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Review

Assignment of measurable costs and benefits to wildlife conservation projects

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Abstract. Success of wildlife conservation projects is determined by a suite of biological and economic factors. Donor and public understanding of the economic factors is becoming increasingly central to the longevity of funding for conservation efforts. Unlike typical economic evaluation, many costs and benefits related to conservation efforts are realised in non-monetary terms. We identify the types of benefits and costs that arise from conservation projects and examine several well developed techniques that economists use to convert benefits and costs into monetary values so they may be compared in a common metric. Costs are typically more readily identifiable than benefits, with financial project costs reported most frequently, and opportunity and damage costs reported much less often. Most current evaluation methods rely primarily on cost-effectiveness analysis rather than cost–benefit analysis, a result of the difficultly in measuring benefits. We highlight improved methodology to measure secondary costs and benefits on a broader spatial scale, thereby promoting project efficacy and long-term success. Estimation of the secondary effects can provide a means to engage a wider audience in discussions of wildlife conservation by illuminating the relevant impacts to income and employment in local economies.

Additional keywords: benefit-transfer, conservation planning, contingent valuation, cost-effectiveness, cost–benefit analysis, economic evaluation, human dimensions, modeling, regional economic analysis and travel cost method.

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Introduction

Many funded projects designed to protect and promote wildlife populations of concern are required to demonstrate the returns from their biological conservation efforts, with their cost-effectiveness measured as the improvement in biological outcome per dollar spent (Busch and Cullen 2009). Annual estimates of conservation expenditures are in the billions of dollars and conservation managers and policy makers must be able to convey the degree to which resources committed to conservation projects produce success (James *et al*. 1999; Ferraro and Pattanayak 2006; Wätzold *et al*. 2006; Halpern *et al*. 2008; Kapos *et al*. 2008; Honey-Roses *et al*. 2011). Whereas success is often measured in biological terms, the benefits and costs associated with conservation projects are often unequally distributed, requiring a broader perspective of conservation-project success that includes community or regional impacts of projects on local economies (Dixon and Sherman 1991; Spiteri and Nepal 2008; Mackenzie and Ahabyona 2012). International funding agencies such as the United Nations require demonstration that their funded conservation programs achieved effective levels of protection to receive compensation for project efforts (Combes Motel *et al*. 2009; Honey-Roses *et al*. 2011). Assessing the success of conservation projects is difficult for a myriad of reasons, including a lack of resources or motivation for project evaluation, unclear project objectives, unavailable data and achievement of objectives that are outside project timelines (Kapos *et al*. 2008). Research related to the ex-post measurement of the economic efficiency of wildlife conservation projects is limited and few analyses provide guidelines on how to conduct such an analysis (Kapos *et al*. 2008).

Methods to evaluate the economic efficiency of wildlife conservation projects usually involve several trade-offs. Lack of data availability or inability to quantify benefits may drive the methods used. However, the ability to convey to donors and other stakeholders the benefits of the project per dollar spent has become one of the most crucial objectives of this type of analysis. The most common method used is cost-effectiveness analysis (CEA) and to a much lesser extent cost–benefit analysis (CBA). Improvements and innovations to these methods have led to the development of other methods, including cost–utility analysis (CUA), threat-reduction assessment (TRA) and conservation output protection years (COPY).

CBA can be used when the output of the conservation project can be assigned a monetary value (Gutman 2002; Engeman *et al*. 2002*a*, 2003; Naidoo and Ricketts 2006; Christie *et al*. 2009). For example, if the goal of the project is to increase the number of birds that have a monetary social value, this value can be used to determine whether the costs of the project were justified (Engeman *et al*. 2002*a*, 2003). Benefit–cost ratios are calculated by dividing the value of units produced by the costs, to provide a ratio of monetary value of benefits for every unit of cost.

CEA and CUA are used most commonly when analysts can quantify the impacts of the conservation project but cannot monetise them (Boardman *et al*. 1996; Naidoo *et al*. 2006; Laycock *et al*. 2009). CEA is most appropriate, for example, if a wildlife conservation project can measure the increase in the number of desirable units (such as e.g. nests, eggs, juveniles, adults) produced through different management efforts and has cost information for each management effort, but is unable to value the increase in desirable units. Economic efficiency is thereby maximised through the management approach that produces the greatest return at a given cost or that produces a given return at the lowest cost (Cullen *et al*. 2001, 2005; Engeman *et al*. 2002*a*, 2003; Caudell *et al*. 2010; Laycock *et al*. 2011). CUA is another popular alternative to CBA that is widely used by health economists to measure improvements to health status per dollar spent (Laycock *et al*. 2011; Boardman *et al*. 1996). These types of analyses also lend themselves to more sophisticated statistical examination such as multivariate regression to quantify the influence of different factors on the effectiveness or cost-effectiveness of alternative management efforts (Shwiff *et al*. 2005; Busch and Cullen 2009; Laycock *et al.* 2009, 2011).

Salafsky and Margoluis (1999) developed a TRA to measure conservation success in terms of a reduction in the threat to biodiversity. For example, instead of measuring project success by the number of birds produced by a conservation project, TRA would instead identify and measure the number of threats to bird recovery in the area before and after project implementation. Quality-adjusted-life-years (QALY) has been used for some time to compare the utility of alternative medical treatments. Conservation output protection year (COPY) was developed to serve an equivalent function in conservation (Cullen *et al*. 1999, 2001, 2005; Hughey *et al*. 2003; Laycock *et al*. 2011). Basically, COPY is a time-weighted measure of improvement in species status. COPY estimates from different conservation plans can be compared, which gives an indication of relative efficiency. Integrating project costs into both the TRA and COPY measures allows for the calculation of cost–TRA and cost–COPY ratios, which provide information on the cost per unit of threat reduction or cost per increase in conservation output protection per year (Laycock *et al*. 2011).

Determining the benefits of biological conservation programs can be extremely challenging; the determination of costs is often more straightforward. Benefits are usually derived from increased production or avoided loss of the species of interest (Kapos *et al.* 2008). It is likely that these species do not have any standard market value, and even when there is some market value, it may understate the species' full social value. Non-market values must be determined by other methods, such as contingent valuation (CVM), travel-cost (TCM), benefit-transfer, or other methods. In addition to primary benefits associated with the conservation of a particular species (e.g. existence value, stewardship value), there may be secondary benefits that accrue to the economy as a result of increased consumptive (e.g. hunting or fishing tourism) or non-consumptive uses (e.g. viewing tourism) of the species. Economic analyses of conservation programs could engage a broader group of stakeholders by estimating the impacts of conservation beyond the primary benefits to include changes in ecosystem services such as increases in harvestable animals and the regional economic implications of conservation outcomes such as increased tourist spending. Engaging a broader group of stakeholders (e.g. the general public) is vitally important to conservation projects because individuals care about the economic impact of wildlife species and factor this into their wildlife conservation decisions (Martín-López *et al*. 2008).

Central to all methods used in economic evaluation of conservation projects is the determination of primary costs. We provide examples of primary project-cost determinations as well as some methods to assess primary benefits. We also highlight some of the shortcomings of each method. Although these methods have been discussed extensively in the conservation literature, the present review provides the framework for linking the primary benefits and costs to the estimation of secondary benefits and costs that arise in local or regional economies as a result of conservation projects. Estimation of secondary effects could provide a means to engage a broader audience in discussions of wildlife conservation by illuminating the relevant impacts to income and employment in local economies.

Determining project costs

There are often many types of costs associated with the implementation of conservation projects (Naidoo *et al*. 2006, 2008; Jantke and Schneider 2009; Adams *et al*. 2010; Armsworth *et al*. 2011; Schneider *et al*. 2011) including acquisition costs, management costs, transaction costs, damage costs and opportunity costs (Fig. 1). Most costs vary depending on the size and location of the conservation project parcel, whereas some costs may be fixed (Jantke and Schneider 2009; Naidoo *et al*. 2008; Armsworth *et al*. 2011; Schneider *et al*. 2011). Project costs are typically assessed either before project initiation when the project is in the planning phase or after project as an assessment of overall project performance. If costs are addressed in the planning phase of a project, often proxy or surrogate costs are used to approximate the actual costs (Adams *et al*. 2010). In the planning process before the initiation of a project, many studies have indicated that the inclusion of estimates for all five cost components is difficult, if not impossible (Cullen *et al*. 2005; Naidoo and Ricketts 2006; Jantke and Schneider 2009; Naidoo *et al*. 2008).

Acquisition, management and transaction costs represent the financial costs of project implementation and typically involve land purchase and/or lease, land management, equipment, labour, supplies, planning, negotiating and other costs crucial to project completion and management. These costs can be calculated by keeping good financial records of all aspects of expenditures related to the project for a post-project assessment of costs.

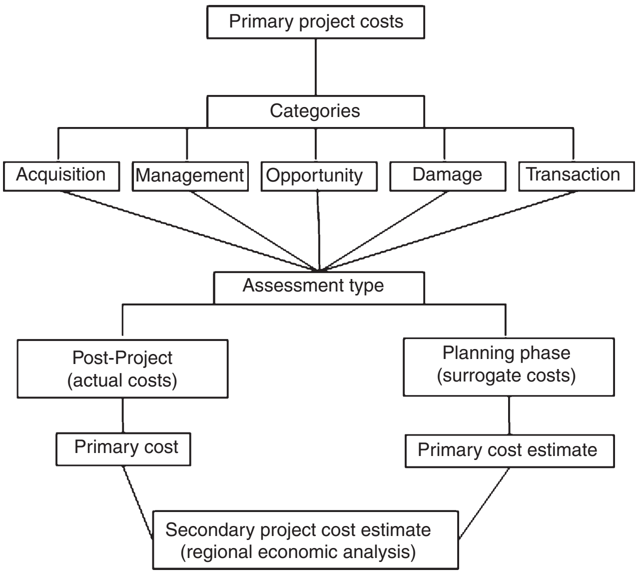

Fig. 1. Framework for assigning cost values to conservation projects.

Studies indicate that site area is the most important driver of acquisition costs, although management and opportunity costs may also be important (Naidoo and Ricketts 2006; Armsworth *et al*. 2011). Management costs often exhibit economies of scale or cost advantages in relation to site-area expansion, which has implications for economic efficiency of site selection (Armsworth *et al*. 2011). Land use surrounding a particular site can also have cost implications (i.e. highly productive cropland *v.* poor productive value). Many examples from the literature suggest that site area has implications for all types of costs and can be a significant driver in damage and opportunity costs (Naidoo and Ricketts 2006; Rondinini *et al*. 2006; Rondinini and Boitani 2007; Mackenzie 2012).

Damage costs, which are also known as spill-over costs, arise from the conservation project but are a burden to those outside of the project. It has been suggested that communities surrounding the conservation area can disproportionately bear the burden of these types of costs, which can have an impact on social acceptability of these types of projects (Nyhus *et al*. 2005; Ninan *et al*. 2007). Examples from the literature of damage costs include livestock predation, crop losses, exclusion from resources, job loss, eviction from areas around a park and others (Butler 2000; Nyhus *et al*. 2000; Ferraro 2002; Naughton-Treves and Treves 2005; Brockington and Igoe 2006; Brockington *et al*. 2006; Cernea and Schmidt-Soltau 2006).

Opportunity costs in a conservation project framework usually arise from reduced agricultural production, lost recreational opportunities, loss of competing species or habitat, increased human conflicts and other forgone uses of the conserved land (Naidoo and Adamowicz 2006; Naidoo and Ricketts 2006; Jantke and Schneider 2009; Adams *et al*. 2010; Naidoo *et al*. 2008; Armsworth *et al*. 2011; Schneider *et al*. 2011). These costs often are more burdensome at the local level, affecting communities surrounding the conservation site the greatest (Adams and Infield 2003). For example, conservation projects may decrease the amount of agricultural commodities grown, financially burdening local communities (Emerton 1999). Valuing the loss of agricultural production is possible, given a variety of techniques including geo-spatial mapping, direct market valuation and regional economic modelling (Naidoo and Ricketts 2006). The use of geo-spatial mapping technologies has allowed for significant improvements to the estimation of opportunity costs in terms of land values. Regional economic modelling allows for the calculation of secondary costs by estimating the 'multiplier impacts' of alternative land uses such as agricultural production (see the section on regional economic analysis).

Capturing the opportunity costs associated with conservation projects is difficult for some of the same reasons as capturing the benefits of these projects. For example, if a conservation project involves restricting access of tourists to a particular area, then the value of that area to the tourists must be estimated using one of the methods described in the benefits section. Similarly, if competing habitat (e.g. other conservation projects, bioenergy plantations or intensively managed forests) or species must be removed, that habitat or species must be monetised and included in the cost calculation (Jantke and Schneider 2009).

Assigning benefit values

The primary purpose of wildlife conservation projects is typically to maintain (avoid damages to) or increase a targeted wildlife species' population size. Success is commonly measured as the number of animals protected or the increase in the number of animals at the end of the project (Kapos *et al*. 2008). Conservation efforts rarely involve species that have an observed market value and therefore require techniques that can estimate value in the absence of any market values. The primary benefit of improving or maintaining wildlife populations may also give rise to secondary benefits that may be estimated using ecosystem service valuation methods and regional economic modelling. Valuation of wildlife conservation and associated spill-over benefits can occur through survey methods such as the CVM and TCM, and non-survey methods, such as benefit-transfer, civil penalties and replacement costs (Fig. 2).

Contingent valuation method (CVM) is a survey-based, stated preference approach used to estimate use and non-use values associated with wildlife species (Kotchen and Reiling 1998). This method solicits responses from individuals regarding their willingness to pay (WTP) for increased wildlife populations. Questions usually describe the outcome of the conservation effort to be valued and then ask individuals if they would pay a certain amount to achieve that outcome. By varying the amount individuals are asked to pay among respondents, a social value of the outcome is constructed (Loomis 1990). CVM has been used extensively in conservation studies, especially to examine habitat conservation associated with wildlife species (see Loomis and Walsh 1997 for an extensive discussion and examples of this method). Chambers and Whitehead (2003) estimated willingness to pay for wolf management and a wolf-damage plan in Minnesota by using the CVM. Their survey was specifically designed to capture both use and non-use values of wolves. They found that aggregate willingness to pay in Minnesota for a management plan that included wolf population and health

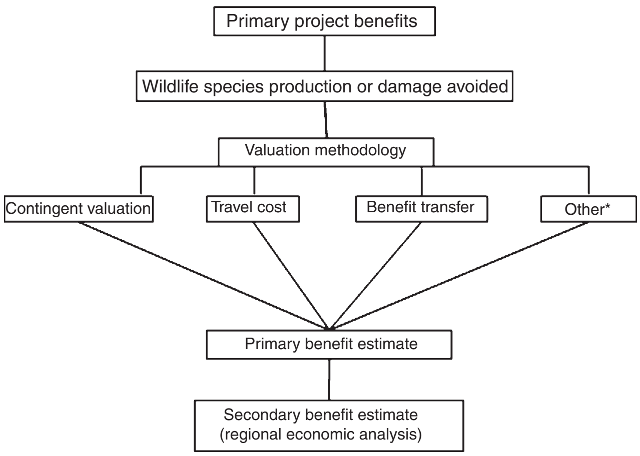

Fig. 2. Framework for assigning benefit values to conservation projects. *All other methodologies not explicitly listed.

monitoring, habitat protection and depredation control was $27 million.

Several factors can affect WTP for wildlife conservation, including the species' usefulness and likeability, information level of respondents, level of economic damage created by the species, and questionnaire design (Brown *et al.* 1994, 1996; Nunes and van den Bergh 2001; Bateman *et al.* 2002; Tisdell and Wilson 2006; Martín-López *et al.* 2007, 2008).

Criticisms of CVM include the hypothetical nature of the questionnaire and the inability to validate responses, causing some to question its usefulness for determining benefits (Eberle and Hayden 1991; Champ *et al.* 2003). Additionally, public goods such as wildlife do not lend themselves to valuation in this manner and, further, this type of valuation typically understates the true non-market value (Pearce and Moran 1994; Balmford *et al.* 2003). To overcome some of these potentially serious issues, surveys must be written appropriately to reduce potential uncertainty and biases (Ekstrand and Loomis 1998; Martín-López *et al.* 2007, 2008). Surveys can be expensive to implement, however, and it may be difficult to identify the target audience. Applications of CVM are increasing in the literature, especially those concerning the value of land that is the target of wildlife conservation efforts (Christie *et al.* 2009).

The TCM is another survey approach that uses costs incurred for travel to quantify demand for recreational activities linked to a species of interest (Kotchen and Reiling 1998). TCM is based on the idea that as some environmental amenity changes, the amount people are willing to pay to use it will change, and that change in willingness to pay is revealed by a change in travel costs (see Loomis and Walsh 1997 for an extensive discussion and examples of this method). For example, suppose a conservation project improves a fish population in a river relative to other, similar rivers. If this improvement is valuable from a recreation standpoint, the river will be used more intensively and the amount of money people spend using it will increase relative to other rivers. Thus, the increase in travel costs becomes a surrogate for the willingness to pay for the outcome of the conservation effort. Zawacki *et al.* (2000) used the TCM to estimate the demand for and the value of non-consumptive wildlife-associated recreation access in the USA by using the National Survey of Hunting, Fishing and Recreation to provide estimates of travel costs; the authors provide per-trip estimates of the benefits of wildlife viewing. These benefits can then be used within a CBA framework.

Criticisms of this method include concerns about the assumption that visitors' values equal or exceed their travel costs. Critics argue that travel costs are simply costs, not an accurate representation of the value. Another concern is that this method requires values to be assigned to the time individuals spend traveling to a site. It is difficult to assign accurate values to the opportunity cost of travellers' time because each person values their time differently, depending on their occupation or the activity they gave up in order to travel to the site. Additionally, applicability of this method may be limited in a conservation setting because not only may human access be limited to conservation sites but human awareness or preference towards the species associated with a chosen recreation site may be limited. If individuals are not willing or able to travel to the conservation site to expend funds, then this method confers no value.

The benefit-transfer method relies on benefit values derived from CVM and TCM studies in one geographical location and species, which are then transferred to another location and similar species. Adjustments to the values can be made by factoring in differences in incomes or prices from one area to the other. Naidoo and Ricketts (2006) used this method to assign bioprospecting values to benefits of land conservation by using data from a previous study that had assessed the WTP of pharmaceutical companies for the potential of tropical forests to contain precursors to new marketable drugs. After making adjustments to the per-hectare value of tropical forests calculated in the previous study, this value was used as a benefit-transfer value to approximate the value of a tropical forest in a different location.

Typical criticisms of this method focus on the reliability of value estimates because this method usually derives its estimates from CVM or TCM (Brouwer 2000; Smith *et al.* 2002). Other criticisms arise from the belief that wildlife in one area are unique and simply transferring the value associated with a species in one location to the same species in another location does not capture local qualities. Although this view may be common, studies have indicated that average values of species are relatively close, regardless of location (Rosenberger and Loomis 2001).The valuation methodologies outlined thus far have focussed on eliciting individuals' preferences for wildlife conservation. However, preferences and willingness to pay for those preferences may not account for all of the benefits of wildlife conservation projects. Ecosystem services are the beneficial functions provided by the ecosystem, such as the production of harvestable plants and animals and the provision of clean water or scenic landscapes (Hanley *et al.* 2007). Wildlife conservation projects often increase the quantity or quality of ecosystem services and it may be possible to estimate the value of the improvement. In relation to wildlife conservation projects, estimation of the value of improved ecosystem services proceeds in three steps. First, the nature and size of the environmental change affecting the ecosystem structure and function driven by the wildlife conservation project must be determined. Second, the value of the ecosystem service that has been affected must be estimated either through market prices or non-market valuation techniques. Finally, a change in social welfare can be estimated on the basis of the extent of the change in ecosystem services and the value of that service (Freeman 2003). For example, Naidoo and Ricketts (2006) calculated ecosystem service values for harvestable timber and bushmeat related to land conservation in Paraguay. Landsat imagery and ground data were used to estimate the change in the production of harvestable timber, and biological information on game species was used to derive estimates of the change in species-specific fraction of biomass that could be sustainably harvested. This information was combined with applicable market prices to determine the expected flow of benefits provided by each hectare of land conserved.

Regional economic analysis

Regional economic analysis (REA) allows for the estimation of secondary benefits and costs associated with the conservation of wildlife species in units of measure that are important to the general public (e.g. revenue, income and jobs). Conservation projects that increase wildlife populations (the primary benefit) may generate measurable secondary benefits such as increased tourism (both consumptive and non-consumptive) (Duffield 1992; Wilson and Tisdell 2003). Increases in tourism have benefits to the regional economy that can be measured through the use of regional economic models such as impact analysis for planning (IMPLAN , Minnesota IMPLAN Group) and regional economic modelling (REMI Inc.).

Static regional economic models exist to estimate 'multipliers' by modelling changes in economic activity stemming from changes in final demand for a particular good (i.e. goods and services associated with birdwatching). Input–output (IO) models are the most widely used tool for modelling the linkages and leakages of a regional economy. IO models use transaction tables to illustrate how outputs from one industry may be sold to other industries as intermediate inputs or as final goods to consumers, and how households can use wages from their labour to purchase final goods (Richardson 1972). This allows for the tracking of annual monetary transactions between industry sectors (processing), payments to factors of production (value added) and consumers of final goods (final demand).

Loomis and Richardson (2001) provided an example of regional economic analysis, in which they estimated the value of the USA wilderness system. They began by estimating primary benefits to visitors of wilderness areas by using established WTP estimates from existing CVM and TCM studies. They then pointed out that secondary benefits exist because visitors spend approximately \$30 per day in the local economy. Tourists' dollars flow through the economy and support other economic sectors, which provide regional jobs and revenue (Shwiff *et al.* 2010). To capture the 'community effect' of this spending, they used a regional IO model (IMPLAN) to estimate the impacts of tourism spending on regional jobs and revenue. Last, they used this information to calculate the expected economic impact if more land was added to the wilderness system. In another example, Duffield (1992) showed that conservation programs designed to increase the number of wolves in and around Yellowstone National Park area have also increased tourism to the park, which has increased economic activity in and around the park.

Arguably, economic impacts generated by conservation projects are dynamic, and therefore require a regional economic model that can account for complex interactions among economic sectors over multiple time periods. A dynamic forecasting and regional economic-policy modelling tool has been developed to generate annual forecasts and simulations that detail behavioural responses to compensation, price and other economic factors (REMI: Model Documentation – Version 9.5). The REMI model incorporates inter-industry transactions, endogenous final-demand feedbacks, substitution among factors of production in response to changes in expected income, wage responses to changes in labour-market conditions, and changes in the share of local and export markets in response to change in regional profitability and production costs (Treyz *et al*. 1991). The dynamic nature of REMI enables it to create a control (baseline) forecast that projects economic conditions within a region on the basis of trends in historical data. Economic impacts are then examined by comparing the control forecast to simulations which can model changes to different policy variables including industry-specific income, value added and employment.

Modelling impacts in this way can translate conservation efforts into regional (e.g. local, state, province) impacts on revenue and jobs, expanding the general public's perception of conservation benefits. Caution must be used, however, because secondary benefits (or costs) cannot be incorporated into CBA models because losses in one region may become gains in another region, leading to potentially offsetting effects. However, these secondary impacts can help estimate the total impact of conservation efforts and engage a broader audience by highlighting implications of conservation efforts for local communities.

Regional economic models are used significantly more in North America and Europe than the rest of the world, which has resulted in the development of multipliers for these economies. However, if regional models are unavailable for a specific region, multiplier estimates can be used from other regions as a proxy. In the USA, the Bureau of Economic Analysis provides regional economic multipliers for state and local economies, produced through their regional input–output modelling system (see www.bea.gov/regional/pdf/overview/Regional_RIMS.pdf, verified 26 September 2012).

Multipliers for the production of agricultural commodities and tourist impacts have been relatively well researched. Suppose, as an illustrative example, that an income multiplier for wheat production in Region X is 1.4, indicating that for every dollar generated in the production of wheat, US\$1.40 is generated in the regional economy. Suppose also that the initiation of a conservation project in region Y will cause 10 ha of wheat production to be forgone. In the absence of the ability to run a REA in Region Y, the wheat multiplier from Region X can act as a proxy for forgone wheat production. This is similar to a benefit-transfer methodology, but for multipliers, and provides a broader estimate of the opportunity costs associated with forgone agricultural production as a result of conservation efforts.

Discussion

The present paper has reviewed methods for estimating primary benefits and costs of wildlife conservation projects and broader secondary impacts. Multiple valuation approaches may be used to account for benefits and costs from all types of uses (e.g. CVM for non-use values; TCM for use values); however, special attention needs to be paid to avoid double-counting the same benefit or cost via multiple approaches. This review also provides some example applications from the existing literature of methods to estimate primary and secondary impacts of wildlife conservation projects to regional communities. Estimation of secondary impacts generates useful information about benefits and costs to local economies.

While examining methods to assign benefits and costs to conservation projects, several useful insights arose. First, site selection is important because the designation of habitat into conservation status is likely to provide the largest source of potential secondary impacts. This is because habitat has many alternate uses, the value of which can be accounted for through economic modelling. Therefore, when possible, optimal site selection will involve a site that has low alternative use values, high ecosystem service values in its current or improved state (e.g. tourism, natural resource harvest, carbon storage), adequate size to achieve economies of scale in management efforts and a location removed from potential conflict areas.

A second insight is that estimation of secondary impacts (e.g. revenue, income and jobs) is crucial to creating a more-complete picture of the value of conservation projects. Many studies we have cited describe ‘global’ benefits of conservation projects, such as species preservation, carbon sequestration and bioprospecting. However, conservation projects may also generate more localised costs that primarily affect communities surrounding the conservation site. This creates a potentially serious imbalance that can undermine project success. When local communities derive few benefits, but bear a disproportionate amount of the project’s costs, then its long-term success may be in jeopardy. A variety of methods exist to estimate primary and secondary impacts of wildlife conservation. Methods such as regional economic modelling and ecosystem service valuation broaden the scope of results to engage a larger audience. If the general public can gain an understanding of the potential impacts to local communities resulting from conservation efforts in their region, this will likely have significant influence on project acceptability and success.

Conservation project managers that have an understanding of the potential economic impacts before the initiation of the project can garner support by focusing on maximising benefits or minimising costs. For local or regional economies, tangible secondary impacts, especially to income and revenue, will be paramount to project success. Future donors and stakeholders of conservation projects will likely look for projects that accomplish biological goals while demonstrating economic efficiency. In the present manuscript, we have provided a blueprint for the design of wildlife conservation projects to better achieve conservation goals in an economically efficient manner, thereby ensuring project longevity and wildlife protection.

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