Its actions on the ozone layer involve funding substitutes for chlorofluorocarbons (CFCs) and this it does by meeting the difference between the cost of the substitutes and the original cost of CFCs - the so-called incremental cost. Its activities 1991-94 in the areas of biodiversity and global warming took place outside the scope of the Rio Conventions which were not signed until 1992. From 1994 onwards it must act via the Conventions of which it is the interim financial mechanism. No international agreements exists on its activities in international waters.

It is important to understand that the GEF is not a development agency as such. It operates via many development projects, but it modifies them so that the technologies used are cleaner than they otherwise would have been. Its purpose is not development as such, but the capture of global environmental value - the value that comes from reducing the 'global bads' of climate change, biodiversity loss and ozone layer depletion.

This specific function shows up in the way the GEF decides how much to spend. It funds only the incremental cost of a project. For example, imagine a developing country would have burned coal for electricity, but that the option to burn more expensive gas is available. From a development perspective it is probably better to burn the coal since it is cheaper. Coal burning becomes the 'baseline' activity. But gas would be cleaner from a global environmental point of view (it has lower carbon dioxide). So, the GEF would consider funding the difference in costs - the incremental cost. It is possible to consider analogous incremental cost activities for biodiversity conservation. In terms of figure 1, the lower section implies a welfare loss for the developing country somehow forced to

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[4]For a somewhat limited critique of the achievements of the GEF pilot phase biodiversity projects see Mittermeier and Bowles (1993).

[5]Pending the decision on a key US contribution. A pilot phase 1991-4 accounted for approximately $0.7bn of expenditure.

comply with the global optimum level of conservation. Thinking of these curves as representing the returns to a potential conservation project which increases the amount of conserved biodiversity from Q_D to Q_O, the function of the GEF is to make a transfer which leaves the developing country at least indifferent between the two states. In other words, the transfer is akin to moving the intersection of the marginal cost and benefit curves along the horizontal axis. As the GEF stands, the fact that the movement of both curves defines the new point of indifference (as opposed to the marginal cost curve simply shifting downwards) is of considerable relevance. As mentioned above, the purely domestic benefits from conservation are an attenuating factor in a host country's conservation calculations vis a vis the globe. The same is true for the conservation increment which is attributable to GEF funds and this issue has in recent GEF proceedings caused considerable debate. In essence, the issue concerns whether these domestic benefits should be identified and deducted from the GEF transfers given that they should in any case be a domestic developmental (and by extension spending) priority. But the counter argument is that such deductions literally do leave countries indifferent about taking on global responsibilities. This argument has even more weight when despite GEF attempts to be fully incremental, most countries remain suspicious of further 'hidden' incremental opportunity costs related to administration and monitoring etc. In an ideal world, abstracting from the domestic benefits would have the GEF-induced marginal cost curve falling to meet the marginal benefit curve of figure 1. This abstraction also circumvents the tricky non market valuation problem implied by the need to deduct benefits. As it happens however, a recently agreed upon simplifying assumption makes the GEF task a bit easier in this area. The heuristic compromise is that only obvious incidental domestic benefits clearly relevant to the national baseline should be liable as deductions.

Basically the GEF rule for intervention should therefore be that the global value obtained exceeds the incremental cost. Since global value is typically not expressed in money terms, the approach tends to be based on cost-effectiveness, e.g. $ per tonne of carbon emission avoided.

The carbon example is relevant to biodiversity because the GEF is at liberty to fund both afforestation projects and projects that avoid deforestation project that are likely to achieve joint benefits. Calculating the baseline and alternative profiles for carbon emissions is not easy, but it can be done because countries typically have well-documented energy plans. As far as biodiversity is directly concerned, again, it is somewhat difficult to put exact numbers to the basic GEF criteria. Typically, things that are by consensus globally significant are deemed GEF- relevant. Incremental cost will consist of the difference in the costs of biodiversity conservation in the baseline and the cost of some intervention. Since biodiversity conservation is not a priority for many developing countries, the whole cost of some interventions will constitute incremental cost -the baseline cost is effectively zero. But many countries have biodiversity plans and these may make up the baseline. In practice the process of identifying the baseline, which is essentially the tasks which avoids GEF duplicating either existing government action or other conventional development projects is a complicated task in project identification[6].

Note the difference between the global warming interventions and the biodiversity cases. In the former case, no *extra* energy is supplied. A given amount that would have been supplied anyway is supplied in a different, cleaner way because of the GEF intervention. In the case of biodiversity it is not a matter of 'supplying' the same amount of biodiversity at a different cost, but of ensuring that *more* biodiversity is saved than otherwise would have been the case. As it happens, the interpretation of the terms of the Convention on Biodiversity may hinder implementation in the unambiguous way of the Framework Convention on Climate Change. For example, explicit reference to the global value of biodiversity, and cost-effectiveness in ranking interventions are notably absent, and this may complicate the calculation of incremental costs and the potential for saving the largest amount of global diversity per dollar.

In its pilot phase GEF (or more correctly its implementing agencies in the World Bank, UNDP and UNEP) gave limited attention to the practicalities of incremental cost financing. The remainder of this chapter is interested with furthering the basic GEF objectives which as summarised by the movement towards the global first-best conservation level.

The GEF role as an investor is invariably partially determined by political considerations. However, it is useful to address the type of information requirements for conducting an analysis of global priority investments. This exercise inevitably touches on much of the analysis of previous chapters, in particular, that relating to the choice of biodiversity surrogates and optimal area selection (chapter two). Data requirements are a limiting factor to the precision of such an exercise. In general this restriction implies a trade-off in the resolution of both the biological and socioeconomic data used.

6.4 Investing in biodiversity

The uniting consensus of the signatories to the Biodiversity Convention is the need to maintain

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[6]This statement is based on the author's personal experience assisting UNDP - an implementing agency of GEF projects - to formalise identification methods for incremental cost projects.

maximum diversity, consistent with the goal of keeping future options open (Humphries *et al* 1995). This imperative raises two problems which the current index attempts to address. Firstly, there is an unavoidable funding constraint which precipitates the 'agony of choice'. Saving all biodiversity is technically impossible; but a cost-effectiveness rule is theoretically consistent with maximum conservation using a finite budget. Secondly, as seen in chapter two a consensus 'currency' to assess the relative potential of competing intervention strategies is absent. In policy terms the message from that chapter was that saving maximum biodiversity will remain a fuzzy goal in the absence of a practical objective. But the exacting data requirements for the use of recently-developed heuristic and optimal algorithms will make these unattainable for many developing countries for some time. A precautionary approach to biodiversity conservation stresses the importance of pre-emptive action. An index approach therefore attempts to reconcile the requirements for an operational economic assessment for priority setting using the species richness surrogate for the countries we wish to rank. Chapter two described the biological properties of species and higher taxon richness measures and the main challenge is to find congruent socioeconomic information. The data used here are by no means perfect, but highlighting information gaps is in itself a partial objective of a priorities exercise.

6.5 Cost-Effectiveness

The limited scope for environmental valuation methods suggests ranking interventions on a cost-effectiveness criterion CEA. The ranking considered here is at the country level and may be formulated in terms of available information on costs, and effectiveness defined in non-monetary terms. In contrast to CBA, CEA introduces a degree of judgement into investment ranking, in that an index might be formulated in terms of any of a number of competing (non-monetary) measures used to describe biodiversity. The issue for the conservation community is therefore to select a best diversity maximand from several suggested definitions.

The elements of the basic cost-effectiveness ratio:

$$\rho\frac{\Delta B}{C}$$

are ρ, $0<\rho<1$, a composite threat/success indicator, C = cost and B = benefit. These elements are discussed in turn.

6.6 On a suitable currency B

The race to guide conservation priorities has produced an array of measures for biological diversity covering among others, measures representing functional diversity (Walker 1992), and measures using different levels of phylogenetic pattern to represent genetic or phenotypic diversity (see chapter 2).

Debate continues on the ultimate location of option value and, to a lesser extent, the economic implications of focusing on anything below species level. A more pragmatic approach here, focuses on surrogates; varying in complexity from centres of endemism (ICBP 1992), use of higher-taxon richness (Williams and Gaston 1994), or the use of environmental (pattern) diversity linked by models to target characters or attributes (Faith and Walker 1996 submitted a,b). Surrogates for priority setting are typically dictated by data availability and the ease with which the same information may be gathered and ground-truthed. Early global exercises have focused on species richness and endemism (Myers 1988,1990, Mittermeier 1988, Mittermeier and Werner 1990, Dinerstein and Wikramanayake 1993, Sisk *et al* 1994, WCMC 1994a). The global scale of the current analysis is restricted to the same resolution, and, accepting that all measures are value-laden, we make no unwarranted claims for species richness other than the potential advantages claimed by Williams and Gaston (1994).

For the countries we consider, B is therefore represented by the sum of higher plants, mammals, birds, reptiles and amphibians, with data drawn from WCMC (1994) (Table 1). Ideally these data should be subjected to the same efficient selection approaches such as complementarity (see chapter two). However optimal approaches have so far been limited in their use of economic information. Firstly, the studies demonstrating complementarity typically imply a constant inter-country area-cost relationship, which in many parts of the world is as erroneous as assuming a fixed species-area relationship. Optimal linear programming can restrict area sets to a budget constraint by maximising the species complement divided by the unit cost of successive selected sites (ie the marginal cost) of conservation). A budget constraint may alter area selection

Table 1 species richness

Country	Higher Plants	Mammals	Birds	Reptiles	Condition	TOTAL
Bangladesh	5000	109	354	119	19	5601
Bhutan	5468	109	448	19	24	6068
Cambodia	n/a	117	305	82	28	532
China	32200	394	1244	340	263	34441
Fiji	1628	4	87	25	2	1746
India	16000	316	1219	389	206	18130
Indonesia	22500	436	1531	511	270	25248
Korea, Rep.	2898	49	0	18	13	2978
Lao P.D.R	8286*	172	651	66	37	9212
Malaysia	12500	286	736	268	158	13948
Myanmar	7000	300	867	203	75	8445
Nepal	6973	167	629	80	36	7885
Pakistan	4938	151	476	143	17	5725
Papua NG	11544	242	578	249	183	12796
Philippines	8931	166	395	193	63	1748
Solomon	3172	47	163	57	15	3454
Sri Lanka	3214	86	221	144	39	3705
Thailand	12625	265	915	298	107	14210
Tonga	463	1	39	6	0	509
Vanuatu	1000	12	84	22	0	1118
Viet Nam	8000*	273	638	180	80	9171
W. Samoa	693	3	44	8	0	748

Notes:

* All figures are from WCMC (1994b) except for the Lao P.D.R. and Viet Nam bird figures which are from Dinerstein and Wikramanayake (1993).

n/a Data not available

significantly. Second, while methods are theoretically consistent with the maximisation of global option value for a given attribute, the prediction of global complementarity-based area selection may conflict with domestic priorities (and action plans).

Note that the cost-effectiveness ratio represents the incremental amount of biodiversity conserved by an intervention as ΔB. A problematic requirement is that this should ideally measure the marginal cumulative diversity accounting for in existing areas. Even using species as the 'characters' of interest (and given the caveat on blurred species boundaries (Rojas 1992)), this is precisely the combinatoric nightmare Weitzman (1992) was attempting to illustrate. While there may be possible proxies to measure "biodiversity saved" at the project level, national conservation strategies typically forego global complementarity by focusing on biodiversity currently in place. Noting the dangers of assuming a constant species-area relationship, the favoured measures of richness and endemism are normalised to be on a km^2 basis[7].

6.7 Investment cost C

There are numerous cost considerations in area selection, though this single most limiting factor is rarely addressed in priority exercises. Biodiversity priorities are set within very real budget constraints, and it makes sense for any priority exercise to maximise diversity while simultaneously minimizing cost[8].

For index purposes C is ideally required to be the marginal unit cost per biodiversity intervention within countries. Ideally this might represent the cost of land purchase for reserve creation. But since it is impossible to categorize the variety of biodiversity investment costs, international investment levels are taken as a best proxy to cost information for countries ranked in this exercise. Therefore the necessary assumption is that investment levels normalized per km^2 give some indication of the necessary cost of intervention. It is clear however that the figures used may be influenced by factors other than cost, not least socio-political stability and a country's conservation track record. Alternative cost interpretations are reconsidered later.

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[7] Although we concentrate on tropical forestry, environmental heterogeneity may be increased with spatial scale and location as well as habitat type.

[8] Note that this is not necessarily the same as minimizing the area of a set of arbitrarily selected sites which maximise diversity.

Several investment series are available (accounting for development assistance (ODA) investments by bilaterals, multilateral and NGOs). The proportion of world ODA currently spent on biological diversity is not known with certainty. Estimates are low, and do not adequately reflect the recent emphasis on biodiversity issues nor the emerging role of GEF spending.

The immediate concern for international donors, such as the GEF, is to be able to establish where available funds would be most cost-effectively spent to achieve the best results for biodiversity. The investment data we require are therefore a total of all international investments made in particular countries. Available indicators are crude. Abramovitz (1993) provides a survey of all global investments by United States institutions for 1991. The data are for the United States only however, and are therefore not ideal for our purposes.

The World Conservation Monitoring Centre (WCMC 1995) has compiled a comprehensive database of all international financial investments in biodiversity conservation. The objective being to quantify expenditure by bilateral and multilateral aid agencies, national governments, and non-governmental organisations in a selection of countries. The WCMC project can be broadly split into international (foreign expenditure) and national (own expenditure) levels of investment, with a related component gathering financial and staff resources for protected areas by country.

The primary cost data requirement for the index is international investment, given by bilateral, multilateral and NGO expenditure. It can nevertheless be argued that national costs (given by government expenditure, and other national expenditure, such as private corporate responsibility investment) are also relevant. National expenditure can be seen as indirectly benefiting the cost-effectiveness of the international dollar spent; higher national expenditure should lead to better infrastructure. Good infrastructure will in turn provide a foundation on which the foreign dollar can build.

The index should still seek to determine the cost-effectiveness of foreign investment; but should perhaps incorporate national expenditure in some other way. The ideal cost data would therefore include national and international expenditure per country; separable into investments on certain habitats and in certain years (for maximum flexibility). The WCMC database, when available, will provide such a breakdown. In the meantime, the cost figure should be expressed as international expenditure, with attention paid to ensuring that the data are consistent across countries (same year/s, same donor classification). It is of note that other sectoral projects not specifically classified as biodiversity projects may nevertheless have biodiversity components. Care is required in compiling information distinguishing general project aid from that directed solely to biodiversity.

The present analysis uses Abramovitz 1993 data (total US institutional spend for 1991) or recently compiled data by WCMC 1995-unpublished) on total funding worldwide 1991-94 on biodiversity conservation. Because of the way information was reported in response to the WCMC questionnaire which gathered data on worldwide miscellaneous conservation expenditures, it was frequently necessary to annuitise lump sum cost information to achieve consistency over annual expenditures.

6.8 The Cost-Effective Priority Investment Index (CEPII)

The CEPII combines the basic cost-effectiveness ratio with the attenuating factors in ρ, which combines factors describing threat and the probability of a successful intervention. These interrelated factors are discussed after the description of the index.

In order to calculate the index data are required for:

A	:	Total country area in km^2
Af	:	Forested Area in km^2
Ap	:	Protected Forest Area in km^2
Au	:	Unprotected Forest in km^2 (calculated as Af minus Ap)

This will provide sufficient data to calculate the probability of success. We now need,

k	:	either the deforestation rate or the population growth rate
B	:	biodiversity measure, species richness or endemism
C	:	cost given by world investment

The threat ratio, species and cost per km^2 of forested area, and consequently the index, can now be calculated.

Let the probability of a successful intervention be:

Where,

$$Ps=\frac{Ap_t}{A_t}*\frac{Au_{t+n}}{Au_t}$$

A_t = total protectable area at time t

Ap_t = area protected at time t

Au_{t+n} = unprotected (protectable) area in year t+n, and,

$0 < Ps < 1$, Now,

$$Au_{t+n}=Au_t(1-k)^n$$

where k represents the rate of growth of a selected threat, given by the rate of land conversion (estimated using either the deforestation rate or the population growth rate).

The term on the right hand side measures existing (and past) commitment. The second term attenuates the first by the degree of threat.

The final index is:

$$E=\frac{Ps*\Delta B}{C}$$

$$E=\frac{\frac{Ap_t}{A_t}*(1-k)^n*\Delta B}{C}$$

where B = biodiversity measure and C = cost measure as discussed in the text

This formula defines our index; the cost effective priority investment index (CEPII), of which the attenuating factors previously summarised as ρ need to be determined.

The probability of success

Probability of a successful intervention will depend on institutional characteristics, notably the commitment of the government or its 'willingness to conserve'. Commitment could be reflected by an indicator which assesses the available infrastructure of a country - examples could be total km^2 of roads, staffing in protected areas or in natural resource-related ministries . "Ability" could also determine success. An "ability" indicator might be the number of students graduating in biodiversity-related disciplines in a particular country or more generally, a country's ranking in the United Nations

Human Development Index (HDI)[9]. Various other indicators are possible but most suffer from data problems. A simple indicator is area of "protectable" land actually protected - the probability of success would therefore be given by the ratio of actual terrestrial protected area, over total protectable area, per country.

On the basis of the above success indicator, a country with a high percentage of area protected would have a high probability of success. It should be noted however, that a very high percentage of protected area - assuming none of which exists solely on paper - could indicate that a country has reached protection "saturation" - there may be an economic maximum protection percentage beyond which a country will be reluctant to protect[10]. Wilson (1992) recommends that total global reserves should be expanded from 4.8% (WRI 1992), to 10% of the land surface, to include as many undisturbed habitats as possible. To protect at least 10% of its total land area might therefore constitute a reasonable objective for a country. This abstracts from the characteristics of these areas; that is, how fragmented or otherwise skewed in terms of biogeographical representativeness they are. We emphasise that such factors as well as size of remaining areas (protected or otherwise), matter because larger areas can withstand higher deforestation rates and may not require immediate interventions. Similarly It should be noted, however, that size of the intervention unit though difficult to incorporate into an index, will also bear on the probability of success (as stressed in Dinerstein and Wikramanayake 1993).

Data used needs to ensure congruence between our surrogates for biodiversity, success and threat (see below). Various data sources are used to identify protected forest area as the basic unit of analysis for countries we rank. Concentration on these biota is partly based on the assumption that a larger proportion of species diversity can best be safeguarded in forests. This proposition is debateable though we do not pursue it here (see Myers 1988, Raven 1988 for further discussion).

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[9] For a review of the HDI which is a composite index of life expectancy, educational attainment and material living standards see McGillivray and White (1993).

[10]No such biological saturation can apparently exist if the character basis of value is constantly reduced by biological insight.

Table 2 Forest area and threat rates

Country	Total Country Area[a] (square km)	Natural Forest Cover[b] (square km) (1990)	Deforestation Rate % (1981 - 1990)	Population Growth Rate[e] % (1995 - 2000)
Bangladesh	144,000	7,690[c]	3.9[c]	2.6
Bhutan	47,000	28,090[c]	0.6[c]	2.3
Cambodia	181,000	121,630[c]	1.0[c]	1.8
China	9,561,000	1,150,600[d]	3.9[d]	1.2
Fiji	18,000	8,100[d]	0.1[d]	1.4
India	3,288,000	517,290[c]	0.6[c]	1.9
Indonesia	1,905,000	1,095,490[c]	1.0[c]	1.6
Korea, Rep.	99,000	8,000[d]	0.0[d]	0.8
Lao P.D.R	237,000	131,730[c]	0.9[c]	2.6
Malaysia	330,000	175,830[c]	2.0[c]	1.9
Myanmar	677,000	288,560[c]	1.3[c]	2.0
Nepal	141,000	50,230[c]	1.0[c]	2.3
Pakistan	796,000	18,550[c]	3.4[c]	2.8
Papua NG	463,000	360,000[c]	0.3[c]	2.2
Philippines	300,000	78,310[c]	3.3[c]	2.1
Solomon	27,000	24,400[d]	0.0[d]	2.8
Sri Lanka	66,000	17,460[c]	1.4[c]	1.1
Thailand	513,000	127,350[c]	3.3[c]	1.3
Tonga	700	< 1,000[d]	n/a	1.4
Vanuatu	12,000	2,400[d]	1.7[d]	3.2
Viet Nam	330,000	83,120[c]	1.5[c]	2.0
W. Samoa	3,000	1,400[d]	n/a	1.0

Notes:

[a] Braatz and others (1992), Table A1 p. 48 (using data from World Bank 1990 and WCMC 1992).
[b] FAO (1993) defines forest cover as including tropical rain forest, moist deciduous forest, dry deciduous forest, very dry forest, desert zone, and hill and montane zone.
[c] FAO (1993) for the year 1990, Table 4b.
[d] FAO (1988) as cited in Braatz and others (1992).
[e] WRI (1992) except for Tonga and Vanuatu, which are from Braatz and others (1992), Table A1 p.48.
n/a Data not available

Table 3: PROTECTED AREAS

Country	Total Protected Area [a] (square km)	Protected Wetlands [b] (square km)	Protected Forest Area [c] (square km)
Bangladesh	968	355	613
Bhutan	9,061	5	9,056
Cambodia[d]	0	20	0
China	283,578	20,000	263,578
Fiji	53	n/a	53
India	137,701	15,300	122,401
Indonesia	192,309	29,000	163,309
Korea, Rep.	7,568	58	7,510
Lao P.D.R	0	0	0
Malaysia	14,880	64	14,816
Myanmar	1,733	40	1,693
Nepal	11,260	261	10,999
Pakistan	36,550	1,380	35,170
Papua NG[d]	290	6,000	290
Philippines	5,729	761	4,968
Solomon	0	n/a	0
Sri Lanka	7,837	766	7,071
Thailand	55,140	410	54,730
Tonga	0	n/a	0
Vanuatu	0	n/a	0
Viet Nam	8,975	495	8,480
W. Samoa	0	n/a	0

Notes:

[a] WCMC (1992) as cited in Braatz and others (1992). Total protected areas for IUCN Categories I - V, Table A2 p. 49.

[b] Scott and Poole (1989) as cited in Braatz and others (1992).

[c] The figure for protected forest areas is obtained by subtracting protected wetland areas from total protected areas.

[d] There are inconsistencies in these data, highlighting the need for more accurate figures.

n/a Data not available

The main reason for limiting the analysis to forests is data limitation, and several problems were encountered. Forests are defined (FAO definition) as including tropical rain forest, moist deciduous forest, dry deciduous forest, very dry forest, desert zone, and hill and montane zone, and excluding plantations. Species, protected areas and cost data are usually presented as totals across all habitats. As our analysis focuses on forests we would need species richness for forests, protected forest areas, and investment on forests for complete accuracy. Complete data for these categories are not available. For the purpose of this analysis an estimate for protected forests was derived by subtracting protected wetlands from total protected area (tables 2 & 3). Recent WCMC figures for protected forest areas (WCMC 1994b) have been calculated by overlaying digitised maps of protected areas (IUCN management categories I - V), with those of forest cover and measuring the degree of overlap. However, the WCMC Biodiversity Map Library does not as yet include 100% cover of the protected areas for countries we rank and gives only partial validation for the protected forest area figures used in the analysis.

The degree of threat

Consideration of threat is more problematic. The need to intervene may be greatest where the threat is highest, but the chances of success may well vary inversely with the threat. As such the appraisal and interpretation of threat or vulnerability is highly relevant for designating priorities.

In practice much depends on the reasons for, and imminence of, the threat. Thus, if it is population growth, the intervention is unlikely to lower the underlying threat. If it is institutional weakness (perhaps exemplified by low national expenditure), the intervention has a higher chance of succeeding. Threat analysis thus becomes essential.

The threats to biodiversity are numerous. The most pressing is the loss of natural habitat - mainly to arable cultivation, logging, settlement, poaching and pollution - resulting from human population growth. More proximate causes of erosion such as ill-defined land and resource rights and market and government failures have also been identified (Pearce and Warford 1993). At the extreme, indicators of political turmoil and repression have been correlated with forest conversion in a cross-section of developing countries (Deacon 1994).

Although there are also many "natural" threats to biodiversity (such as floods fire or hurricanes), they occur more or less randomly and are impossible to plan for; making them unsuitable as indicators.

As it is generally the human enterprise causing the loss, the most obvious threat indicator would be population growth and or density. Deforestation rates are also indicative of threat and are in fact frequently based on population predictions. There is considerable debate about the exact definition and extent of deforestation in may parts of the world (see Houghton *et al* 1991). However we note that forest loss is the only type of habitat alteration adequately quantified on a global scale (FAO 1993). Without making any claims for the preponderance of any particular species-habitat relationship, we adopt deforestation as a threat indicator. We recognize that rates are subject to a range of policy variables which are themselves subject to alteration, and that past deforestation may be a poor predictor of current rates. The index is amenable to other threats information such as population or an emerging number of alternative data sets recording global human disturbance (eg Hannah *et al* 1995).

Deforestation data derives from a number of sources, (table 2).

6.9 Previous Index approaches

There are numerous limitations to the traditional qualitative focus of priorities on species rich countries or biogeographic units of high endemism. Firstly, lists produced by such exercises rarely provide a rationale for selecting among competing countries. Secondly, they do not refer to the costs of intervention, and as such they cannot provide absolute priority guidance to the investment decision-maker. Thirdly, those that mention threat or elements enhancing the probability of a successful intervention, normally fail to justify either as a priority ranking device. A high degree of threat for example is typically viewed as a reason to intervene - whereas, in fact, it may constitute a reason for not intervening.

While the influence of a project's socioeconomic environment is now recognized (see for example Braatz *et al* 1993), the lack of a guiding framework to manage the increased volume of information is apparent. This has implications for the efficiency of decision making. The exact motives of funding agencies are uncertain but it is assumed that most subscribe to common objectives primary among which, cost-effectiveness and consistency in fund allocation and a preference for methods which allow priorities to be quickly identified. Objective assessment of competing criteria is not straightforward and the development of indices offers the opportunity of introducing some consistency to the debate.

In a widely-cited application for the Indo-Pacific region, Dinerstein and Wikramanayake (1993) introduce the Conservation Potential/Threat Index (CPTI). Although failing to develop a single comprehensive index, the analysis does consider the extent of protected areas, forest habitats, species richness and deforestation rates. Cost information is used (represented by Abramovitz data and an unexplained reference to GEF funding), but is not related to benefit (species richness or endemism). Moreover the benefit [species] analysis is not related to or attenuated by the potential/threat or cost information used.

Dinerstein and Wikramanayake use the cost (investment) data to interpret how well countries are financed, not how cost effective the investments have been. Bhutan is ranked at the top of Category I in the cost analysis, which means it is well funded. The authors do not, however, state that it should therefore receive the highest priority. Neither do they allocate this priority to the countries receiving the lowest levels of funding - their slightly ambiguous conclusion is that high conservation potential exists in several countries which have received meagre support. We are given no clear indication as to whether existing high (or low) expenditure should warrant more expenditure or less.

Less ambitiously, Sisk *et al* 1994 offer a set of low resolution data-bases to correlate anthropogenic threats (forest loss and population pressure), with species rich biota on a global basis. Indices derived using their Global Conservation Analysis Package relate principally to threats which are not combined into any singe measure. Instead, the authors classify critical cases - one assumes worthy of equal priority treatment - as those falling in the top 25% of any index. Again, although high index rankings coincide for several countries, there is no unambiguous method to prioritise. Nevertheless despite several notable omissions when comparing global with continental rankings the authors claim the method to be as valid as any other currently available for setting global priorities.

The ad hoc nature of these Conservation Potential-Threat studies is indicative of the difficulties encountered in reducing several data sets to a single comprehensive index number which objectively captures underlying interactions particularly those related to success and threat. Alternative mainly GIS (Geographical Information System) - based approaches for amalgamating similar information have been suggested (Olsen and Dinerstein 1994), but their interpretation is no less problematic.

An index should aim to influence spending by a wide range of funding agencies while considering as many relevant parameters as possible. There is a premium on an index which presents information with some clarity, circumvents data restrictions and allows simple but informed choice. There is nothing incontrovertible about the elements used in the following application. The suggested approach is flexible in the sense that it is amenable to adaptation using other surrogates of threat and or success as well as other forms of cost information.

6.10 An application of the CEPII

To facilitate comparison with the studies of Dinerstein and Wikramanayake (*op cit*) and Braatz and others (*op cit*) the mechanics of index derivation are illustrated with reference to data for the Asia-Pacific countries. Index rankings and referenced data sources for Central and South America and Sub-saharan Africa are presented in turn.

Patchiness of site level data dictates the national focus of the current analysis. This is also the relevant scale of decision making by national governments and foreign aid donors[11]. The three regions are however differentiated by broad macroeconomic aggregates, which invariably impact, albeit historically, on the effort invested in biodiversity monitoring and conservation. Since historical performance provides the basis for the CEPII, it seems reasonable to segregate countries which have benefitted from relatively high monitoring effort (SE Asia), from those where conservation investments have traditionally been low (Sub-Saharan Africa). The regions can be differentiated on other grounds. Generally, South-East Asian countries are under severe threat from deforestation. The rate of forest clearance has roughly trebled in Asia since the early 1960's and is still rising. FAO figures confirm that South-East Asia along with Central America, has the world's fastest rates of tropical deforestation (FAO 1993). Both regions should be distinguished from sub Saharan Africa in terms of current domestic funding and possibly the nature of threat[12].

For all three regions, "protectable" area has been specified as "protectable" forest, area protected as protected forest, and unprotected area as unprotected forest. As previously stressed this is partly because of data availability, but also because diversity is strongly linked to forests. In addition, the severe deforestation threat makes an analysis of forested areas in two of three regions particularly pertinent.

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[11]Notwithstanding a recent emphasis on cross boarder initiatives for species corridors and national parks (ie South Africa and Mozambique; GEPRENAF (Gestion Participative des Ressources Naturelles et de la Faune (Burkina Faso, Cote d'Ivoire and Mali).

[12]In the first instance the chosen threat surrogate for the CEPII is deforestation rates. Low or even negligible forest cover (in several countries), combined with high population growth mean that the latter is arguably a better threat surrogate in SSA

For the Asia-Pacific region, countries considered in the analysis are Bangladesh, Bhutan, Cambodia, Fiji, India, Indonesia, (the Republic of) Korea, Lao P.D.R., Malaysia, Myanmar, Nepal, Pakistan, Papua New Guinea (NG), the Philippines, the Solomon Islands, Sri Lanka, Thailand, Tonga, Vanuatu, Vietnam and Western Samoa. These are the same as those surveyed in Braatz and others (1992), except for Kiribati and the Maldives, which have been excluded from this analysis because they have negligible forest cover (Braatz and others). In Central and South America, omitted countries are French Guyana and several Caribbean small island states. For sub-Saharan Africa, data gaps exist for the Central African Republic, Sierra Leone and Togo.

6.11 Index calculation and country rankings

Data requirements for the CEPII are outlined in appendix 1. Notwithstanding several inconsistencies (such as that for Pakistan reporting 18,550 km^2 of natural forest cover, and 36,550 km^2 under protection), they are sufficient for index calculation.

The threat ratio is calculated by dividing unprotected forest in ten years time by unprotected forest now. We therefore need to assume that deforestation rates will remain constant over the next 10 years. The biodiversity indicator is species richness. In line with Dinerstein and Wikramanayake, the analysis is focused at the species level and assumes that all taxonomic groups have equal importance. Species richness is given by the total of certain well described taxa per country.

The index calculation and subsequent rankings are presented in Tables A1-8. which also vary cost and threat data series used in calculation. All the tables are based on the same index formulation described previously. Table A1 shows the mechanics of index calculation from raw data input using the deforestation rate as the threat and the Abramovitz cost data, which, although limited to US funding only, is more comprehensive in that costs for most countries are provided. A priority ranking derived from these data is shown in table A2, which also highlights data gaps. Tables A3 and A4 presents the index ranking derived using the WCMC (1995) cost data based on world wide levels of investment. Table A5 and A6 considers the impact of using the population growth rate as the threat (combined with the Abramovitz cost data), as opposed to deforestation. Tables A7 and A8 present ranking for Latin America and Sub-saharan Africa both using WCMC cost data and deforestation rates.

Asia-Pacific

Regional priorities (table A2), are Pakistan, Bangladesh, China, Vietnam and Sri Lanka. Allowing for the observed inconsistencies in the Pakistan data (an over-inflated success ratio)[13], the index is consistent in selecting 2 countries in the top five in all three versions, while both threat measures recommend the same top five countries in terms of biodiversity cost-effectiveness, albeit in a different order. Only the use of the incomplete WCMC cost data causes the index selection to include three other countries (Korea, Malaysia and Bhutan). This would seem to indicate that a decrease in forests is linked to an increase in population, as the rates are trending together within countries. Care should be taken in making such generalisations, but it is probably fair to say that both deforestation and population growth rate are acceptable surrogates for the threat of land conversion.

The rankings based on different costs cannot, strictly speaking, be compared. The WCMC data are not complete for all countries and the index is therefore not calculated for these. The WCMC cost data is also for a longer period - the costs have been annuitised and discounted for the period 1991 to 1993. Nevertheless, the analysis does show that rankings will differ when global cost data are used for the index instead of just US data:

Although the good result for Pakistan should be treated with caution, it can nevertheless be stated with a certain degree of confidence that Pakistan has a very good protection record and is therefore worthy of investment. Bangladesh achieves a high index figure essentially because of a high species richness indicator (second highest of all countries listed), and a relatively low cost figure (ie low level of investment). This result is interesting because it occurs despite severe deforestation. Bangladesh is almost completely deforested with less than 5% of the original forest cover remaining[14]. Looking at methods to ground-truth index results for specific cases therefore becomes important. How accurate is the species richness indicator ? Has it occurred because of sampling bias (ie collection in favoured areas), or is there genuinely a rich biodiversity resource ? Are most of these species occurring in mangrove swamps or forests ? As there is very little forest area left to work with in Bangladesh - it may be ideal for investment if high biodiversity is concentrated in forests.

China receives a high ranking which is consistent with the Sisk *et al* and ICBP rankings. China's high index is attributable (mainly) to the low level of funding it receive. Of interest, however, is the fact that India is only ranked 7th in our index whereas it receives high priority in some of the other studies. Apart from cost (investment) India and China have very similar ratios except that China is

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[13]It is worth noting, however, that Pakistan would still achieve the highest ranking even if its success ratio were only 0.2.

[14]Beazley, M. (1990).

more threatened. If China and India had the same cost (investment) then India would achieve the higher score ie ranking. This is not the case however and therefore China does well because of cost.

Additional information pertaining to Vietnam is also useful in interpreting its index result. The remaining forests in Vietnam are very isolated, meaning that corridors are very important for biodiversity conservation. Investment might therefore focus on corridor provision.

Despite being quite expensive (high level of funding) and having medium species richness, Sri Lanka does well because of a high success ratio and low level of threat. It is worth noting, however, that population density is extremely high and projected to increase to 500 people per km2 by the year 2125. This might constitute an important additional consideration in an investment decision.

Worthy of mention is that Indonesia and Malaysia, which traditionally score highly in most ranking procedures, do not do particularly well in this one (they are ranked 10th and 11th respectively). This would appear to be due to their relatively low levels of species richness - Indonesia, however may be particularly disadvantaged here because of normalisation. This issue requires further investigation.

Central and South America

Regional cost-effective priorities are El Salvador, Venezuela, Surinam, Panama and Cuba. The rankings for El Salvador, Surinam and Panama are derived from alternative cost figures[15] which are restricted to US investments and may therefore be understating true costs. Several megadiverse nations appear well funded. Interestingly, none of these countries is highlighted by the review of priority methods provided by Sisk *et al*[16].

Sub-Saharan Africa

Regional priorities for sub-Saharan Africa coincide with some of the continent's poorest countries. In order these are: Niger, Mali, Liberia, Nigeria, Rwanda, Angola, Mozambique. Several data problems are noteworthy: Cost figures for Mali, Niger and Liberia are again from Abramovitz. Protected forest area coverage for Mozambique is zero according to WCMC Biodiversity Map library

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[15]Abramovitz 1991

[16]Sisk et al *op.cit.* review the regional priority countries of 4 methods: 1. Regional hot-spots (Myers 1988, 1990 *op.cit.*); 2. Megadiversity countries (Mittermeier 1988 *op.cit.*); 3. Range-restricted birds (ICBP 1992 *op.cit.*); 4. their own areas of critical global or continental concern.

or 3750km^2 according to source tables in Sharma[17]. Information on species richness and the extent of protected forest areas for several countries are derived from a number of sources (see table 6). Of the 7 countries listed, only Nigeria and Angola are considered as globally critical by Sisk *et al.*

6.12 Interpretation of the results

It is important to appreciate that the basis of the rankings in tables A1-4 is the unweighted interaction of biodiversity per km^2, cost (or investment), threat and success surrogates. The ranking of several species poor nations underlines the importance of country knowledge of alternative socioeconomic indicators to corroborate the findings. Aside the obvious data discrepancy due to an over-inflated success ratio, the good result for Pakistan for example, should be treated with caution as there is little reliable information on recent changes in the composition of forest cover[18]. Bangladesh achieves a high index figure essentially because of a high species richness indicator (second highest of all countries listed), and a relatively low cost figure (ie low level of investment). This result is interesting because it occurs despite severe deforestation. Bangladesh is almost completely deforested with less than 5% of the original forest cover remaining (Beazley 1990). Several questions may be posed: How accurate is the species richness indicator? Has it occurred because of sampling bias (ie collection in favoured areas), or is there genuinely a rich biodiversity resource ? Are most of these species occurring in mangrove swamps or forests? As there is very little forest area left to work with in Bangladesh - it may be ideal for investment if high biodiversity is concentrated in forests. Such questions demonstrate the difficulties involved in making totally informed decisions at the global level.

The index is consistent in selecting 2 countries in the top five in all three versions while both threat measures recommend the same top five countries in terms of biodiversity cost-effectiveness, albeit in a different order. Only the use of the incomplete WCMC cost data causes the index selection to include three other countries (Korea, Malaysia and Bhutan). This would seem to indicate that a decrease in forests is linked to an increase in population, as the rates are trending together within countries. Care should be taken in making such generalisations, but it is probably fair to say that both deforestation and population growth rate are acceptable surrogates for the threat of land conversion.

The rankings based on different costs cannot, strictly speaking, be compared. The WCMC data are not complete for all countries and the index is therefore not calculated for these. The WCMC cost

[17] Sharma, (1992).

[18] Prof. Hanif Quazi, Pakistan Agricultural Research Council (personal communication 1995)

data are also for a longer period - the costs have been annuitised and discounted for the period 1991 to 1993. Nevertheless, the analysis does show that rankings will differ when global cost data are used for the index instead of just US data.

6.13 Sensitivity analysis and cost revisited

To ascertain whether the index is robust in assigning ranks, consider an example of the index calculation for Bangladesh, before undertaking the sensitivity analysis.

From Table 1, for Bangladesh:

$$Ps = \underset{\text{(Success Ratio)}}{0.08} \times \underset{\text{(Threat Ratio)}}{0.67} = 0.0536$$

$$\text{CEPII} = \frac{0.0536 \times 0.73}{5.49} \times 100 = 0.710$$

The threat ratio signifies the percentage of unprotected forest remaining in Bangladesh in ten years time. The higher this ratio, therefore, the lower the threat. The opposite is true for the success ratio - the higher the percentage of forest currently protected, the higher the probability of success.

other scenarios might be envisaged:

1. An increase in threat

As threat increases, the index gets lower. Suppose the threat ratio for Bangladesh decreases from 0.67 to 0.50. *Ceteris paribus*, this results in an index of 0.53, which is lower than 0.710. This result would seem to bear out the proposition presented earlier in this paper and by others (see Faith and Walker 1995), that it should not necessarily be a priority to invest where threat is highest. Threat is attenuated by probability of success and biodiversity, however. If both are very high for a particular country - it may still prove cost effective to invest.

2. An increase in the probability of success

This leads to a higher index, *ceteris paribus*. As we would like the index to show that an investment should be more cost effective if the probability of success is high, this is the desired result. The same is also true of biodiversity - the higher the species richness, the higher the index, *ceteris paribus*.

3. A lower level of investment

Lower cost (investment) will lead to an increase in the index, *ceteris paribus*. This will therefore favour those countries where conserving biodiversity is relatively "cheap". The biodiversity in place is there despite the threat and the low funding. Assuming that investment does reflect cost, we can choose however to interpret low investment (apparently cheap) countries differently. Low funding may for example signify that such countries are too expensive. Similarly the index result for highly funded countries could just as well be interpreted to signal cheap investments as showing that a country is too "expensive", or alternatively that current high investment levels already occurring put the country beyond a critical saturation level after which costs begin to rise.

Care is needed in interpreting our cost data, but the countries which have not received much funding - however the denominator is interpreted - to date will also do well in the index (relative to the other variables). Furthermore, whichever interpretation we choose; the data provide only a static picture and it should be noted that continued funding commitment is deemed to be extremely important in biodiversity conservation (Wilson 1992). Once a country has received funding and built up infrastructure, it must continue to receive assistance - projects cannot simply be abandoned. These considerations should be kept in mind in any interpretation of the index rankings over time.

The foregoing discussion shows how cost data introduces potential ambiguity as well as being the differentiating factor in the priority rankings of table A2 and 3. Reliable cost data are likely to be difficult to come by on a consistent manner and even where investment cost information is available, a comprehensive treatment of costs should ideally also consider the role of *opportunity costs* in the threat element of the index.

Opportunity costs represent the returns from the next best alternative land use. These returns will vary over countries and regionally within countries, but are powerful indicators of imminent economic threat. The main competition to biodiversity conservation arises from the returns to agricultural production and rents from timber and mineral extraction. In most countries good geographically-referenced data are generally available for all of these sectors. Inclusion of appropriate opportunity cost figures also allows the index to account for the distortionary effects of government intervention in sectors impacting most heavily on biodiversity conservation. That is, the addition of opportunity costs to existing investment costs will lower the ranking of countries where for example, existing government subsidies artificially increase timber extraction revenues. Hence in economic terms, in needing to cover normal set-up costs and compensate for revenues foregone, any investment will become relatively less cost-effective.

Note that the use of opportunity costs introduces the complex task of achieving some correspondence between threatened forest areas and reliable data on competing returns.

To demonstrate the role of opportunity costs, appendix 2 replaces investment cost figures with those of annual actual timber rent per km2 for Indonesia, Malaysia and Philippines (from Repetto 1988). The recalculated index in fact confirms the initial ranking of table 1, but the example is sufficient to demonstrate the flexibility of the index approach.

6.14 Limitations of the index

While there is no claim for the adequacy of a single index, the need to assess rapidly a range of criteria gives the approach some appeal. However, it is important to stress that the CEPII is merely a suggestion of how a cost-effectiveness indicator may be approached and that the example outlined is not intended for providing concrete real world examples.

With that in mind numerous caveats should be stressed. First, it is important to recall the dichotomy between investments in states and individual projects. In other words, the crude approach used here uses no project-specific criteria which are arguably of greatest importance for case specific interventions. As an example, the consideration of two competing elephant conservation projects in 2 separate countries might raise the issue of which project most effectively conserves at least the minimum viable land area for that species. But since such information is not included in the CEPII, the index is immediately limited. Similarly an issue such as the extent of park fragmentation is an important variable affecting successful biodiversity conservation. It has been shown (Wilson 1992) that the larger the protected area (park) and the more intact (no fragmentation), the higher the biodiversity (number of species). These factors are difficult to integrate into the index - ideally bigger parks should receive a positive weighting. We have not attempted this here due to data limitations.

The index provides a snap-shot, the only intertemporal assumption being that related to constant rates of deforestation. Funding availability, costs and domestic and international policy environments are all subject to unforeseen change which will not be reflected on the data for any particular ranking. Again, there is a premium on simplicity, but it is as well to consider some of these dynamics.

Further idiosyncrasies emerge from the data tables. Consider the construction of the index again. In the numerator two elements should be apparent. First, as demonstrated by the case of Benin in table A8, the absence of any protected forest area is sufficient to invalidate the index by returning an apparent zero ranking. If a country is certifiably denuded, then the interpretation put on the resulting ranking is that it is a hopeless case in terms of attracting further investments. Caution is required if in actual fact such an outcome is simply the result of data deficiency.

Likewise, as the value of k (deforestation or population growth rate) increases, so will any country's ranking fall. In a sense, both examples demonstrate potential ambiguity in the interpretation of a priority country. In other words, is it the highest or the lowest ranking countries that are the priorities? High equates to good bet through low threat. As already mentioned, low indicates a possible desperate case. Of course, the latter is precisely how many conservationists opposed to the *triage* strategy may choose to define a priority.

6.15 Conclusions

The aim of this chapter has been to demonstrate some of the complexities involved in deriving the optimum level of biodiversity investment at the global scale. The optimum in question should be qualified by saying that in practice it is not a static concept in the sense that the like evolutionary turnover, the costs and benefits are in a constant flux. This issue raises serious intergenerational issues related to equity and tastes, which are not pursued any further here. A related point about this flux, is that we have such a limited view of the indirect effects of diversity loss that the existence of benefit curves as drawn here is at best speculative. A notional alternative, considering issues of resilience and potential discontinuities in the damage function is suggested by Perrings and Pearce (1994).

Relative to another global environmental problem, that of climate change, there is at least one further complicating aspect related to the sovereign nature of biological resources and the requirement for economic bargains to be struck across boarders. The feasibility of such solutions has hardly been explored in theory or practice although there is a significant literature on both property rights and the use of covenants and easements within countries for much the same purpose. But as should be clear in referring to the valuation of biodiversity, there are a number of additional reasons why the efficient minimum cost solutions cannot come about in practice. The difficulties associated with valuing biodiversity offer a role for alternative policy approaches. Space constrains an exhaustive survey of existing instruments and regulatory approaches which in specific cases are viable alternatives alternatives to the issue of valuation (and much of the focus of this thesis). In the global context, in the mere existence of a body like the GEF there is some mechanism for the type of transfers at issue in this chapter. However the existence of funds is perhaps a necessary rather than a sufficient condtion for an efficient solution. The next question is then how to plan the efficient use of GEF resources.

There is no single correct method for setting biodiversity conservation priorities at any level of organization and the political acceptability of defining global priorities is an additional complication for the Convention on Biodiversity and its financial mechanism the GEF.

Biological information indicates the certain parts of the planet have much higher concentrations of low cost biodiversity than others, thereby providing a rationale for a strategy which jointly implements the global diversity objective through cost-effective targeted investments. If they are to be set, global priorities cannot be fully addressed by descriptive biological analysis alone. Social and economic factors - as the driving forces behind habitat loss, must be taken into account and the development of a unifying paradigm is clearly an area for cross-disciplinary research. The CEPII presents one heuristic economic perspective which explicitly addresses the issue of relative investment costs. Ideally, accurate cost information should be incorporated into complementarity-based selection methods. In the absence of a suitable global diversity surrogate or sufficient cost data, the method qualifies available species richness data with information on cost/investment levels. In addition to cost and biodiversity [benefit], the novel additions are the combined consideration of threat and the probability of success in index form.

The methodology is no more than a suggestion of how the scope of priority assessment may be broadened to provide a more effective filter for national investments. While there are alterative methods for representing similar information, the index approach offers distinct advantages of flexibility and convenience.

We caution against the overemphasis of the results presented here. Index rankings are relative and we suggest no absolute values to distinguish the performance of any one country relative to another.

Priorities can be set at global, national and local levels but data availability only allows us to assess at the level of analysis which happens to be furthest from where biodiversity actually resides. The national scale may well be appropriate for decision making but analysis at this level provides an incomplete picture of the dynamics of biodiversity conservation or loss at local and regional levels, and as such a the index does not obviate the need for a project by project approach irrespective of national boundaries. The issue of whose priorities we are actually setting matters and global priorities can only be accurately interpreted accompanied by some understanding of local priorities, demands and un-met needs. Again, data requirements are an invariably limiting resource.

Appendix 1 data tables

TABLE A1: Index Calculation Using a) Abramovitz Cost Data and b) Deforestation Rate as the Threat

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	
	Total area km2	Forested km2	Protected Forest km2	Unprot'd Forest km2	Success Ratio	Annual Deforest'n	Unprt'd 10 yrs time	Threat Ratio	Total SR	SR per km2	Cost	Index** (x 100)	
	A	Af	Ap	Au t	Ps	Rate	Au t+n	Au t+n/Au t	B	B/Af km2	C/Af km2		
	(Braatz 92)*	(FAO 93)*	(Braatz 92)*	(Af - Ap)	Ap/Af	(FAO 93)*			(WCMC 94)		(Abram 93)*		
Bangladesh	144,000	7,690	613	7,077	0.08	0.039	4,754	0.67	5,601	0.73	5.49	0.710	Bangladesh
Bhutan	47,000	28,090	9,056	19,034	0.32	0.006	17,922	0.94	6,068	0.22	45.29	0.145	Bhutan
Cambodia	181,000	121,630	0	121,630	0.00	0.010	110,000	0.90	532	0.00	n/a	n/a	Cambodia
China	9,561,000	1,150,600	263,578	887,022	0.23	0.039	595,893	0.67	34,166	0.03	0.71	0.644	China
Fiji	18,000	8,100	53	8,047	0.01	0.001	7,967	0.99	1,746	0.22	0.00	n/a	Fiji
India	3,288,000	517,290	122,401	394,889	0.24	0.006	371,825	0.94	17,881	0.03	2.10	0.367	India
Indonesia	1,905,000	1,095,490	163,309	932,181	0.15	0.010	843,048	0.90	25,315	0.02	2.98	0.105	Indonesia
Korea, Rep.	99,000	48,000	7,510	40,490		0.000	40,490	1.00	2,978	0.06	n/a	n/a	Korea, Rep.
Lao P.D.R	237,000	131,730	0	131,730	0.00	0.009	120,343	0.91	9,043	0.07	0.03	0.000	Lao P.D.R
Malaysia	330,000	175,830	14,816	161,014	0.08	0.020	131,560	0.82	13,691	0.08	5.26	0.102	Malaysia
Myanmar	677,000	288,560	1,693	286,867	0.01	0.013	251,682	0.88	8,445	0.03	n/a	n/a	Myanmar
Nepal	141,000	50,230	10,999	39,231	0.22	0.010	35,480	0.90	7,885	0.16	13.84	0.225	Nepal
Pakistan	796,000	18,550	35,170	(16,620)	1.90	0.034	(11,760)	0.71	5,725	0.31	3.45	12.001	Pakistan
Papua NG	463,000	360,000	290	359,710	0.00	0.003	349,063	0.97	12,796	0.04	1.04	0.003	Papua NG
Philippines	300,000	78,310	4,968	73,342	0.06	0.033	52,435	0.71	9,748	0.12	81.80	0.007	Philippines
Solomon	27,000	24,400	0	24,400	0.00	0.000	24,400	1.00	3,454	0.14	6.15	0.000	Solomon
Sri Lanka	66,000	17,460	7,071	10,389	0.40	0.014	9,023	0.87	3,704	0.21	14.79	0.505	Sri Lanka
Thailand	513,000	127,350	54,730	72,620	0.43	0.033	51,918	0.71	13,897	0.11	8.57	0.391	Thailand
Tonga	700	500	0	500	0.00	n/a	500	1.00	509	1.02	232.00	0.000	Tonga
Vanuatu	12,000	2,400	0	2,400	0.00	0.017	2,022	0.84	1,118	0.47	26.51	0.000	Vanuatu
Viet Nam	330,000	83,120	8,480	74,640	0.10	0.015	64,170	0.86	9,171	0.11	1.91	0.507	Viet Nam
W.Samoa	3,000	1,400	0	1,400	0.00	n/a	1,400	1.00	748	0.53	0.00	n/a	W.Samoa

Key:
SR: species richness
B: biodiversity (benefit)
C: investment (cost)

Notes:
n/a Not applicable - see Table 2 and Appendix 4 for details
** The index is calculated by multiplying columns [(5) x (8) x (10)] and dividing the result by column (11)
* For detailed source data see tables in Appendix 4
(..) Denotes negative number

TABLE A2: Countries Ranked According to Table 1 Index

Ranking	Index	Country	Reason for Index n/a
1	12.001	Pakistan	
2	0.710	Bangladesh	
3	0.644	China	
4	0.507	Viet Nam	
5	0.505	Sri Lanka	
6	0.391	Thailand	
7	0.367	India	
8	0.225	Nepal	
9	0.145	Bhutan	
10	0.105	Indonesia	
11	0.102	Malaysia	
12	0.007	Philippines	
13	0.003	Papua NG	
14	0.000	Lao P.D.R	
14	0.000	Vanuatu	
14	0.000	Solomon	
14	0.000	Tonga	
	n/a	Fiji	Zero cost.
	n/a	W.Samoa	Data unavailable - deforestation rate. Zero cost.
	n/a	Cambodia	Data unavailable - cost.
	n/a	Myanmar	Data unavailable - cost.
	n/a	Korea, Rep.	Data unavailable - cost.

TABLE A3: Index Calculation Using WCMC Cost Data

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	
	Total area km2	Forested km2	Protected Forest km2	Unprot'd Forest km2	Success Ratio	Annual Deforest'n	Unprt'd 10 yrs time	Threat Ratio	Total SR	SR per km2	Cost	Index** (x 100)	
	A	Af	Ap	Au t	Ps	Rate	Au t+n	Au t+n/Au t	B	B/Af km2	C/Af km2		
	(Braatz 92)*	(FAO 93)*	(Braatz 92)*	(Af - Ap)	Ap/Af	(FAO 93)*			(WCMC 92)		(WCMC 94)		
Bangladesh	144,000	7,690	613	7,077	0.08	0.039	4,754	0.67	5,601	0.728	270.87	0.01	Bangladesh
Bhutan	47,000	28,090	9,056	19,034	0.32	0.006	17,922	0.94	6,068	0.216	216.87	0.03	Bhutan
Korea, Rep.	99,000	48,000	7,510	40,490	0.16	0.000	40,490	1.00	2,978	0.062	1.15	0.84	Korea, Rep.
Lao P.D.R	237,000	131,730	0	131,730	0.00	0.009	120,343	0.91	9,043	0.069	3.23	0.00	Lao P.D.R
Malaysia	330,000	175,830	14,816	161,014	0.08	0.020	131,560	0.82	13,691	0.078	2.82	0.19	Malaysia
Pakistan	796,000	18,550	35,170	(16,620)	1.90	0.034	(11,760)	0.71	5,725	0.309	6.58	6.29	Pakistan
Sri Lanka	66,000	17,460	7,071	10,389	0.40	0.014	8,023	0.87	3,704	0.212	156.53	0.05	Sri Lanka
Viet Nam	330,000	83,120	8,480	74,640	0.10	0.015	64,170	0.86	9,171	0.110	36.93	0.03	Viet Nam

Key:
SR: species richness
B: biodiversity (benefit)
C: investment (cost)

Notes:
n/a Not applicable - see Table 2 and Appendix 4 for details
** The index is calculated by multiplying columns [(5) x (8) x (10)] and dividing the result by *column* (11)
* For detailed source data see tables in Appendix 4
(..) Denotes negative number

TABLE A4: Countries Ranked According to Table 3 Index

Ranking	Index	Country
1	6.292	Pakistan
2	0.844	Korea, Rep.
3	0.190	Malaysia
4	0.047	Sri Lanka
5	0.030	Bhutan
6	0.026	Viet Nam
7	0.014	Bangladesh
8	0.000	Lao P.D.R

TABLE A5: Index Calculation Using Population Growth Rate as the Threat

	(1) Total area km2	(2) Forested km2	(3) Protected Forest km2	(4) Unprot'd Forest km2	(5) Success Ratio	(6) Population Growth Rate	(7) Unprt'd 10 yrs time	(8) Threat Ratio	(9) Total SR	(10) SR per km2	(11) Cost	(12) Index** (x 100)	
	A	Af	Ap	Au t	Ps		Au t+n	Au t+n/Au t	B	B/Af km2	C/Af km2		
	(Braatz 92)*	(FAO 93)*	(Braatz 92)*	(Af - Ap)	Ap/Af	(WRI 92)*			(WCMC 92)		(Abram 93)*		
Bangladesh	144,000	7,690	613	7,077	0.08	0.026	5,438	0.77	5,601	0.73	5.49	0.81	Bangladesh
Bhutan	47,000	28,090	9,056	19,034	0.32	0.023	15,083	0.79	6,068	0.22	45.29	0.12	Bhutan
Cambodia	181,000	121,630	0	121,630	0.00	0.018	101,427	0.83	532	0.00	n/a	n/a	Cambodia
China	9,561,000	1,150,600	263,578	887,022	0.23	0.012	786,147	0.89	34,166	0.03	0.71	0.85	China
Fiji	18,000	8,100	53	8,047	0.01	0.014	6,989	0.87	1,746	0.22	0.00	n/a	Fiji
India	3,288,000	517,290	122,401	394,889	0.24	0.019	325,961	0.83	17,881	0.03	2.10	0.32	India
Indonesia	1,905,000	1,095,490	163,309	932,181	0.15	0.016	793,325	0.85	25,315	0.02	2.98	0.10	Indonesia
Korea, Rep.	99,000	48,000	7,510	40,490	0.16	0.008	37,365	0.92	2,978	0.06	n/a	n/a	Korea, Rep.
Lao P.D.R	237,000	131,730	0	131,730	0.00	0.026	101,222	0.77	9,043	0.07	0.03	0.00	Lao P.D.R
Malaysia	330,000	175,830	14,816	161,014	0.08	0.019	132,909	0.83	13,691	0.08	5.26	0.10	Malaysia
Myanmar	677,000	288,560	1,693	286,867	0.01	0.020	234,391	0.82	8,445	0.03	n/a	n/a	Myanmar
Nepal	141,000	50,230	10,999	39,231	0.22	0.023	31,087	0.79	7,885	0.16	13.84	0.20	Nepal
Pakistan	796,000	18,550	35,170	(16,620)	1.90	0.028	(12,511)	0.75	5,725	0.31	3.45	12.77	Pakistan
Papua NG	463,000	360,000	290	359,710	0.00	0.022	287,966	0.80	12,796	0.04	1.04	0.00	Papua NG
Philippines	300,000	78,310	4,968	73,342	0.06	0.021	59,317	0.81	9,748	0.12	81.80	0.01	Philippines
Solomon	27,000	24,400	0	24,400	0.00	0.028	18,368	0.75	3,454	0.14	6.15	0.00	Solomon
Sri Lanka	66,000	17,460	7,071	10,389	0.40	0.011	9,301	0.90	3,704	0.21	14.79	0.52	Sri Lanka
Thailand	513,000	127,350	54,730	72,620	0.43	0.013	63,713	0.88	13,897	0.11	8.57	0.48	Thailand
Tonga	700	500	0	500	0.00	0.014	434	0.87	509	1.02	232.00	0.00	Tonga
Vanuatu	12,000	2,400	0	2,400	0.00	0.032	1,734	0.72	1,118	0.47	26.51	0.00	Vanuatu
Viet Nam	330,000	83,120	8,480	74,640	0.10	0.020	60,986	0.82	9,171	0.11	1.91	0.48	Viet Nam
W.Samoa	3,000	1,400	0	1,400	0.00	0.010	1,266	0.90	748	0.53	0.00	n/a	W.Samoa

Key:
SR: species richness
B: biodiversity (benefit)
C: investment (cost)

Notes:
n/a Not applicable - see Table 2 and Appendix 4 for details
** The index is calculated by multiplying columns [(5) x (8) x (10)] and dividing the result by column (11)
* For detailed source data see tables in Appendix 4
(..) Denotes negative number

TABLE A6: Countries Ranked According to Table 5 Index

Ranking	Index	Country	Reason for Index n/a
1	12.7674	Pakistan	
2	0.8491	China	
3	0.8126	Bangladesh	
4	0.5201	Sri Lanka	
5	0.4815	Viet Nam	
6	0.4801	Thailand	
7	0.3215	India	
8	0.1968	Nepal	
9	0.1218	Bhutan	
10	0.1030	Malaysia	
11	0.0984	Indonesia	
12	0.0078	Philippines	
13	0.0022	Papua NG	
14	0.0000	Lao P.D.R	
14	0.0000	Vanuatu	
14	0.0000	Solomon	
14	0.0000	Tonga	
	n/a	Fiji	Zero cost
	n/a	W.Samoa	Zero cost.
	n/a	Cambodia	Data unavailable - cost.
	n/a	Myanmar	Data unavailable - cost.
	n/a	Korea, Rep.	Data unavailable - cost.

TABLE A7: CEPII Index Central /South America and the Caribbean. Calculation Using a) WCMC Cost Data and b) Deforestation Rate as the Threat

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	
	Total area km2	Forested km2	Protected Forest km2	Unprot'd Forest km2	Success Ratio	Annual Deforest'n	Unprt'd 10 yrs time	Threat Ratio	Total SR	SR per km2	Cost	Index** (x 100)	
	A	Af	Ap	Au t	Ps	Rate	Au t+n	Au t+n/Au t	B	B/Af km2	C/Af km2		
	(WRI 94)*	(FAO 93)*	(WCMC 94)	(Af - Ap)	Ap/Af	(FAO 93)*			(WCMC 94)		(see notes)*		B/C
Bolivia	1084380	493170	62380	430790	0.12648782	0.012	381,799	0.88627693	19277	0.03908794	15.1469067	0.02892924	0.00258059
Brazil	8456510	5611070	67180	5543890	0.01197276	0.006	5,220,096	0.94159435	59214	0.01055307	3.28849934	0.00361775	0.00320908
Colombia	1038700	540640	42720	497920	0.07901746	0.007	464,143	0.93216434	54443	0.10070102	24.2675348	0.03056496	0.00414962
Ecuador	276840	119620	14110	105510	0.11795686	0.018	87,985	0.83390174	21999	0.18390737	78.147467	0.02314847	0.00235334
Guyana	196850	184160	580	183580	0.00314944	0.001	181,752	0.99004488	7558	0.0410404	9.74695917	0.0013129	0.00421058
Paraguay	397300	128590	1090	127500	0.00847655	0.027	96,970	0.76055111	8961	0.0696866	4.56489618	0.00984161	0.01526576
Peru	1280000	679060	20310	658750	0.02990899	0.004	632,869	0.96071237	20880	0.03074839	2.83480105	0.03116699	0.01084675
Suriname	156000	147680	4560	143120	0.03087757	0.001	141,695	0.99004488	6117	0.04142064	0.40628386	0.31166302	0.10195
Venezuela	882050	456900	86450	370450	0.18920989	0.012	328,321	0.88627693	23132	0.05062815	1.18844386	0.71437564	0.04260037
Belize	22800	19960	2200	17760	0.11022044	0.002	17,408	0.98017904	3691	0.18491984	40.2805611	0.04959702	0.0045908
Costa Rica	51060	14280	4090	10190	0.28641457	0.029	7,592	0.7450618	13550	0.94887955	595.518207	0.03400192	0.00159337
Cuba	109820	17150	1390	15760	0.08104956	0.01	14,253	0.90438208	7038	0.41037901	29.4460641	0.10215521	0.01393663
Dom. Rep.	48380	10770	276	10494	0.02562674	0.028	7,900	0.7527706	6071	0.56369545	217.084494	0.00500924	0.00259666
El Salvador	20720	1410	50	1360	0.03546099	0.022	1,089	0.80055016	3562	2.52624113	3.54609929	2.02238274	0.7124
Guatemala	108430	42250	220	42030	0.0052071	0.017	35,407	0.84243263	9930	0.23502959	155.526627	0.0006629	0.00151119
Haiti	27560	230	40	190	0.17391304	0.048	116	0.61146202	5606	24.373913	2986.95652	0.08677567	0.00816012
Honduras	111890	46050	1100	44950	0.02388708	0.021	36,354	0.80877351	6745	0.14647123	35.4397394	0.00798457	0.00413297
Mexico	1908690	485860	10370	475490	0.0213436	0.013	417,170	0.87734727	28519	0.05869798	22.9613469	0.00478702	0.00255638
Nicaragua	118750	60130	21500	38630	0.35755862	0.019	31,887	0.82544867	8760	0.14568435	102.527856	0.04193806	0.00142092
Panama	75990	31170	8850	22320	0.28392685	0.019	18,424	0.82544867	11452	0.36740456	30.542188	0.28192977	0.01202941
Trin. &Tob.	5130	1550	270	1280	0.17419355	0.021	1,035	0.80877351	2888	1.86322581	2836.77419	0.00925337	0.00065681
Jamaica	10830	239	20	219	0.08368201	0.072	104	0.47367423	3651	15.2761506	11552.3013	0.00524152	0.00132235

Notes to table A7

1. Total (closed) forested area for El Salvador and Jamaican protected forest area both from sources cited in Sharma, N. (1992) , *Managing the World's Forests: Looking for Balance Between Conservation and Development*, Kendall-Hunt, Iowa

2. Cost data for Surinam, El Salvador, Haiti and Panama from Abramovitz, J. (1991), *Investing in Biological Diversity: U.S. Research and Conservation Efforts in Developing Countries*, World Resources Institute

3. Protected forest areas for the Dominican Rep., Haiti and Nicaragua are arbitrarily assumed to be approximately half total protected area, *World Resources 1994-95*, World Resources Institute

TABLE A8: CEPII Index African Region. Calculation Using a) WCMC Cost Data and b) Deforestation Rate as the Threat

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
	Total area km2	Forested km2	Protected Forest km2	Unprot'd Forest km2	Success Ratio	Annual Deforest'n	Unprt'd 10 yrs time	Threat Ratio	Total SR	SR per km2	Cost	Index** (x 100)
	A	Af	Ap	Au t	Ps	Rate	Au t+n	Au t+n/Au t	B	B/Af km2	C/Af km2	
	(WRI 94)*	(FAO 93)*	(WCMC 94)*	(Af - Ap)	Ap/Af	(FAO 93)*			(WCMC 94)*		(see notes)*	
Angola	1246700	230740	13000	217740	0.05634047	0.007	202,969	0.93216434	6411	0.02778452	1.49518939	0.09759322
Benin	11062	49470	1	49469	2.0214E-05	0.013	43,401	0.87734727	2813	0.05686275	0.00220336	0.04576915
Botswana	566730	142610	103000	39610	0.72224949	0.005	37,673	0.95111013	924	0.00647921	103.688381	0.0042925
B. Faso	27380	44160	7860	36300	0.17798913	0.007	33,838	0.93216434	1703	0.03856431	108.627717	0.00589021
Burundi	25650	2330	230	2100	0.09871245	0.006	1,977	0.94159435	3205	1.37553648	2433.90558	0.00525296
Cameroon	465400	203500	11060	192440	0.05434889	0.006	181,200	0.94159435	9515	0.04675676	57.4791155	0.00416283
Chad	1259200	114340	33000	81340	0.28861291	0.007	75,822	0.93216434	2267	0.01982683	7.70509008	0.06922833
Congo	341500	198650	6600	192050	0.03322426	0.002	188,243	0.98017904	5128	0.02581425	223.483514	0.00037616
C. d'Ivoire	318000	109040	5520	103520	0.05062362	0.01	93,622	0.90438208	4588	0.0420763	39.0040352	0.00493893
E. Guinea	28050	18260	335	17925	0.01834611	0.004	17,221	0.96071237	3761	0.20596933	14.1840088	0.02559417
Ethiopia	1101000	141650	6223	135427	0.04393223	0.003	131,419	0.97040178	7707	0.05440875	22.4708789	0.01032247
Gabon	257670	182350	8910	173440	0.04886208	0.006	163,310	0.94159435	7477	0.04100356	4.41458733	0.04273338
Gambia	10000	970	20	950	0.02061856	0.008	877	0.92281941	1587	1.63608247	256.701031	1.3457E-09
Ghana	227540	95550	190	95360	0.00198849	0.013	83,664	0.87734727	4677	0.04894819	327.263213	2.6094E-05
Guinea	245860	66920	97	66823	0.00144949	0.012	59,224	0.88627693	3748	0.05600717	113.508667	6.3387E-05
G. Bissau	28120	20210	0.01	20209.99	4.948E-07	0.008	18,650	0.92281941	1429	0.07070757	76.7936665	4.2043E-08
Kenya	569690	11870	1190	10680	0.10025274	0.006	10,056	0.94159435	8208	0.69149115	3778.18029	0.00172768
Liberia	96750	46330	930	45400	0.02007339	0.005	43,180	0.95111013	3074	0.0663501	0.50628103	0.2502081
Madagascar	581540	157820	2310	155510	0.01463693	0.008	143,508	0.92281941	10259	0.06500444	398.434926	0.00022037
Malawi	94080	34860	2460	32400	0.07056799	0.014	28,139	0.86849865	4798	0.13763626	62.8227194	0.01342743
Mali	122019	121440	3500	117940	0.02882082	0.008	108,837	0.92281941	2517	0.02072628	0.03046772	1.80927465
Mozambique	784090	173290	3750	169540	0.02164003	0.007	158,039	0.93216434	6616	0.03817878	0.86560102	0.08897223
Namibia	823290	125690	10000	115690	0.07956082	0.003	112,266	0.97040178	3995	0.03178455	21.0040576	0.01168325
Niger	1266700	25500	2380	23120	0.09333333	0	23,120	1	1791	0.07023529	0.26666667	2.45823529
Nigeria	910770	156340	3100	153240	0.01982858	0.007	142,845	0.93216434	6095	0.03898554	0.42314827	0.17029234
Rwanda	24670	1640	250	1390	0.15243902	0.003	1,349	0.97040178	3108	1.89512195	287.195122	0.09761304
Senegal	192530	75440	52	75388	0.00068929	0.007	70,274	0.93216434	2853	0.03781813	0.02120891	0.11457133
Sudan	2376000	429760	16000	413760	0.03723008	0.011	370,434	0.89528831	3108	0.00723194	6.91316083	0.00348687
Tanzania	886040	335550	1930	333620	0.00575175	0.012	295,680	0.88627693	11701	0.03487111	5.28386232	0.00336422
Uganda	199550	63460	690	62770	0.01087299	0.01	56,768	0.90438208	6935	0.10928144	126.946108	0.0008465
Zaire	2267600	1132750	51510	1081240	0.04547341	0.006	1,018,089	0.94159435	12604	0.0111269	1.61112337	0.02957106
Zambia	743390	323010	71200	251810	0.22042661	0.011	225,443	0.89528831	5797	0.01794681	8.64679112	0.04095994
Zimbabwe	386670	88970	2831	86139	0.03181971	0.007	80,296	0.93216434	5631	0.063291	41.2386198	0.00455225

Notes to table A8

1. Protected forest areas for Angola, Botswana, Burkina Faso, Chad, Malawi, Mali, Namibia, Niger and Sudan from sources cited in Sharma *op.cit.*

2. Benin has certifiably zero protected forest area (and is covered 100% in the WCMC Biodiversity Map Library). Protected forest area is arbitrarily set to a small amount to avoid invalidating index calculation. Similarly, Guinea Bissau shows no recorded IUCN protected areas or forest reserves. Protected forest area is set to an arbitrarily small amount.

3. For some countries there is no evidence of the overlap between forest and existing protected areas (IUCN categories). Ethiopia for example has 62230km^2 area protected but none is recorded as protected forest. We arbitrarily assume 10% is protected forest.

4. Where species richness figures are not indicated for a taxonomic group but endemism figures are available, these have been included in total species richness instead. This should understate total species richness.

Appendix 2

Opportunity costs and index calculation

Opportunity costs (timber rents 1979-82) and CEPII recalculation

Country	Actual rent form log harvest (in million US dollars)	Equivalent rent/opportunity cost ($ /km^2/year/forested area)	Recalculated CEPII
Indonesia	4,409	1341	0.00023
Sabah, Malaysia	2,064	3912	0.00014
Philippines	1,001	6402	8.819E-05

Adapted from Repetto (1988)

$$Ps = \underset{\text{(Success Ratio)}}{0.15} \times \underset{\text{(Threat Ratio)}}{0.90} = 0.135$$

$$CEPII = \frac{0.135 \times 0.02}{1341} \times 100 = 0.00023$$

Chapter 7
Conclusion

7.1 Introduction

This thesis has examined the economic valuation of biodiversity in a cost-benefit framework. The broad aim of this approach is to level the playing field which currently pits an undervalued (possible zero or negative value) biodiversity stock, against a range of development pressures. Biodiversity has many obvious values. The non-market aspect of value is almost as problematic as the scientific concept itself. This concluding chapter articulates the lessons that seem to emerge from the issues discussed, and suggests appropriate avenues for further research and policy design. Discussion will cover the issues of economic valuation, its relation to biological value and links to the issue of investment priorities. Reference is made to alternative approaches to valuation and other unaddressed but related issues.

7.2 Lessons

However described, biodiversity has economic value and to an extent can be accommodated within existing social cost-benefit methods. This thesis has examined the implications of the many biological uncertainties which also emphasise alternative views to the concept of value. Many of the methodological uncertainties of the economic value paradigm (adopted here) were discussed. The approach distinguishes between the value of biodiversity - the emphasis on the diversity - and biological resources. The latter are the less precise focus of much of the existing valuation literature. There are several theoretical strands to a formalised theory of diversity contributing to an understanding of the investment priority problem. But there is currently no universally tractable method arising from these, and much biological work relies on surrogate methods. Assuming that a biodiversity measure or index can be formulated, uncertainty still remains about what it is about diversity that is valuable. Chapter two attempted to investigate a theoretical link between diversity (distance) concepts and direct and indirect uses, which are the most obvious sources of value. The issue was partly about an operational basis for option value, the most commonly evoked theoretical rationale for conservation. It is possible to speculate that the bounds of option value may be based on some unspecified link between distance and the probability that something is either directly or indirectly useful. This link gives substance to the basic arguments for caring about the earth's biological resources.

Avoiding such speculation, diversity itself is surely of value, but the choice of a biological numeraire is totally arbitrary. Moreover, the choice to maximise a particular character (eg genetic diversity), need not translate into the simultaneous maximisation of more anthropocentric or 'charisma' characters which are arguably closer to the popular perception of biodiversity. This issue of what we value and thus conserve lies at the heart of competing views held by the stakeholders in the conservation debate. One finding from the line of enquiry in chapter two is that the concept of option value represents a common line of enquiry from economic and biological perspectives.

Given the complexity of biodiversity and the pace of scientific endeavour, some short cuts are necessary and inevitable. It turns out that a default use of surrogates such as species richness may not be a problem in biological terms. As suggested in chapter two, limited research suggests that precise character representation objectives can be translated into workable conservation criteria. That is, the focus on higher taxonomic groups may be the best conservation strategy. Furthermore such a strategy may also tie in with the ecological view that habitats are the best units to target. If so, and given what is known about species-area requirements, then there is also a good basis for economic enquiry without the valuation methods which have a useful but limited role.

The problem of market failure can be addressed using valuation, and much of chapters three, four and five focused on the development of CVM for informing conservation decisions. CVM and other stated preference methods offer the flexibility of direct inquiry, but are prone to numerous technical problems. Strictly speaking, many of these are not economic shortcomings. Several dimensions to a well-executed CV survey require additional inputs from psychology and other disciplines although many practitioners are inclined to do without. The shortcomings that have been addressed are common to many recent applications. These include what respondents know, and what they can be told without undue influence on their own preferences. Biodiversity is an extreme case in point. Moreover, the focus on the dichotomous choice approach for eliciting WTP shows that many of the more fashionable aspects of CVM have large pitfalls for the unwary. In short, several lessons were suggested in the CV chapters, and I only reiterate those related to information provision and the slightly dubious preference for discrete choice over open-ended formats. Overall however, there are limits to using CV and it would be helpful for these to be spelt out so that CV and similar methods are correctly located in the conservation tool box.

7.3 Institutional and policy response

Valuation is not the over-riding policy objective to emerge from this thesis. In the first place it seems somewhat disingenuous to require biodiversity to compete in the market when it is known that the supply of many other environmental goods such as air and water is enacted partly (and without question) using regulations and instruments.

The focus on global policy issues showed that it may be possible to make rational (cost-effective) assessments without value information. Furthermore, the identification of institutional failures as proximate causes can be corrected without recourse to the valuation of externalities. Other regulatory methods, possibly based on safe minimum standards, may be enacted irrespective of cost-benefit criteria. While such decisions sanction obvious costs and related distributional issues, they are ultimately more transparent than the use of non market valuation. Furthermore the issue of market creation and property rights for biodiversity conservation is also circumscribed by the distributional caveat. In isolation valuation will frequently be of limited worth without the vital step of translating potential Pareto improvements into actual resource flows, i.e value capture (Pearce 1995).

The conclusion drawn from a review of applications of CV, is that there is a gulf between theory and practice. When it comes to valuation, many organisations that matter in regard to biodiversity are in a learning stage or are too sceptical. How this is overcome is an institutional issue, although again, a clear view of the limits to CV should be conveyed to avoid potential confusion. It is basically impossible to value biodiversity (in the strict sense employed in chapter two), only elements of biological resources. There are natural limits which follow from this, and it is in any case unreasonable to conduct valuation *ad infinitum*. One way around this is to establish values libraries or data bases for transferring benefits estimates. Ideally this approach would involve the establishment of a range of 'representative' unit values for species and ecosystem/landscape types. However, one problem is that this it does not avoid the need for a consensus view or value judgement about what is representative. It is also possible to look to other methods.

Chapter two noted the use of algorithm methods for priority area selection. These are a means to introduce efficiency criteria from a biological perspective. Development of these methods arises from a realisation that default and ad hoc conservation programmes in many countries are inefficient by any criteria. It would be preferable for conservation decisions to be guided by the criteria mentioned in relation to these methods, and there is no reason why the resulting biologically efficient choices cannot be subject to some method to value competing options. In the first instance cost considerations are a decisive factor for area selection methods. If non-market valuation is an issue then this may be considered on a case by case basis as appropriate. This does not immediately imply CV or any other stated reference method. At the end of chapter three, the idea of Consensus Conferences and or

Values Juries was mooted. These have been used to guide preferences on similar complex issues.

Other obvious policy approaches are available. The win-win potential in correcting a range of policy failures is suggested but not given too much coverage in this thesis. Numerous policy interventions and potential instruments for biodiversity conservation are extensively reviewed by the author in OECD (1996). Such options should be prior to embarking on any extensive programme of valuation.

7.4 Suggestions for further research

There are numerous suggestions for future research, many which are not strictly in the domain of economics. Biodiversity is and will remain a field requiring the reconciliation of a number of competing and complementary views.

This thesis required some involvement with various aspects of the science of biodiversity conservation and the author is aware of the limitations in that field. Clearly our understanding of the world's biological wealth is not as complete as we would like. This is a consideration in drawing up any wish list for improving economic methods, and in fact explains why economic progress is so slow relative to other global environmental problems. Accordingly the suggestions are the improvement of taxonomic information dealing with phylogenetics and the development of distance measures described in chapter two. The investigation of the role and adequacy of surrogate species or habitats is also a vital cost consideration in relation to the type of crude assessment in chapter six. Either of these aspects (distance or species data), might be the basis of a tractable indicator approach to monitoring biodiversity and potentially the basis of a wealth adjustment in resource accounting.

In economic terms the research agenda must be about valuation for priority setting. With or without perfect biological data numerous issues arise because *in situ* conservation requires so much land. In the developing world, basic information such as costs of reservation and management of existing areas cannot be taken for granted. However there is enough biological information to make a decent hand of priority setting using basic mathematical programming.

There are several requirements to address for dealing with the more specific issues raised by stated preference methods. The main challenge is the refinement of information provision possibly in an experimental setting. This involves checks on respondent understanding and cognition of concepts of biodiversity. This is not a task in which economists are expert. In terms of design and estimation, more robust (or at least some consensus on) statistical methods would seem to be necessary .

From the economic perspective the inclusion of non market benefits and costs is desirable and some investigation of the requirement for consistent use in policy appraisal is necessary. Outside environmental economics there is little consensus about the role of valuation. Other value approaches are frequently raised in the conservation debate, and even if these are less operationally useful, it is of some interest to examine how competing views may be reconciled. This thesis suggests that there is a potential shared objective in option value, which in theory reconciles biological and economic objectives and at once addresses the criticism of economic values as incomplete or exclusive[1]. Aside the location of option value this interface, along with the public preference route (figure 7 chapter 2), requires further investigation. In fact, given the complexity of the phylogenetic story and supposing that future choices could be presented in such terms (i.e. consensus conferences), such approaches may be the most appropriate consensus methods. In the field of biology such issues are now being confronted in complex debates ranging from genetically modified organisms to the human genome project. The general availability of data on these subjects also opens them up to economic enquiry.

Other value paradigms such as rights-based and contractarian approaches also suggest some interesting research related to the use of resource compensation thorough mitigation rather than in monetary terms. Mitigatory action is something that happens a lot at local level in some countries, but issues of economic and biological efficiency are rarely addressed in the literature. It is possible to dress mitigation up in the same utility theoretic framework of the Random Utility Model of discrete-choice analysis. Choices about how when and where to mitigate raise several further ecological and economic complications.

Other areas of research require attention. For example, issues such as the value of biodiversity in a system perspective, resilience and system integrity. By extension where are the sustainability constraints on biodiversity erosion and where should standards be set? Finally, although the developing world remains the repository of much of the worlds rarest diversity the development angle has received only cursory attention. Instruments to take advantage of gains from trade are lacking, particularly those that transfer the domestic experience with rights attenuation and covenants, to the international sphere. Furthermore, how to guide development decisions taking biodiversity into account? What do we know about the macroeconomic policies and diversity, and what if anything can be said about the income elasticity of demand for biodiversity?

[1]Such criticisms might legitimately be levelled at specific methods for eliciting some economic values.

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