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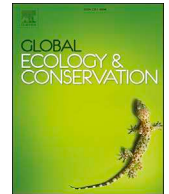
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Original Research Article

Local costs of conservation exceed those borne by the global majority

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ABSTRACT

Cost data are crucial in conservation planning to identify more efficient and equitable land use options. However, many studies focus on just one cost type and neglect others, particularly those borne locally. We develop, for a high priority conservation area, spatial models of two local costs that arise from protected areas: foregone agricultural opportunities and increased wildlife damage. We then map these across the study area and compare them to the direct costs of reserve management, finding that local costs exceed management costs. Whilst benefits of conservation accrue to the global community, significant costs are borne by those living closest. Where livelihoods depend upon opportunities forgone or diminished by conservation intervention, outcomes are limited. Activities can be displaced (leakage); rules can be broken (intervention does not work); or the intervention forces a shift in livelihood profiles (potentially to the detriment of local peoples' welfare). These raise concerns for both conservation and development outcomes and timely consideration of local costs is vital in conservation planning tools and processes.

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1. Introduction

Spatially explicit land use planning for conservation has advanced substantially through the modelling of multiple ecosystem service values alongside biodiversity (Bateman et al., 2013; Cimon-Morin et al., 2013). This work has highlighted the risk of making suboptimal choices if not all of the benefits that can arise from different land use decisions are accounted for (Bateman et al., 2013). Likewise, multiple costs exist and there is broad and increasing recognition of the importance of considering economic costs in biodiversity conservation planning (Ando et al., 1998; Armsworth, 2014; Naidoo et al., 2006; Polasky, 2008). For many parts of the world, however, there are limited data on the spatial distribution of costs, and even fewer studies on how who bears them (although see Badola et al., 2010; Emerton, 1999; Ferraro, 2002; Franks, 2008). This is all the more important because the benefits of conservation disproportionately accumulate to the international community, whilst we expect the costs mainly to accrue locally (Badola et al., 2010; Balmford and Whitten, 2003; Franks, 2008; Ghate, 2003; Wells, 1992). These issues feed directly into important wider debates about who should pay for conservation, where those payments should be directed, and by what mechanism should they be made (Balmford and Whitten, 2003; Ferraro and Kiss, 2002; Pechacek et al. 200) as well as around the integration of strategies for poverty reduction and biodiversity conservation (Adams et al., 2004).

There are three important uses of cost data in conservation. First, they can greatly improve cost effectiveness of systematic conservation planning because spatial variation in cost is frequently greater than (and not congruent with) spatial variation in biodiversity value (Di Minin et al. 2013; Naidoo et al., 2006). Second, conservation measures are regularly justified by the value of ecosystem services that they generate. However, net value cannot be calculated unless the cost of providing or maintaining those services is also known (Badola et al., 2010; Balmford et al., 2011). Third, whilst conservation leads to increased welfare amongst some in society, it can impose a burden upon others (e.g. Cernea and Schmidt-Soltau, 2003; Cernea and Schmidt-Soltau, 2006). Recognising and addressing inequalities in the distribution of costs and benefits is essential to developing equitable and effective conservation interventions (Di Minin et al. 2013; Ferraro, 2002; Pechacek et al., 2013; Sheil et al., 2013; Turner and Fisher, 2008).

Protected areas (hereafter “reserves”) remain instrumental in current conservation efforts (Joppa and Pfaff, 2011). In much of the biodiverse tropics, however, data on the local costs that reserves impose – such as opportunity and damage costs – are scarce (although see Hariohay and Røskaft, 2015; Rantala et al., 2013) and rarely are they spatially explicit (Naidoo and Iwamura, 2007). When land is set aside for conservation, an opportunity cost results, which is at least equal to the net benefits obtained if the land were available instead for development to some other productive use (Naidoo and Adamowicz, 2006). Damage costs are those associated with the elevated wildlife damage to crops or property that occur around reserves (Naidoo et al., 2006).

It is timely, then, to compare multiple costs in a high priority conservation area where development and conservation concerns compete for finite resources and attention. We consider the magnitude and distribution of these three cost types – reserve management costs, wildlife damage costs and opportunity costs – and highlight the importance of considering local costs in conservation planning.

2. Methods

2.1. Study area

The Eastern Arc Mountains (EAM) in Tanzania support tropical moist forests with exceptionally high species richness and endemism (Burgess et al., 2007). They also support miombo woodland that has lower biodiversity value but which, as an important source of ecosystem goods such as charcoal, and firewood, is converted at much higher rates (Green et al., 2013; Lund and Treue, 2008).

To understand how the costs of conserving a tropical biodiversity hotspot are distributed, we compare subnational spatial models of three conservation costs that vary in their distribution from local to international. In the EAM the direct costs of reserve management, borne nationally and internationally, have been modelled spatially (Green et al., 2012). We also develop models for opportunity cost and damage costs, which villagers living adjacent to reserves elsewhere in Tanzania considered the most important local costs (Kideghesho & Mtoni, 2008).

Below we first describe the data that we used to estimate income from farming through, which came from existing literature and from our own farmer survey. We then describe the models used to estimate opportunity cost and calculate damage cost, and finish by describing the spatial data and units of analysis.

2.2. Yield data

We use estimated yields of maize and bean across East Africa (approx. 18.5 km resolution; Thornton et al., 2009) under current smallholder farming practices for sowing time, planting density and fertiliser application. Maize is a vital food crop in East Africa (Thornton et al., 2009) and a farmer survey in 2009 (below) showed that it accounted for 63% of sampled fields in the EAM. We also mapped bean yields for areas in which the climate allows two harvests per year: maize in season one, followed by beans in season two. This second harvest is important to annual farm productivity (Thornton et al., 2009).

2.3. Farmer survey

During August–October 2009, we surveyed 135 farmer households from 23 EAM villages. We collected information on planted crops, yields, prices, and labour. We also asked whether the farm experienced crop damage and, if so, the species causing the damage, its frequency and extent, and the measures taken to guard crops. Median maize and bean farm gate prices were 0.2 and 0.6 USD/kg, respectively (maize: range = 0.05–0.7 USD/kg, $n=58$; bean: range = 0.4–0.7 USD/kg, $n=12$). 54% of fields were planted with bought seed at a rate of 20 kg/ha for maize and 66 kg/ha for bean (maize: range = 2–100 kg/ha, $n=72$; bean: range = 3.5–328 kg/ha, $n=26$). We multiplied seed application rates by crop price to derive the input cost for seed. Annual application of N was assumed to be 5 kg/ha at 0.6 USD/kg (Thornton et al., 2009). Annual household labour input was 55 days/ha for maize farming and 49 days/ha for bean farming. Labour was multiplied by the median unskilled wage available to surveyed villagers (1.7 USD/day; range = 0.4–4.0 USD/day; $n=16$). We calculated net rent by multiplying crop yields by their market value and subtracting seed, labour and fertiliser costs (Equation (1)).

$$NetRent_{it} = (Y_i \times P) - (S + F + L) \quad (1)$$

where $NetRent_{it}$ is the annual net rent from maize or bean production on a one hectare land parcel i , Y_i is the yield (kilograms) per hectare, P is the price of maize or bean per kilogram, S is the costs of seed per hectare, F is the fertiliser cost per hectare and L is the labour cost per hectare. We did this separately for maize and beans; wherever farming maize or bean was profitable (i.e. net rent exceeded zero), we summed profits across seasons (maize plus bean) to derive annual net rent.

2.4. Opportunity costs of reserves

The opportunity cost of reserves is calculated from the most likely profitable alternative if the land were not conserved. Where market data are unavailable, land price can be estimated from the expected net present value of future profits from the land, which will vary across space, reflecting changes in land quality and transportation costs (Naidoo and Adamowicz, 2006). We assume that the most likely alternative use in the EAM is agriculture, which accounts for >80% of Tanzania's employment and 30% of its gross domestic product (Arndt et al., 2012). In addition to benefits from agricultural production, we also quantify one-off “windfall” benefits. Upon conversion, we assume that woody biomass is processed and sold as charcoal, given the role of charcoal production as a major mechanism of forest and woodland conversion across Africa (Fisher et al., 2011). We used published data on charcoal value (net of licensing, production and transportation costs), which vary for different land covers: 400–900 USD per ha for woodland and 1450–2,000 USD per ha for forest (Fisher et al., 2011).

The opportunity cost of land is calculated from its net agricultural and charcoal value multiplied by the probability that it will be converted (Naidoo and Adamowicz, 2006). Thus we calculate the expected net value (bounded between zero and the expected rent from the land if it were converted immediately), rather than the potential net value. An area of land may be highly fertile, but if conversion is unlikely due, say, to inaccessibility, this is important for its valuation (Schaafsma et al., 2012). We use a published model of conversion probability for all remaining forest and woodland in the study area (Green et al., 2013). This model was parameterised for forest and woodland outside of statutory protection (driven mostly by topography and accessibility variables) and extrapolated to all forest and woodland, including that which lies within reserves. To calculate present values of future costs, we use a $r = 15\%/y$ discount rate, applied over a 25-year time horizon, but also report results for $r = 5\%/y$ and $r = 20\%/y$. These reflect the median (15%), minimum (4%) and maximum (21%) discount rates reported by the Bank of Tanzania from December 2005–November 2010.

We first calculate the net present value of harvest of a given cell in any particular year (Equation (2)).

$$Ag.NPV_{it} = \frac{(NetRent_{mi} + NetRent_{bi})}{(1 + r)^t} \quad (2)$$

where $NetRent_{mit}$ and $NetRent_{bit}$ are the expected net rent from maize and bean, respectively, for a particular land parcel i in year t (Equation (1)), and r is the discount rate. We assume that the probability of conversion, p , remains constant and that, once converted, land remains farmed. We then multiply the probability that the cell has been converted by any particular year (Equation (3); Green et al., 2013) by the net present value of farming in that year, $Ag.NPV_{it}$, to give an expected value of conversion in each year. Summing this across a 25-year time horizon, gives the opportunity cost of agriculture if that cell is set aside for conservation (Equation (4)).

$$p_{it} = 1 - (1 - p_i)^t \quad (3)$$

$$OppCost_{agric} = \sum_{t=1}^{25} Ag.NPV_{it} \times p_{it} \quad (4)$$

Similarly, we calculate the net present value of charcoal production $Ch.NPV_{it}$ for each year $t=1 \dots 25$. We assume all charcoal benefits accrue at the time of conversion. Therefore, the probability of conversion in each year, p_{it} , is multiplied by $Ch.NPV_{it}$ to

estimate the expected charcoal value of conversion in each year. We sum across a 25-year time horizon to derive the opportunity cost of charcoal (Equation (5)).

$$OppCost_{ch} = \sum_{t=1}^{25} Ch.NPV_{it} \times p_{it} \quad (5)$$

2.5. Wildlife damage costs of reserves

Damage goes largely unreported in Tanzania and, although we collected data on crop damage, these were insufficient to develop reliable spatial models. However, the most consistently reported predictor of damage by wildlife in Tanzania and elsewhere is distance to reserves (e.g. [Hariohay and Røskaft, 2015](#); [Mackenzie and Ahabyona, 2012](#); [Naughton-Treves and Treves, 2005](#)). Therefore, we estimated damage costs using work by [Naughton-Treves and Treves \(2005\)](#) that was conducted in parts of East Africa with similar farming techniques and damaging species to those found in our study area ([Kideghesho, 2008](#); [Green, 2012](#)). This work demonstrates that over 90% of wildlife damage from a reserve occurs within 0.2 km of its boundary. Within this zone, mean damage varies from 4 to 9.4% of crop yields ([Naughton-Treves and Treves, 2005](#)). Therefore, to estimate damage costs, we constrained the rent derived from farming to currently cultivated areas and quantified losses of 4%, 7% and 10% of yield within 0.2 km of habitat borders. For consistency with estimates of opportunity cost we focus on damage to maize and beans. Maize was also the most commonly damaged crop in our farmer survey (48% of events) and in an earlier studies ([Kideghesho, 2008](#); [Mc Guinness and Taylor 2014](#)).

2.6. Spatial analyses

Land cover data (1ha resolution) from a survey by [Hunting Technical Services \(1997\)](#) were updated to the year 2000 by local experts ([Swetnam et al., 2011](#)). We estimated human population density within and adjacent to the reserve network using the derived product LandScan 2008 ([Bright et al., 2009](#)). This layer is a modelled surface, based upon administrative boundaries, land cover data, topography, roads and population centres. We further modified it to exclude populations from National Parks and Game Reserves and to match ward-level census data for the year 2002 ([NBS, 2002](#); [Platts et al., 2011](#)). Reserve boundaries are from [IUCN & UNEP-WCMC \(2010\)](#), modified to correct errors ([Larrosa, 2011](#)). Reserve management costs were from a spatially explicit model for the EAM, which extrapolates managers' estimates of necessary expenditure to all areas (including non-reserves) based on topography and human pressure ([Green et al., 2012, Table 1](#)).

We mapped all costs to all areas for several reasons. First, we wanted an EAM-wide assessment of each cost to allow us to make a consistent comparison between the three. This also ensures that we do not bias our comparison by the tendency for reserves to be sited in areas of lower human pressure ([Joppa and Pfaff, 2009](#)). Second, in demonstrating the costs of conserving the remaining natural habitat in the hotspot (forest and woodland), we also argue that unpaid opportunity and damage costs for existing PAs may still need to be met if conservation is to be accepted and supported by local people.

The median size of state-owned reserves in the EAM is 9 km², so we chose this as an appropriate scale for modelling damage costs and management costs across the study area ([Green et al., 2012](#)). We also aggregated opportunity cost estimates to this resolution (3 km × 3 km) to allow for direct comparison. We modelled costs across the study region, both inside and

Table 1

Comparison of cost data layers. For all three cost types, protected area shapefiles are from UNEP-WCMC (2010) and annual costs are discounted over a 25-year time horizon, using discount rates of 5%, 15% and 20%, to estimate net present cost.

Data layer	Description of derivation	Data sources	Resolution
Management cost	Data from a survey of 40 protected area managers in the Eastern Arc Mountains, responsible for 96% of the mountains' protected areas, were modelled according to human population pressure. This model was extrapolated to the entire Eastern Arc Mountains.	<ul style="list-style-type: none"> • Questionnaire and modelling procedure: Green et al. (2012). • Population density: Landscan 2008 (Bright et al., 2009) • Population pressure calculation: Green et al. (2012). 	Data modelled at 9 km ² , the median size of protected areas in the Eastern Arc Mountains.
Opportunity cost	Input costs and farm gate crop values were collected during farmer interviews. These were combined with a regional map of crop yields to estimate net potential value. An Eastern Arc Mountain-specific model of conversion probability was then applied to derive net expected value.	<ul style="list-style-type: none"> • Crop yield: Thornton et al. (2009). • Farmer interviews: this study. • Conversion probability: Green et al. (2013). 	Data were modelled at 0.1 km ² for estimating the cost of the existing reserve network. Data were also aggregated to 9 km ² for comparison with other data layers.
Damage cost	We used farmer interviews to determine the type of species causing crop damage and farmers' responses. These data were used to identify crop damage studies that were conducted in similar farming systems those in the Eastern Arc Mountains and with a similar subset of damaging species. Using local data for farm productivity and land cover, we then applied the simple models described in these studies to the Eastern Arc Mountains.	<ul style="list-style-type: none"> • Farmer interviews: this study • Damage models: Naughton-Treves and Treves (2005). • Crop yield: Thornton et al. (2009). • Interview data: this study. • Land cover: Hunting Technical Services (1997) updated to year 2000 (Swetnam et al., 2011). 	Data were modelled for 9 km ² pixels to illustrate their variation across the study region (Fig. 1b). For all other calculations, however, they were applied only to boundaries between either forest/woodland and cropland or protected areas and cropland (land cover data at 0.1 km ² resolution).

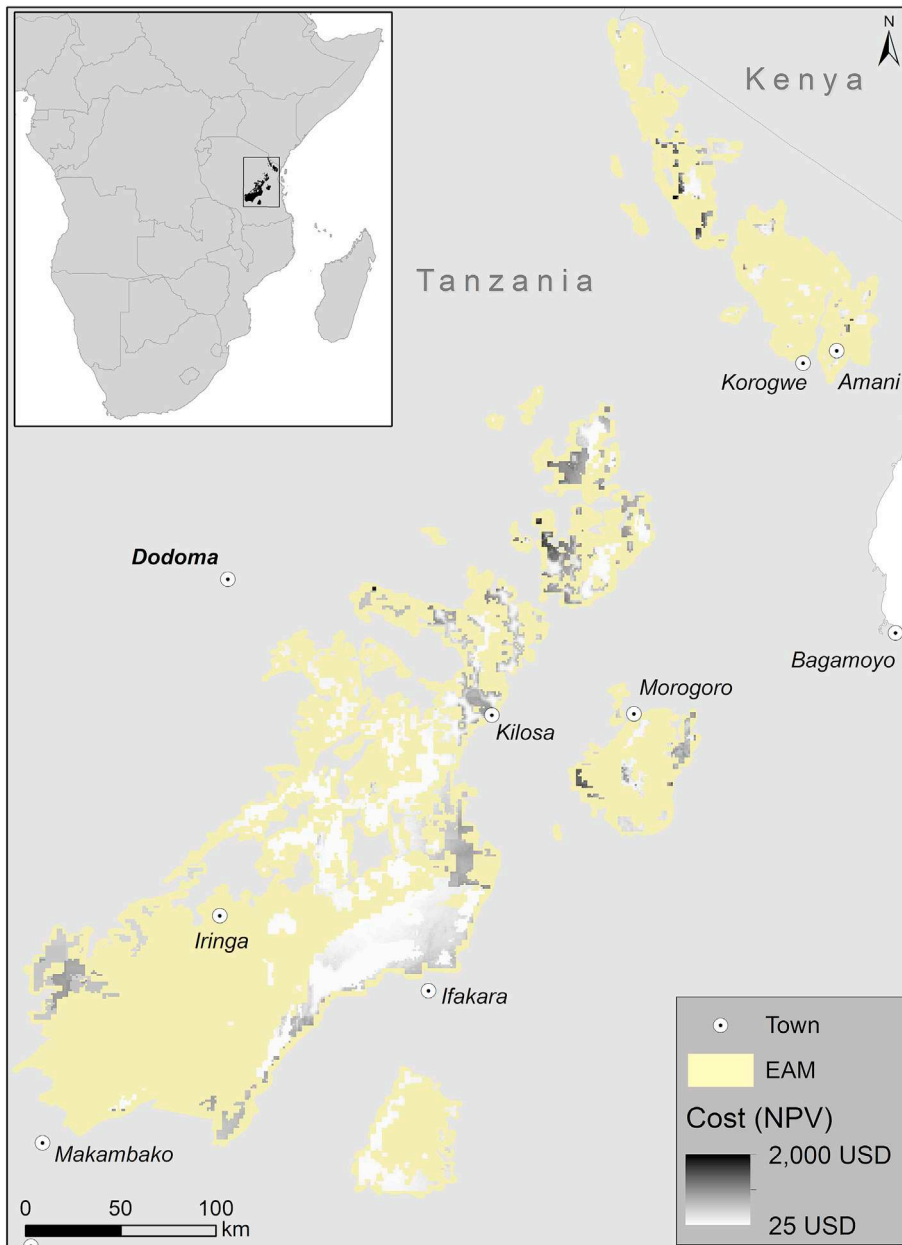


Fig. 1. Spatial variation in the costs of conserving forest and woodland in the Eastern Arc Mountains, summed for management, damage and opportunity costs (net present value, $r = 15\%$). Yellow area indicates habitat other than forest/woodland within the Eastern Arc Mountains. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

outside of reserves and provide summaries by vegetation type (woodland and forest). We calculated the level of association between these cost layers using a Spearman-rank correlation test. We also provide results based on the existing reserve network, which we estimated using the native resolution of the analyses (Table 1). We used R 2.15.1 (R Development Core Team., 2009) and ArcGIS 10 (ESRI, 2010) for analyses. Monetised values are for 2010 USD, using an exchange rate of 1 USD = 1,450TZS.

3. Results

Across the EAM, opportunity cost varies substantially, with estimates for a medium discount rate ($r = 15\%$) ranging from 0 to 1336 USD/ha (mean = 191 ± 234 SD; Fig. 1; Table 2). Opportunity costs are greater near mountain bloc edges, where population densities and accessibility are higher, resulting in a greater probability of conversion. Although the value of

Table 2

Comparing mean (\pm SD) net present cost/ha across reserved and non-reserved forest and woodland in the Eastern Arc Mountains. Results are presented for high and low discount rates ($r = 20\%$ and 5% , to give low and high estimates, respectively), but we use the medium estimate (in bold; $r = 15\%$) for all further results and comparisons. As well as accounting for low and high discount rates, the net present costs of wildlife damage are further adjusted for low and high rates of yield loss (4% and 10% , respectively). The net present cost of forest and woodland in the current reserve network ($553,500$ ha) is given in the final column.

Cost Type	Estimate	Forest		Woodland		Forest and Woodland			Reserve Network
		Reserves	Non-Reserves	Reserves	Non-Reserves	Min.	Max.	Mean	Total
Opportunity Cost	Low ($r = 20\%$)	34 ± 56	36 ± 41	106 ± 104	181 ± 190	0	940	134 ± 167	\$37,643,030
	Medium ($r = 15\%$)	49 ± 81	54 ± 61	160 ± 154	256 ± 265	0	1336	191 ± 234	\$56,331,583
	High ($r = 5\%$)	181 ± 300	202 ± 234	635 ± 589	867 ± 949	0	4566	668 ± 831	\$219,723,355
Damage Costs	Low ($yl = 4\%$; $r = 20\%$)	6 ± 8	8 ± 9	2 ± 4	4 ± 6	0	45	4 ± 7	\$2,087,302
	Medium ($yl = 7\%$; $r = 15\%$)	12 ± 17	18 ± 20	4 ± 10	9 ± 14	0	98	9 ± 15	\$4,584,191
	High ($yl = 10\%$; $r = 5\%$)	35 ± 50	52 ± 59	12 ± 27	27 ± 41	0	283	27 ± 43	\$13,243,022
Management Costs	Low ($r = 20\%$)	40 ± 5	40 ± 5	39 ± 3	39 ± 3	25	57	39 ± 3	\$21,742,969
	Medium ($r = 15\%$)	50 ± 6	50 ± 6	49 ± 4	49 ± 3	31	72	49 ± 4	\$27,287,160
	High ($r = 5\%$)	101 ± 13	101 ± 12	99 ± 8	99 ± 7	63	145	99 ± 9	\$55,179,884

yl = yield loss due to damage by wildlife.

charcoal stocks and agricultural yields are greater in forest (Fig. 2a), forest is converted at far lower rates than woodland, resulting in expected opportunity costs for forest that are substantially smaller than those for woodland (Fig. 2b and c).

The mean wildlife damage cost of forest and woodland is far lower than our estimated opportunity costs, at just 9.3 USD/ha, in part because of the many areas of forest and woodland that do not border cropland (Table 2). However, this varies widely and is highest in the centre and northeast of the study area, where high value cropland borders forest and woodland habitat (Fig. 1; Fig. S1).

Management and opportunity costs are greater near more populated and accessible areas, while damage costs are associated with cultivated areas (Fig. 1; Fig. S1; Green et al., 2012; Green et al., 2013). Spearman-rank coefficients measuring spatial correlation between them, however, are low at these scales of analysis (forest: 0.20–0.36; woodland: 0.13–0.41; Tables S1 and S2). Nevertheless, we estimate that in 2002, just 0.6% of Tanzania's population of 36 million lived within 200 m of the EAM reserve network and just 2.5% within 2 km (Table S3).

Across the entire study area, opportunity and management costs are roughly equal for forest cells, accounting for 44% each of the total cost, whilst damage costs account for 12% (Fig. 3). The average woodland cell has much higher total costs that are dominated by opportunity costs at 80%, followed by management costs (17%) with damage costs accounting for just 3% (Fig. 3). Within just the current reserve network, opportunity costs account for around two thirds of the total, management costs slightly less than one third and damage costs just 5% (Fig. 3).

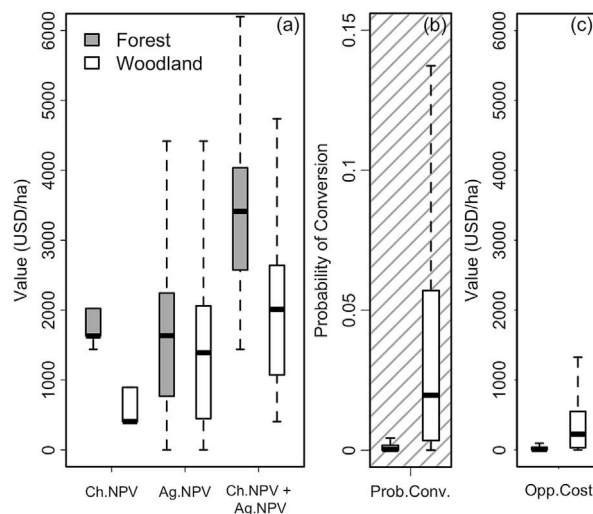


Fig. 2. Calculating opportunity cost of conservation for the forest (grey boxes) and woodland (white boxes). Black bars show median; boxes show interquartile range; whiskers indicate points within 1.5 times the interquartile range above and below the upper and lower quartiles. (a) The first step in calculating opportunity costs of conservation is to estimate the net present value ($r = 15\%$; time horizon = 25y) of converting the existing biomass to charcoal (Ch.NPV) and then farming (Ag.NPV) on the converted land to derive an estimate of the private benefits (Ch.NPV + Ag.NPV) that could be earned by converting the land now (plotted values assume conversion in year 1). However, this does not account for the expected rates of habitat conversion (Prob.Conv.) in the absence of protection (b), which are much lower for forest. (c) Once the modelled probability of conversion is accounted for the expected value (or opportunity cost) is substantially lower.

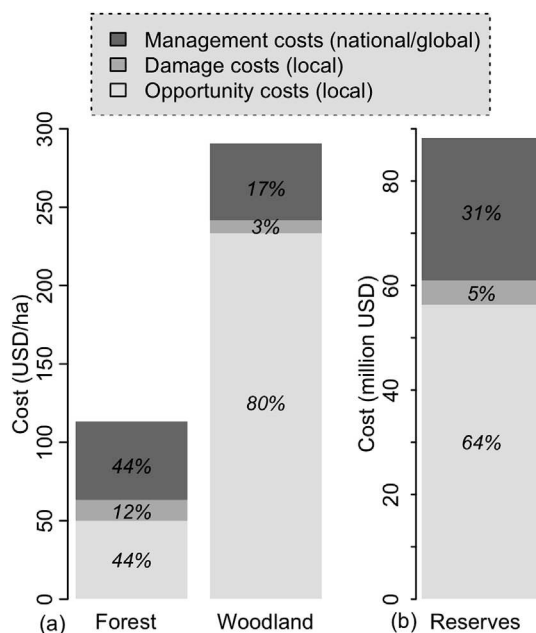


Fig. 3. Breakdown of the different costs of conservation. (a) In forest pixels, the mean present cost of management is approximately equal to the opportunity cost; between them, they account for almost 90% of the costs considered here. In woodland pixels, opportunities costs dominate, with a relatively small proportion arising from damage to crops by wildlife. (b) The greatest cost of the current reserve network (553,500 ha) is opportunity costs, followed by management costs and then damage costs.

4. Discussion

Conservation costs accrue disproportionately to poorer nations and communities (Balmford and Whitten, 2003; Wells, 1992). Our analyses allow us to investigate this within a tropical biodiversity hotspot. Even accounting for the relatively low rate of forest conversion, we find that opportunity and damage costs, largely borne by those living within the vicinity of the reserves, are greater than reserve management costs, which are borne at national and international scales, paid for via taxes, and revenues from wildlife tourism and hunting licenses (Kideghesho, 2008; Green, 2012).

The incidence of poverty in districts in and around the Eastern Arc Mountains is high; over thirty percent of people live below the national poverty line, and access to improved water, electricity and decent shelter is low on average (Fisher, 2012). Even within the EAM, however, poverty in more rural districts is higher (Fisher, 2012). As in much of the world, conservation areas in Tanzania tend to be located away from urban areas, so that rural households bear the opportunity costs of conservation and wildlife damage costs. Poverty in Tanzania has been described as “mainly a rural phenomenon” (Schaafsma, 2012) and the incidence of poverty in rural areas is substantially higher than in urban areas, whether measured as the ability of households to meet their basic needs or as access to infrastructure and education (IMF, 2010; JBC, 2006; Schaafsma, 2012).

Taken together with our findings, this highlights the disproportionate burden that conservation has on the rural poor. This corroborates earlier work that shows the poorest suffer the greatest negative impact from reserves (Ferraro, 2002; Franks, 2008; Ghatte, 2003; Kideghesho, 2008; Kideghesho and Mtoni, 2008). As well as the total amount being greater, the nature of local costs means that they accrue to fewer people.

The economic costs of damage by wildlife are an order of magnitude smaller than opportunity or management costs. However, the loss of harvest associated with wildlife damage can be a significant burden on those struggling to subsist nutritionally or economically (Mc Guinness and Taylor, 2014). We caution against dismissing them as trivial for four reasons. First, although average costs are low, potential losses from a rare but devastating wildlife damage event warrants the attention of conservation policy makers. Second, if perceived costs are high then, even if actual costs are low, critical local support for conservation may be missing (Kideghesho, 2008; Nyirenda et al., 2013; Pechacek et al., 2013). Third, although comparatively small, damage costs accrue very locally – to the immediate neighbours of reserves. This land, most subject to wildlife damage, may be the least desirable and, consequently, occupied by the poorest or most marginalised farmers. Fourth, we have not considered the opportunity costs of time spent guarding crops against damage, which are likely to exceed the direct costs (Kideghesho, 2008). On average, farmers in our study spent 25 days per year (median), which equates to 43 USD per year at the unskilled labour wage. We expect the indirect costs of guarding, however, to outweigh such economic considerations considerably (Nyirenda et al. 2013). The indirect costs can include loss of sleep, loss of time to do other chores, and, for children in farming households who are required to guard fields rather than attend school, loss of educational opportunities, potentially reducing their future welfare (Kideghesho, 2008; Mackenzie and Ahabyona, 2012; Nyirenda et al. 2013).

The hidden costs of wildlife damage extend even further to, for example, elements of psychosocial wellbeing and the potential economic or nutritional impact of switching to less susceptible crops (Barua et al., 2013; Mc Guinness and Taylor, 2014).

Although our estimates of opportunity cost are dominated by topographical and accessibility parameters, farming decisions are in reality conditioned by a range of factors including property rights, subsistence needs, cultural values, market price volatility and yield uncertainty (Parks, 1995). We constrain our analysis to the opportunity costs of smallholder farmers, which is justified in this context where small-scale conversion to agriculture dominates habitat conversion (Fisher et al., 2011). By confining our study to a single stakeholder group and ignoring costs associated with crop guarding, however, our estimates of both types of local cost are conservative (Adams et al., 2010; Nyirenda et al. 2013). Moreover, a wide body of literature describes costs and risks to livelihoods that are associated with involuntary loss of (access to) land. These are multi-dimensional, mediated through landlessness, joblessness, homelessness, marginalization, food insecurity, increased morbidity, loss of access to common property resources, community disarticulation, and loss of educational opportunities (Cernea 2000, 2005). By using net profits from land as a proxy for local costs, we do not include these factors, which would need further consideration during planning. However, whilst we acknowledge the limits to economic information, it forms a vital part of understanding the impacts of conservation, alongside other sociological data and considerations (Cernea, 1999). Moreover, their ready incorporation into spatial conservation planning tools mean that spatially explicit data on locally-borne costs provide an important step forward for integrating and mainstreaming local considerations into early stages of conservation planning processes (Ban et al., 2009; Pechacek et al., 2013), which has traditionally incorporated acquisition and management costs (Armsworth, 2014). It is crucial to understand differences between cost types and to preserve these through the planning process to account for unequal impacts on different stakeholder groups (Armsworth, 2014; Mazor et al., 2014).

The reported costs, and underfunding, of biodiversity conservation has focused largely on management costs (e.g. McCarthy et al. 2012); locally borne costs have received less attention. Protected areas are a vital component of conservation efforts, yet unless those living closest to a resource receive benefits that are equal to or exceed the cost that they bear under conservation, the system is likely to be less effective from either conservation or development perspectives (Ferraro, 2002; Sheil et al., 2013; Smith et al., 2009). Where livelihoods depend upon opportunities forgone or diminished by conservation intervention, outcomes are limited. Activities can be driven elsewhere, so displacing threats, rather than removing them (leakage; Robinson and Lokina, 2011); the rules can be broken, reducing the effectiveness of the conservation strategy; or the conservation intervention forces a shift in livelihood profiles, potentially to the detriment of local peoples' welfare. All of these raise concerns for both people and wildlife and timely consideration of the magnitude and distribution of local costs in the conservation planning process is, therefore, vital.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <https://doi.org/10.1016/j.gecco.2018.e00385>.

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