Evaluation of predator-proof fenced biodiversity projects

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Abstract

There has been recent debate over the role of predator-proof fences in the management of New Zealand's biodiversity. The debate has arisen due to concern that investments in fenced sanctuaries are less productive than are alternative ways to manage biodiversity. Predator-proof fences are costly and budget constraints limit the area of habitat that can be fenced. The area of habitat enclosed within fences, and number of individuals of species supported, determines project's ability to contribute to biodiversity goals. Many fenced sanctuary projects require substantial, continuing volunteer input to monitor fences and other tasks. These projects often pursue a number of goals including species protection, habitat restoration, education and community engagement. In this paper we examine methods to evaluate fenced biodiversity projects. While Cost Benefit analysis can potentially be used to evaluate these projects, cost - effectiveness measures and multi criteria analyses provide useful ways to inform decision-makers.

Introduction

New Zealand has a very important and significant part to play in contributing to global biodiversity. It is estimated that 80,000 species of native animals, plants and fungi call New Zealand 'home' (Ministry for the Environment, 2007, p. 349). The term biodiversity includes the variety of "all live on earth – plants, animals, fungi and microorganisms- as well as the variety of genetic material they contain and the diversity of ecological systems in which they occur" (Australian Government – Department of Industry, Tourism and Resources, Biodiversity Management, p. 4). It comprises of genetic variety regarding the variety among individuals of a single species, species diversity, which refers to the variety of species in a particular geographical area, and ecological diversity which describes the variety of ecosystems, such as deserts and wetlands, and their interactions between them (Ministry for the Environment, 2007, p. 351).

As is the case around the world, New Zealand's ecological biodiversity is crucial in providing vital ecosystem services such as clean air and water, recycle nutrients, maintain healthy soils and regulate climates (Ministry for the Environment, 2007, 351). The biological diversity of ecosystems is very important to maintain the level of services which they provide. Much is unknown about the number of species and the extent of biodiversity. However, there is no debate within the scientific community on the importance of biodiversity for human life (Callan, & Thomas, 2007, p. 13).

The biodiversity of ecosystems in New Zealand is under pressure from introduced pest plants and animals, and human activities such as agriculture. In this regard, New Zealand has experienced one of the highest rates of loss of biodiversity in the world since the arrival of Europeans in the 18th century. Even today, around 2,500 out of the approximately 80,000 native species are listed as threatened (Ministry for the Environment, 2007, p. 349). As Atkinson (2001) highlights, New Zealand's geographical isolation led to the evolution of species which have not developed strategies to co-exist with or defend themselves against introduced species such as deer, cats, rats or possums. Overall, 25,000 plant species, 54 mammal species, and about 2,500 invertebrate species have been introduced in New Zealand since early human settlement (Ministry for the Environment, 2007, p. 356).

Controlling these pests which are "unwanted organisms that adversely affect ecosystems and directly compete with native or commercial species" (Department of Conservation and Ministry for the Environment, 2000, quoted in Ministry for the Environment, 2007, p. 393) is a very important part in biodiversity management. Pest management can include tight control of pest numbers in valuable biodiversity areas of the country, and/or Biosecurity systems to exclude unwanted organisms at the border (Ministry for the Environment, 2007, p. 393).

To sustain the current level of biodiversity, public and private organisations in New Zealand attempt to actively manage the protection of native plants and animals. The New Zealand government has introduced the New Zealand Biodiversity Strategy 2000 which reflects the country's commitment to the protection and management of its biological diversity (Ministry for the Environment, 2007, p. 361). One of the more prominent instruments to respond to the decline in biodiversity has been predator-proof fence projects (Chug, 2011). Sanctuaries such as *Zealandia* have been praised as a cost-effective way to avoid catastrophe and disaster (Clapperton & Day, 2001). Scofield, Cullen and Wang (2011) have recently challenged the perceived cost-effectiveness of such projects. This paper will outline the argument and address the question on how best to evaluate predator-proof fencing projects. I will do so by discussing three methods to analyse such projects – namely cost benefit analysis, cost-effectiveness analysis and cost-utility analysis. Biodiversity projects such as predator-proof fences in New Zealand highlight the need for transparent and well-informed project prioritization mechanisms.

Locking Adam out of Eden – is excluding humans (and other predators) from ecosystems the answer?

The effective allocation of limited funding for the protection of biodiversity is a crucial issue. With a budget of NZ$ 40 million allocated to biodiversity management, the New Zealand government is able to specifically manage about 15 percent of the 2,700 species listed as threatened (Hitchmough, et al. 2007).

As a response to the strong decline of biodiversity on mainland New Zealand, there have been several investments in predator-proof fence projects around New Zealand. Besides public organisations involved such as the Department of Conservation, private not-for-profit organizations have particularly been advocates for these fences to keep predators out of areas which host native fauna threatened by them. Scofield, Cullen and Wang (2011) attempt to answer the question on how cost effective these fence projects are for the management of biodiversity in New Zealand. According to the authors, some previous studies which compared the cost-effectiveness of fence projects and conventional pest control have been flawed. Scofield et al. (2011) highlight the importance of allocating financial resources to projects where costs and outcomes are clear and transparent. Based on their study in 2011, they consider predator-proof fences as “little more than the creation of expensive zoos” (Scofield, Cullen, & Wang, 2011). In response to Scofield et al. (2011), Innes, Lee, Burns, at al. (2011) have highlighted the many objectives and roles which pest-fenced projects have that are difficult to value in monetary terms including social, ecological and educational goals. They criticise Scofield et al. (2011) for not clearly defining key terminologies in their survey study, for misrepresenting the projects’ objectives and for not allowing appropriate time spans to assess and evaluate the effects of the projects. Following this debate the question arises which method of analysis is best suited to assess such projects as pest-fenced sanctuaries.

Conventional economic evaluation instruments such as cost-benefit analysis (CBA) are one option to consider. As Hajkowicz highlights (2008), CBA has been used in thousands of public policy decision making processes such as flood control projects in the USA. CBA focuses on allocative efficiency, and project investments are warranted if their expected benefits are in excess of the estimated costs. For such an analysis, both costs and benefits must be valued in dollar units (Hajkowicz, et al. 2008). However, application of CBA in the context of environmental policies is often challenging as there is a lack of monetary values for non-marketed environmental outcomes. Non market valuation techniques have been developed to overcome the absence of market generated values for many items, but those techniques are labour intensive and costly to complete. Benefit transfer approaches are sometimes applied to reduce the cost of completing new non market valuation studies for a specific site.

Environmental projects whose benefits are difficult to measure in dollar values might be better assessable through cost effectiveness analysis (CEA). In CEA, costs are measured in dollar value and compared to outcomes of projects (effects). As Hajkowicz et al. (2008) highlight, CEA considers only one single attribute e.g. threat status of a species. This singularity makes the comparison across a broad set of environmental projects impossible e.g. predator proof fences compared to traditional conservation projects such as systematic trapping of pests (Cullen, et al. 2001). In addition to the problem of single attributes, Hajkowicz (2008) notes that many outcomes of environmental projects may be intangible such as "improved human health, landscape scenery, biodiversity conservation, recreational opportunities and clean drinking water".

A recently developed tool to assess and rank environmental resource projects is the Investment Framework for Environmental Resources (INFFER). INFFER is suppose to enable decision makers to compare aspects such as value for money, degrees of confidence in technical information and the likelihood of achieving stated goals (INFFER, 2011). It focuses on assets which are considered to have high value from a public perspective. The assessment process has seven steps including identification of valuable assets, project development, project assessment, selection, monitoring, evaluation and adaptive management. Non-market information such as likelihood of success is included as well as market information such as project budgets. The collected information is then put into a Cost-Benefit analysis. This approach of combining market information in dollar terms and non-market information in a CBA has not been applied in the context of New Zealand biodiversity projects (David Pannell, personal communication, January 2012). It also requires the availability of information. In case of knowledge gaps, further research of data collection is required making this approach potentially costly (INFFER, FAQs, 2011).

An alternative to classic CBA and CEA is cost utility analysis (CUA). Historically applied in the context of health care economics, the method allows the measurement of "output of a program by way of utility, where utility refers to the worth of a health status" (Cullen, et al. 2001). As Hajkowicz et al. (2008) highlights, CUA is an extension of CEA in that it considers the attainment of multiple attributes and aggregates them into a utility function. Costs of alternative projects or programmes are expressed in monetary terms while benefits are expressed via a utility function. Utility functions combine indicators such as amount and timing of conservation achieved, the value of species/habitat protected, number of species covered, and the area of habitat protected. Environmental economists such as Cullen et al. (2001, 2005), Hajkowicz et al. (2008), Laycock et al. (2011) have applied CUA in environmental economics over the last decade.

In the context of predator-proof fenced sanctuaries, Scofield et al. (2011) undertook a survey of project managers of the 18 known predator-proof fenced sanctuaries of which they received 12 responses. In the survey, the authors asked project managers about funding sources, capital costs, methods of calculating depreciation, and maintenance costs. The respondents of the survey were also asked what their perceptions of achievable outcomes were. In this regard, they had to rate five types of benefits including research, ecosystem restoration, education and recreation, tourism and providing habitat for species, using a score of 1 for the most important and 5 for least important. Scofield, Cullen and Wang then compared the perceived outcomes with the actual outcomes using the stated goals of the New Zealand Biodiversity Strategy and threat assessment criteria of Townsend et al. (2008). The latter include: total population size, area of occupancy, degree of fragmentation of populations, rate of decline in total population, decline in habitat area, and predicted decline due to existing threats.

The survey showed the high costs of predator-proof fencing in New Zealand. With over 109km of fences, the area enclosed is 7133 ha. The capital costs for these fences exceeded NZ$24 million (in 2006 dollar terms). Scofield et al. were also able to highlight the high depreciation and maintenance costs involved in fence projects. They calculated that depreciation costs are around NZ$ 880,000 per year. However, they were only able to receive data from only few respondents. Therefore their calculations might include some room for error. However, Scofield, Cullen and Wang stressed the importance of these costs included in any cost-utility calculation.

As for the achieved benefits, ecosystem restoration was the main priority of organisations involved in the fencing projects. In regard to initial objectives, the provision of habitat for species was the second most important objective, in regard to perceived benefits which had been achieved, education and recreation was the second most important goal achieved. Compared to these initial objectives and achieved benefits, the actual outcomes of fencing for biodiversity showed no improvement of any species' threat status. As for the cost-effectiveness, Scofield et al. (2011) showed that for every million dollars spend on fencing projects only 297 ha of habitat have been protected. This suggests approximate costs of NZ$3,365 per hectare over the life-span of 25 years, which is one-two orders of magnitude greater compared to effective fence-free mainland islands projects which costs between NZ$11-96 per hectare per year (Scofield, et al. 2011).

In conclusion of their study, Scofield, Cullen and Wang (2011) argue for a reassessment of fenced sanctuaries and investments in them. The authors "plead for consistent, timely and more complete information on fence benefits, costs and pitfalls to be disseminated and published". They indicate that the ultimate goal of many private organisations involved in fence projects is to re-establish pre-human ecosystems. By doing so, advocates of fence projects consider ecosystems as static and ignore the dynamic evolution of them. Hence, a restoration of ecosystems to their pre-human status is impossible as Scofield et al. highlight.

Alternatives to predator-proof fences?

With biological pressure on many endemic species in New Zealand, there are a number of alternative ways that biodiversity can be managed. Legal protection of land for conservation purposes and general pest management contribute to the management of biodiversity by protecting native species in direct and indirect ways. Biodiversity management can also focus more directly on single species projects. With around 2,500 species listed as threatened, New Zealand manages around 15 % directly through extensive programmes (Moran, Cullen, & Hughey, 2005, p. 2). Specific projects can include captive breeding, translocation, pest animal control, weed control, legal actions, and education (Joseph et al., 2008, p. 332). One example of such a programme is the Kiwi sanctuaries. The programme includes five kiwi sanctuaries each protecting a different species of kiwis. These areas are specifically managed to keep pests out to allow kiwi populations to regain sustainable sizes (Ministry for the Environment, 2007, p. 399).

As highlighted earlier, the financial resources needed to fund conservation projects for all threatened species are much greater than the funding available. In New Zealand, 15% of all threatened species were directly managed in 2005 (Moran, Cullen, & Hughey, 2005, p. 2).

The Department of Conservation has only a relatively small budget of NZ$32 million per year specifically allocated to improve the status of threatened species. It is therefore crucial to allocate the financial resources in the most effective way to manage threatened species (Joseph, Maloney, Possingham, 2008, p. 329).

Moran, Cullen and Hughey (2005) stress the opportunity cost due to the budget constraint in that funding one species programmes has implications for the funding of other programmes. Opportunity costs are even more important to consider because limited funds can put species at increased risks to be extinct in the future and may increase future expenditure on efforts to save the species later. By doing so, Moran, Cullen and Hughey illustrate the high risk of not achieving the objectives of the New Zealand Biodiversity Strategy of halting the loss biodiversity by 2020 due to underfunding of projects.

Results of a study by Joseph et al. (2008) show that with factors such as cost and likelihood of success of projects included in prioritization settings, the number of species managed increases. They also highlight the trade-off between funding allocated to a greater number of cost-efficient and less risky projects and funding allocated to a smaller number of projects for species with higher value and greater project costs such as predator-proof fencing projects. This highlights that species do not possess the same value as assumed by prioritization settings which do not include cost factors.

Conclusion

Sustaining the current level of biodiversity is a challenging task. Not only is cost structures complex (Moran et al., 2005, p. 3), but also different interest groups can have effects on the outcome and efficiency of biodiversity management projects. Predator-proof fences are very likely not a sustainable and cost-effective way to achieve the goals of New Zealand's biodiversity strategy. Considering their importance, it is rather surprising to see that accurate estimates of costs of programmes are not always included in the preparation of recovery plans for threatened species and populations. As Moran et al. (2005) highlight information on the costs of programmes can contribute to a "more realistic understanding of the level of commitment required [...] and achieve greater efficiency in management" (p. 3). However, as the studies by Scofield et al. (2011) and Joseph et al. (2008) have demonstrated, the instruments such as cost-utility analysis to potentially improve efficiency in allocation of financial resources are there. It is up to policy makers to use them.

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