Economic analyses to aid nature conservation decision making

Ross Cullen, Kenneth F.D. Hughey, Geoff Fairburn and Emma Moran

Abstract Nature conservation can often be costly and the resources available are often less than are required. Resource allocations and expenditures for nature conservation have an opportunity cost in terms of foregone projects, but are rarely subjected to as much scrutiny as other public activities. Failure to apply economic tools to nature conservation decision making can result in errors in project selection, wasted use of scarce resources, and lower levels of conservation than it is possible to achieve from the resources available. In some instances where economics has been proposed for use in nature conservation research, the methodologies applied provide information that is of limited usefulness to decision makers. Non-market valuation has limited potential to help in nature conservation decision making, is costly to complete and should be supplanted by more useful techniques that focus on the cost and the payoff from nature conservation actions. In contrast, Cost Effectiveness Analysis and Cost Utility Analysis are practical tools that can provide valuable information for conservation decision makers and improve conservation achievement.

Keywords Cost Effectiveness Analysis, Cost Utility Analysis, decision making, economics, nature conservation.

Introduction

Nature conservation is big business in many countries. Global expenditures on nature reserves are estimated to total US $6 billion annually (James *et al.*, 1999). Additional expenditure of $16.6 billion per annum is estimated, however, to be necessary to provide and maintain a broadly representative system of nature reserves (James *et al.*, 1999). In one Global Hotspot, New Zealand (Given & Mittermeier, 1999), approximately 800 indigenous species are listed as acutely and chronically threatened. Despite annual expenditures of NZ $106.5 million on management of natural heritage, 92% of these species do not receive enough help and 77% have no programme specifically targeting their recovery (Department of Conservation, 2004).

Arguments for nature conservation expenditure are rarely based upon economic analysis, and resource allocations and expenditures for nature conservation are infrequently subjected to more than token economic evaluation. The case for nature conservation expenditures is typically based upon such imperatives as the need to avoid loss of some habitat or species in the face of continuing development pressures, or the need to counter the threats posed by aggressive invasive species. Balmford *et al.* (2002) provide a contrasting approach in estimating the potential global benefits from nature conservation, comparing them to the estimated global costs of nature conservation, and concluding the likely benefit cost ratio is at least 100:1.

Nature conservation actions focused on inventory and rescue have been described as the fire-brigade period (Edwards & Abivardi, 1998), and can be contrasted with nature conservation decisions based upon formal economic analysis. Nature conservation projects have both financial costs and an opportunity cost (the resources used on a conservation project could have been used for an alternative conservation project, or for projects in other areas) and should be, but rarely are, subjected to as much scrutiny as other public activities. Horta *et al.* (2002) for example note the weak monitoring and evaluation within the Global Environment Facility despite the allocation of nearly US $1.4 billion for 470 biodiversity projects in 160 countries over 1991–2001. The failure to apply economic tools in nature conservation decision making can result in errors in project selection, wasted use of scarce resources, and lower levels of conservation than may be achievable from the limited resources available. One example of the potential payoff from inclusion of economic factors in decision making is provided by Ando *et al.* (1998) who demonstrate that when pursuing a target of providing habitat for 453 endangered species, consideration of both economic and ecological factors

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can achieve the target at one sixth of the cost compared to when only ecological factors are considered.

In some cases where economics has been proposed for use in nature conservation, inappropriate economic methodologies are advocated. In this paper, we illustrate why Benefit Cost Analysis and non-market valuation have limited potential to help in nature conservation, and we report on how Cost Effectiveness Analysis (CEA) and Cost Utility Analysis (CUA) can provide valuable information for decision makers.

Economics for decision making

Expenditures on nature conservation have an opportunity cost and it is important to ensure the resources are used to best effect. Statements from conservation organizations espouse wise use of resources and implicitly recognize expenditures have an opportunity cost. For instance, the New Zealand Department of Conservation Statement of Intent states the aim of achieving the maximum conservation with the resources available (DOC, 2003). Economic evaluation techniques can help assess whether a particular conservation project provides a better use of the resources than an alternative project would provide. The failure to complete economic evaluation of projects, however, means that conservation organizations and society as a whole must rely upon guesswork when judging the merit of conservation projects. This is likely to result in less conservation output being achieved from a given budget than could be achieved if decision makers have available to them appropriate evaluation information.

Despite the lack of economic evaluation, a wide range of methodologies are available to aid conservation decision making. The most widely used technique for evaluating projects and programmes is Benefit Cost Analysis, which has been used to evaluate some nature conservation issues (Engermann *et al.*, 2002, 2003). An obvious impediment to the use of Benefit Cost Analysis, however, is the absence of market generated prices for nature and nature's services. A typical nature conservation project can involve land purchase, annual expenditures on planning, habitat management, predator control, and monitoring (Cullen *et al.*, 2001). Such expenditures can usually be measured using market determined prices and total expenditures. In cases where a nature conservation project requires maintaining existing habitat such as forest, the value of any timber foregone can be calculated to determine the true economic cost of nature conservation (Montgomery *et al.*, 1994). In some instances avoided cost approaches can be used to provide a value of the output from nature conservation projects (Engermann *et al.*, 2002). However, avoided costs data are only occasionally available, the outputs from nature conservation projects are rarely sold, and innovative methods are required to quantify the outputs produced by these projects. Economists have been successful in developing and refining a number of non-market valuation techniques to tackle these situations. The Travel Cost Method, Contingent Valuation Method, Hedonic Price Method, and Choice Modelling have been developed and applied in a wide range of situations to generate market-like price information (Loomis & White, 1996; Loomis & González-Cabán, 1998; Freeman, 2003).

It has been argued that such non-market valuation studies can provide valuable information to aid nature conservation decision making (Loomis & White, 1996; Carson 1998; Edwards & Abivardi, 1998; Loomis & González-Cabán, 1998). We argue that non-market valuation studies have been of limited practical importance in most nature conservation decisions, there are significant obstacles to widespread use of non-market valuation studies, and alternative economic techniques have greater ability to provide useful information for decision makers.

Non-market valuation studies have in many cases been used to provide a measure of aggregate Willingness To Pay (WTP) for protection of a species. An example is provided by Bandara & Tisdell (2004) who use the Contingent Valuation Method (CVM) to determine WTP for protection of Asian elephant *Elephas maximus* in Sri Lanka. They estimated the '. . . annual return for the total extrapolated WTP of urban residents (Rs 2,012.43 million) for conserving the elephant in Sri Lanka is nearly twice the crop and property damage caused to farmers by elephants (Rs 1,121.42 million) per annum'. The results of this study indicate there is a solid economic basis for maintaining Asian elephants in Sri Lanka. This is useful information but it is unlikely to be particularly helpful to decision makers as it does not indicate the '. . . socially optimal level for the population of wild elephants' (Bandara & Tisdell, 2004). Conservation decision making focuses in many cases on the level of protection to provide a species, habitat or ecosystem, and it is rarely the case that conservation is a stark choice between conservation and abandonment of a species (Montgomery *et al.*, 1994).

Non-market valuation studies require skilled researchers to complete careful survey research to gauge individuals WTP for the item to be valued. The definition or description of the item to be valued is a crucial feature of non-market valuation studies. Respondents to contingent valuation studies or choice modelling studies must have a reasonable understanding of the item being valued. Description of the outputs and expected outcome from nature conservation projects is often difficult, and many respondents will struggle to understand what it is they are being asked to value. Turpie (2003) provides some illuminating comments on the level of knowledge among CVM survey respondents about ecosystems and species richness of biomes. She reports that almost one third of South African respondents to a CVM study had never heard of the internationally significant fynbos, and '. . . three quarters of respondents underestimated species richness by at least an order of magnitude' (Turpie, 2003). In such situations the results from non-market valuation studies are likely to be imprecise and of limited value to conservation decision makers.

Non-market valuation studies are relatively costly to conduct. Value Transfer approaches that carefully adapt existing WTP, rather than completing new non-market valuation studies, and habitat-based evaluations have been proposed as ways to avoid the cost of non-market valuation studies for each single species (Loomis & White, 1996). There are, however, alternative economic methodologies available to ensure that decision makers are provided with useful information to aid selection of projects. The alternatives are likely to be helpful in situations where decision makers wish to select least cost means to accomplish a nature conservation objective. Willingness To Pay studies do not focus on this question and hence alternative evaluation techniques are required to assist decision makers.

Biophysical units

Nature conservation projects aim to produce outputs such as an increase in a species' population, or an increase in the area of ecosystem managed. In almost all cases such projects involve financial expenditures, and in many cases they also incur opportunity costs. Measurement of costs is a first crucial step to evaluation of actual or potential projects, and the second is measurement of the outputs produced by projects. It is at this point where Cost Effectiveness Analysis and Cost Utility Analysis are most useful. CEA and CUA both require a unit of measurement for calculation of cost per unit of output ratios. The units of output in nature conservation projects can be measured in biological or physical terms, and do not require the difficult step of estimating individual's WTP for these items. Obvious examples of biophysical units include population size and area of habitat (Montgomery *et al*., 1994). Here we illustrate three situations where CEA and CUA techniques, using biophysical units, have provided information that can be used to improve decision making.

Choice of project to manage a species

In many instances conservation managers must choose which sites to manage to aid recovery of a single threatened species. If the quantity of outputs relative to the quantity of inputs used at each site (i.e. productivity) can be quantified, CEA can be employed to determine the mix of projects that are likely to maximize conservation output from a conservation budget (Fairburn *et al*., 2004). We demonstrate one example of the application of CEA. The North Island kokako *Callaeas cinerea wilsoni*, is managed at many sites by the New Zealand Department of Conservation. The North Island Kokako Recovery Group has set a goal in the recovery plan (Innes & Flux, 1999) to 'Improve the status of North Island kokako from endangered by restoring the total national population to *c*. 1,000 pairs by the year 2020, in sustainable communities throughout the North Island.' In addition the Recovery Plan states: 'In order to attain the stated goal of this plan, we state 23 key sites which represent the necessary minimum management sites required to improve the status of kokako by 2020'.

Achievement of the population goal can be tackled by seeking increase in population numbers at one or more sites. Choice of site or sites can be based upon a number of criteria, including the cost effectiveness of the projects at each of the sites. In this case, because the goal has been defined, a suitable unit of output to use is additional male/female pairs of North Island kokako, and the economic criteria to consider is cost per additional male/female pair of North Island kokako. A cost effectiveness formula for kokako can be expressed as follows:

$$PAYOFFk = \sum_{t=0}^{T}\left[\frac{K_n - K_i}{(1+d)^t}\right] \Big/ \sum_{t=0}^{T}\left[\frac{C_t}{(1+d)^t}\right]$$

where $PAYOFFk$ is the change in the number of discounted male/female kokako pairs per discounted conservation dollar spent on kokako protection at a site, K_n is the number of female/male pairs at the end of the planning period, K_i is the number of female/male pairs at the beginning of the operational period, C_t are the direct costs of protection at a site in year t, and d is the discount rate. Discounting applies a weight of magnitude $1/(1 + d)^t$ to costs or outputs associated with the project; the higher the discount rate and/or the greater magnitude is t, the smaller is the weight applied. Cost and outputs that have been discounted are referred to as 'present values'. Costs in this paper are measured in NZ$, and NZ $1.00 ≈ US $0.70 in April 2005.

Table 1 provides information on the cost per additional pair of North Island kokako at three sites as reported in Fairburn *et al*. (2004). An assumption for these calculations is that average costs provide a useful approximation of the additional costs per pair (marginal costs) for the last pair added to the population. The cost effectiveness ratio of Matarua can be compared to the estimated cost effectiveness of any proposed kokako management sites to help determine whether additional

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Table 1 Cost effectiveness of North Island kokako conservation at three sites (data from Fairburn *et al.*, 2004). Costs are in NZ $ and the discount rate, *d* (see text for details), is 6%.

Site	Hectares	No. of years	Present value of total cost	Annual amortized cost	Effectiveness ($K_n - K_i$) pairs	Annual amortized cost per hectare	Present value of cost per additional kokako pair
Matarua	440	5	287,078	57,416	9	155	31,898
Mapara	1,400	12	1,780,312	148,359	45	152	39,562
Otamatuna	1,300	3	401,280	166,962	7	115	57,326

sites are warranted, given a budget constraint. The greater cost effectiveness of Matarua compared to Otamatuna is valuable information for decision makers who can weigh up the likely cost savings through concentrating on the Matarua project, versus any perceived benefits from managing kokako at several sites. Without information of this type, decisions on choice of North Island Kokako projects can only be based upon non-economic criteria, and are certain to result in less conservation output being achieved than is possible with the limited conservation budget.

Comparisons across single species projects

New Zealand has classified 2,373 species and subspecies as being threatened or endangered (Hitchmough, 2002). The New Zealand Biodiversity Strategy provides a 20-year plan to halt the decline in New Zealand's native species (DOC & MFE, 2000a). The Department of Conservation has developed Recovery Plans for 46 of these species, and single species programmes are being carried out to recover many species. A single species approach is also employed in several other countries, including the USA. Single species programmes can be studied to determine which programmes are providing the best payoff from the resources invested in them. A new problem to be met is how to measure and compare the outputs produced by a range of single species programmes. These programmes are likely to include species with varying life spans and breeding rates, often facing different threats. In these circumstances CEA is of limited value as it uses a simple unvarying measure of output, and a more sophisticated unit of output is necessary to enable comparison to be made across single species programmes. Cost Utility Analysis provides a means to overcome this problem. CUA measures the success or output of projects with time varying units of output (Drummond *et al.*, 1997; Cullen *et al.*, 2001). We demonstrate here how we have applied CUA to compare the relative success and cost effectiveness of a range of single species programmes.

Measuring the success of threatened species recovery projects requires comparison of the conservation status of a species over time with the project to what its status would have been over time without the project. The contribution of a project to threatened species conservation is the sum of any differences between the conservation status of a species with the project and the species' status without the project. We asked New Zealand project and species managers to estimate the status of a species on a continuum from 0.00 (Extinct) to 1.00 (Not Threatened). The continuum is linked to the categories on the Department of Conservation Threat Classification System (Molloy *et al.*, 2002) and uses a quadratic scale that ensures conservation status scores increase at a diminishing rate as a species moves closer to 1.00. The quadratic scale places more value on improving the conservation status of endangered species than improving the status of less threatened species. Use of a continuum from 0.00 to 1.00 allows for accuracy in measuring changes in the conservation status of species. We sum the yearly (with the project minus without the project) scores for a threatened species, for the number of years selected, to calculate the output measured in units called Conservation Output Protection Years (COPY).

Before discounting is completed, contribution of a project to the conservation of a threatened species present at a site is measured using $\text{COPYi} = \sum_t (Sitw - Sitw/o)$, Where $Sitw$ is species i conservation status in year t with management W, and $Sitw/o$ is species i conservation status in year t without management.

A species management project, for example, that improves a species' conservation status from 0.30 (its without project status) to 0.50 (its with project status), and that maintains that status gap for 10 years, produces $(0.50 - 0.30) \times 10 = 2.00$ COPY. The annual with project minus without project scores can be discounted using a range of discount rates when calculating the present value of COPY. Discounting is widely used in appraisal of projects that span multiple years and typically involves weighting of measures of costs or output by a factor $1/(1 + d)^t$ where d is the chosen discount rate expressed as a decimal and t is the number of years after commencement of the project. This measure of output is for a selected time period, and a project may deliver some conservation benefits after the study period, even if there are no further project expenditures.

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Table 2 Cost effectiveness of single species recovery programmes (data from Tables 5 & 7, Cullen *et al.*, 2001). Costs are in NZ $ and the discount rate, *d* (see text for details), is 6%.

Species recovery programme	Present value of total cost	Present value of COPY* produced	Present value cost per present value of COPY*
Brothers Island tuatara *Sphenodon guntheri*	13,694	0.33	40,780
Cook Strait tuatara *Sphenodon punctatus*	13,694	0.18	76,457
Campbell Island teal *Anas anas nesietis*	39,940	0.39	103,178
Short tailed bat *Mystacina tuberculata*	318,938	1.73	184,570
Yellow-eyed penguin *Megadyptes anipodes*	603,013	1.97	305,344
Hector's dolphin *Cephalorynchus hectori*	773,844	0.74	1,048,245
Black stilt *Himantopus novaezelandiae*	2,441,822	2.26	1,077,724
Takahe *Porphyrio hochstetteri*	3,278,178	1.41	2,327,560
Mean	935,390	1.23	645,482

*Conservation Output Protection Years (see text for details)

Table 2 reports some results from use of the CUA approach for New Zealand single species programmes. The research is more fully reported in Cullen *et al.* (2001). It is clear that there are large differences in cost per COPY between these single species programmes. This information is invaluable to conservation managers who need to determine which single species programmes are providing the greatest payoff per dollar invested. CUA appears to provide the only practical means available at present to answer such questions.

With some requisite adaptations CUA could be used, together with other information such as uniqueness of each species, to select new projects based upon projected numbers of COPY, and projected costs of the projects. Conservation managers equipped with this information will be well placed to consider which new projects will best contribute to a goal of recovering as many threatened species as possible with a limited budget.

Comparison of multiple species projects

In New Zealand conservation of many species is based upon offshore islands that provide sanctuary from predators and competition (DOC & MFE, 2000b). There are a limited number of offshore islands available for these projects, and suitable habitats are not available for all species on these islands. Mainland habitat islands have recently been introduced to provide an alternative means to recover species. Many offshore island projects and all mainland habitat island projects manage multiple threatened species. Multiple species projects have been introduced in New Zealand with the expectation they will be high cost and high risk, but have high output (Saunders, 1999). They have also been introduced in other countries, including the USA, for cost efficiency reasons. Multiple-species projects can spread expenditures over several species and may provide a lower cost way to manage several threatened species than can be achieved with several single-species projects and programmes (Tear *et al.*, 1995). Projects that produce several products together at lower cost than could a group of single product projects are described as achieving economies of scope (Milgrom & Roberts, 1992). The success and cost effectiveness of multiple species projects can be measured, and compared to the success and cost effectiveness of single-species projects and programmes to determine if there is evidence of economies of scope occurring.

We studied six New Zealand multiple species projects to determine which are the most successful and cost effective from a species conservation perspective. Three of the projects are offshore islands and three are mainland habitat islands. Using CUA we have measured the success of these projects at conserving species, and calculated the total numbers of COPY produced at each site (Cullen *et al.*, 2005). Table 3 reports the total COPY and present value of expenditures for the six projects. We used that information to calculate the present value of cost per present value of COPY (final column of Table 3) for each of the six sites, and compare the performance of the projects.

The results must be treated with caution as we have data from only six projects and there is considerable variability in the productivity of both the offshore island and the mainland habitat island projects, but this analysis shows that the three mainland habitat islands are on average more costly and less productive than are the three offshore islands. The proportion of a species population managed at a site is a key factor influencing relative success of the projects. These results can also be compared to the information on single species programmes (Table 2). It appears that single species programmes are on average more productive and have greater cost effectiveness than do multiple species projects. We have found no evidence that there are economies of scope in the six multiple species projects.

Table 3 Comparison of six multiple species projects (data from Table 4, Cullen *et al.*, 2005). Costs are in NZ $ and the discount rate, *d* (see text for details), is 6%.

Project and location	Area (ha)	Present value of costs	Annualized cost per ha	Present value of COPY* produced	Present value cost per present value of COPY*
Offshore islands					
Little Barrier Island	2,817	780,345	28.52	1.83	427,385
Tiritiri Matangi	218	1,547,381	730.84	0.08	19,516,305
Maud Island	320	2,162,521	695.80	1.54	904,821
Mean offshore island	1,118.3	1,496,749	485.06	1.15	6,949,504
Mainland habitat islands					
Rotoiti	825	1,408,457	347.18	0.00	undefined
Hurunui	12,000	863,498	25.78	1.04	828,510
River Recovery	11,000	3,966,070	45.22	0.28	14,111,199
Mean mainland island	7,941.7	2,079,342	139.56	0.44	7,469,855

*Conservation Output Protection Years (see text for details)

Similarly Boersma *et al*. (2001) have found that species in single-species recovery plans in the USA were four times more likely to be improving in conservation status than were species included in multiple-species recovery plans.

Conclusions

The resources available are often insufficient to fund adequate, or in many cases, any specific programmes for nature conservation. In such circumstance resources must be used wisely to ensure greatest conservation gains are achieved from the limited resources available. A recent review article noted the paucity of literature on economic evaluation of nature conservation (Hughey *et al*., 2003). Despite the large expenditures by major organizations over many years on nature conservation, few if any organizations conduct more than process evaluations of nature conservation projects. A major publication in this field, for example, provided no insight into how to investigate the cost effectiveness of nature conservation projects and programmes (Jenkins & Kapos, undated). Some recent publications have, however, noted the need to emphasize economic analysis in nature conservation decision making (Possingham, 2001; Engermann *et al*., 2002, 2003), and provided rare examples of comprehensive economic evaluation of nature conservation projects.

The failure of almost all conservation agencies to apply economic analysis to nature conservation activities is a serious error, which is partially attributable to misplaced emphases by many economists on non-market valuation techniques. Those techniques can potentially inform decision makers if the benefit from conserving a species or habitat will exceed the costs of conservation, but many other conservation issues are more pertinent for decision makers. The error can be remedied by applying simple, practical economic tools such as Cost Effectiveness Analysis and Cost Utility Analysis that focus on the supply side of conservation. While these tools are not new, or problem free, they can considerably improve conservation decision making and policy analysis if used appropriately.

The application of economic tools to choice of project to manage a species, selection of most productive single species projects, and selection of most productive multi-species projects demonstrates the power of CEA and CUA in providing information to aid decision making. In comparison, non-market valuation techniques do not focus on supply of conservation activities and cannot provide useful information for such conservation decisions. Useful tools, including CEA and CUA, can be adopted by conservation agencies worldwide to assist them when completing project evaluation and making informed project selection decisions. The information those techniques can provide will allow decision makers to determine which are the most productive projects and programmes. If conservation investments are guided by information from such studies this will increase the conservation gains that can be achieved from the limited resources available for nature conservation.

Ferraro (2003) and Babcock *et al*., (1997) have argued that the efficiency gains that may be achieved by focusing on both benefits and costs in environmental projects are influenced by the correlation and relative heterogeneity of costs and benefits. In cases where the costs of projects are closely correlated with the respective benefits of projects, decision makers who select projects based solely upon their benefit rank, rather than their benefit to cost ratio, may incur only small efficiency losses. This insight may be comforting to decision makers if they have some prior knowledge of the correlation and relative heterogeneity of costs and benefits and have made decisions based solely on benefit ranks. Where this information is not available to decision makers, investment in collection of both benefit and cost data and use of appropriate economic methodologies may provide significant efficiency gains.

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Biographical sketches

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Geoff Fairburn is a financial analyst. His research focuses on evaluation of nature conservation projects and a range of finance topics.

Emma Moran is carrying out research on the evaluation of nature conservation projects, costs of species management projects, and methods for ranking species management projects.