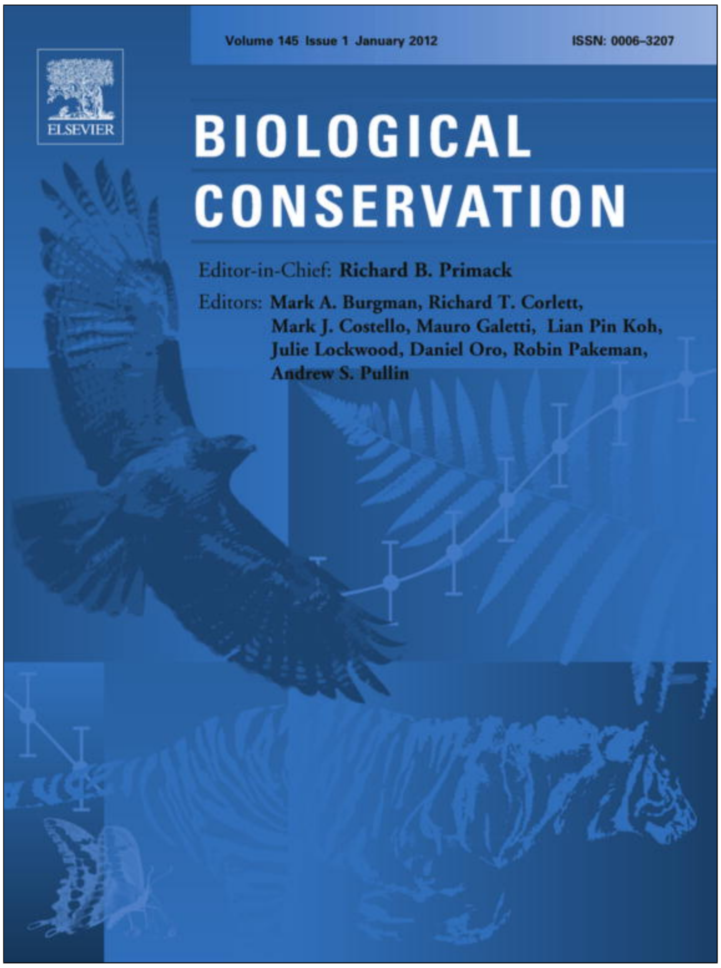
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Estimating management costs of protected areas: A novel approach from the Eastern Arc Mountains, Tanzania

Jonathan M.H. Green [a,*], Neil D. Burgess [b,c], Rhys E. Green [a,d], Seif S. Madoffe [e], Pantaleo K.T. Munishi [e], Evarist Nashanda [f], R. Kerry Turner [g], Andrew Balmford [a]

[a] Conservation Science Group, Department of Zoology, University of Cambridge, Downing Street, Cambridge, CB2 3EJ, United Kingdom
[b] Centre for Macroecology, Evolution and Climate, Department of Biology, University of Copenhagen, Universitetsparken 12, DK-2100, Copenhagen, Denmark
[c] WWF-US, Conservation Science Programme, 1250 24th Street, Washington DC, USA
[d] Royal Society for the Protection of Birds, The Lodge, Sandy, Bedfordshire SG19 2DL, United Kingdom
[e] Faculty of Forestry and Nature Conservation, Department of Forest Biology, Sokoine University of Agriculture, Chuo Kikuu, Morogoro, Tanzania
[f] Forestry and Beekeeping Division, Ministry of Natural Resources and Tourism, P.O. Box 426, Dar es Salaam, Tanzania
[g] Centre for Social and Economic Research on the Global Environment, University of East Anglia, Norwich, NR4 7TJ, United Kingdom

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Despite chronic underfunding for conservation and the recognition that funds must be invested wisely, few studies have analysed the direct costs of managing protected areas at the spatial scales needed to inform local site management. Using a questionnaire survey we collected data from protected area managers in the Eastern Arc Mountains (EAMs) of Tanzania to establish how much is currently spent on reserve management and how much is required to meet conservation objectives. We use an information theoretic approach to model spatial variation in these costs using a range of plausible, spatially explicit predictor variables, including a novel measure of anthropogenic pressure that measures the human pressure that accrues to any point in the landscape by taking into account all people in the landscape, inversely weighted by their distance to that point.

Our models explain over 75% of variation in actual spend and over 40% of variation in necessary spend. Population pressure is a variable that has not been used to model protected area management costs before, yet proved to be considerably better at predicting both actual and necessary spend than other measures of anthropogenic pressure.

We use our results to estimate necessary spend at a 9 km^2 resolution across the EAM and highlight those areas where the management costs of effective management are predicted to be high. This information can be used by conservation planners in the region and can be estimated for future scenarios of population growth and migration.

1. Introduction

Assessing the various costs of conserving protected areas (PAs) is important for three reasons. Firstly, it provides information to managers, non-governmental organisations and others to enable them to make the best use of current resources. Because the full costs of conservation are rarely met by existing budgets, such work can also provide part of the justification, alongside the benefits of conservation, for additional resources (Turner et al., 2003).

Secondly, systematic conservation planning (SCP), which attempts to optimise the allocation of scarce resources to achieve specific objectives (Polasky et al., 2001), requires information on spatial variation in the costs of conservation (e.g. Frazee et al., 2003; Polasky, 2008; Smith et al., 2008). In practice, these often vary more widely than biodiversity values, so improvements in quality and spatial representation of cost data typically lead to greater gains in SCP efficiency than would similar efforts to improve species distribution data (Balmford et al., 2000, 2003; Grantham et al., 2008; Moore et al., 2004; Naidoo et al., 2006; Naidoo and Iwamura, 2007; Polasky, 2008). Nevertheless a paucity of information on conservation costs, particularly in developing countries means that many SCP studies use the area of a reserve as a proxy for its cost, which makes the assumption that cost is predicted by PA size, rather than by its geographical or socio-economic attributes (see Balmford et al., 2003; Carwardine et al., 2008; Naidoo et al., 2006; Naidoo and Ricketts, 2006). However, management cost often does not scale in direct proportion to size (e.g. Bruner et al., 2004; Frazee et al., 2003), so instead explicit consideration needs to be given to investigating both the relationship

* Corresponding author. Tel.: +44 7952943820.
E-mail address: jmhg2@cam.ac.uk (J.M.H. Green).

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between reserve size and management costs per unit area and to other potential cost predictors (such as anthropogenic pressure).

A third reason for quantifying and mapping conservation costs is that it clarifies where costs are borne and by whom (Adams et al., 2010; Balmford and Whitten, 2003; Moore et al., 2004). Given that the various components of cost (e.g. management costs, opportunity costs and costs of damage by wildlife) usually accrue differently to different stakeholders, this is key to identifying equitable ways of spreading the burden of conservation (Knight et al., 2006; Linnell et al., 2010).

We focus here on the costs of PA management – i.e. those that are directly incurred in maintaining a system of PAs (Bruner et al., 2004; Dixon and Sherman, 1990; James et al., 1999a,b; Morrison and Boyce, 2009). Management costs are crucial to consider because PAs are the cornerstone of much effective conservation (Jenkins and Joppa, 2009; Joppa et al., 2008; Joppa and Pfaff, 2011; Leroux et al., 2010) and because PA effectiveness may be predicted by spending on management (Brooks et al., 2004; Leader-Williams, 1990). The potentially sizeable opportunity and damage costs of EAM conservation are examined in a separate analysis (Green et al., in preparation).

PA managers in our study region commonly complain of insufficient funding to manage their reserves effectively, so we explore variation both in current spending on management and in estimated necessary spend. Using data reported by PA managers in the EAM, we model these in relation to widely available mapped socioeconomic and geographic variables. By then applying these models of current and necessary management spend across the study region, we are able to address questions of funding shortfalls under the current system (i.e. the cost of making the current system effective), whilst also generating key information for examining how the system might be expanded beyond currently protected areas most efficiently and effectively (i.e. the cost of expansion of the reserve network). We are able to compare our findings on management costs with data on pole and timber cutting in PAs (Madoffe and Munishi, 2010) to investigate whether increased spending is associated with improved management effectiveness.

Global and international models of PA costs have been constructed in previous studies (e.g. Balmford et al., 2003, 2004; Bruner et al., 2004; Moore et al., 2004); however, these are largely based on national-level variables and are unlikely to perform as well if applied at sub-national scales. At sub-national scales, estimates of actual or necessary management costs are either not modelled in a spatially explicit manner, so cannot be estimated beyond the current reserve system in question (e.g. Blom, 2004; Culverwell, 1997; Howard, 1995) or they were developed for very specific habitat types and require explanatory variables that do not exist in the EAM (Frazee et al., 2003). In addition, none have looked at spatially explicit variation in both actual and necessary spend.

2. Materials and methods

2.1. Study area

The EAM are a chain of mountains stretching from the Taita hills in the south of Kenya through eastern Tanzania to the Udzungwa mountains in south-central Tanzania (Burgess et al., 2007; Platts et al., 2011). The forests on these mountains are noted for their exceptionally high biodiversity and form part of the Eastern Afromontane biodiversity hotspot (Burgess et al., 2004, 2007). They represent remnants of a once vast forest ecosystem that was contiguous with the forests of Central Africa (Lovett, 1985). These remnants have persisted due to the high orographic rainfall that these mountains receive from moist winds that arrive from the Indian Ocean and rise up the slopes of the EAM, depositing their moisture on the mountains' eastern flanks. Since the tertiary period, as Africa gradually dried, the surrounding low-lying areas became savannah leaving the EAM as a refuge for many species (Conrad et al., 2011). They are also extremely important to human wellbeing through the ecosystem services that they provide (Burgess et al., 2007).

PAs in the Eastern Arc fall under the control of three agencies: Tanzania National Parks (TANAPA) manage all National Parks (NPs); Nature Reserves (NRs) and National Forest Reserves (NFRs; also called catchment forest reserves) are managed by central government, under the Forestry and Beekeeping Division; last, local governments manage Local Authority Forest Reserves (LAFRs) and village governments manage Village Land Forest Reserves (VFRs; in conjunction with district authorities). Due to the lack of georeferenced boundaries and financial data on VFRs, these were excluded from the present analyses. Median reserve size (LAFRs, NFRs, NRs and NPs) within the current system is 8.8 km^2 (Fig. 1a).

2.2. Cost data

During April–June 2010 we conducted 40 interviews with district forest officers, district catchment managers and nature reserve conservators across the 22 districts of the Eastern Arc. The interviews were structured around previous studies of PA funding (Burgess and Kilahama, 2004; Craigie, 2010; James et al., 1999b). We gathered information on the money spent on management of PAs in the financial year 2008/2009 (the financial year runs from July to June) from these managers, who were responsible for administering, or assisting with administering, 482 PAs out of an estimated 500 within the EAM districts (including VFRs). The management structure and funding pathways for reserves in the EAM rarely operate simply. Funds can come from local and/or central government and may be divided between several reserves, which makes collation of financial information and its attribution to a reserve (or reserves) much more challenging (McCrea-Strub et al., 2010). Therefore, surveying had to be as comprehensive as possible, interviewing all government forestry departments in each district to cover all significant funding routes for the reserves.

To investigate existing funding provision, managers were asked about the amount currently spent (hereafter "actual spend") on PA management in the EAM. An earlier study (Madoffe and Munishi, 2010) found that PA management effectiveness in the EAM varied, with just one of 15 state-owned reserves classed as having "good" management effectiveness. Because this performance might be due to inadequate current spending on management, we also asked managers to estimate the amount necessary to enable them to meet their conservation objectives. Such data on "necessary spend" is crucial, both for future conservation planning in the region, as well as for planning how best to use existing resources and identifying funding shortfalls.

Wherever possible, we obtained the budgets of individual reserves but, in most cases, the manager could only provide a spending estimate for an aggregate of reserves (e.g. all LAFRs in a district). In these cases, we used these aggregates as the units of analysis (hereafter referred to as "reserve groups"; Fig. 1b). This lumping could hide or dilute the effects of explanatory variables – particularly PA size, which has been shown to have a negative relationship with spend per unit area (Balmford et al., 2003; Bruner et al., 2004; Frazee et al., 2003; McCrea-Strub et al., 2010). However, joint management of reserves in groups like this is the reality for many PA managers both in Tanzania and elsewhere, so an analysis of such units is highly relevant.

All analyses were conducted in Tanzania Shillings (TZSs) but we express figures in United States Dollars per hectare per year (USD ha^{-1} y^{-1}), using an exchange rate of 1450 TZS = 1 USD and a national deflator index (Index Mundi, 2010).

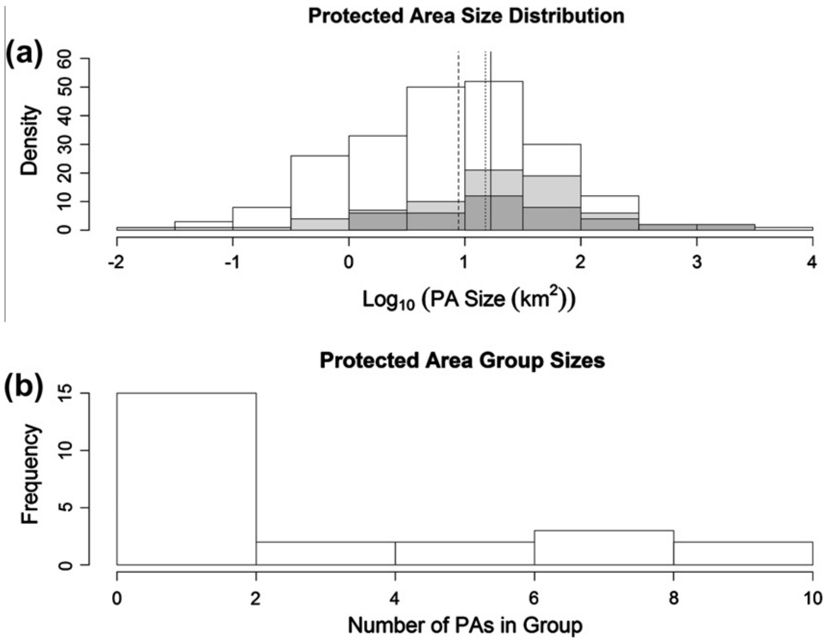

Fig. 1. (a) Frequency distribution of $\log_{10}$(protected area size). White bars and dashed line illustrate the size frequency distribution and median size (8.8 km^2), respectively, of all reserves in the Eastern Arc Mountains ($n = 220$). Light grey bars and dotted line show the size frequency distribution and median size (15 km^2) for reserves used in our analyses of actual spend ($n = 74$). Dark grey bars and solid line shows the size frequency distribution and median size (17 km^2) for reserves used in our analyses of necessary spend ($n = 40$). (b) Frequency distribution of number of protected areas in reserve groups (median = 1; $n = 24$). In extrapolating our models across the study area, we standardised number of protected areas in reserve group to be equal to one and total reserve area to be 9 km^2.

2.3. Cost types

Management of PAs is a complex process involving many kinds of outlay, with a common division being between recurrent expenditure (often calculated per annum) and capital expenditure, which, in Tanzania, is often only available when there is externally funded project support. However, even this dichotomy is not always easily defined, so we have drawn on previous studies (Bruner et al., 2004; Frazee et al., 2003) and our own experience to develop a classification of management spending (Table 1). For all analyses, we modelled recurrent plus capital expenditure (but not PA establishment costs) per hectare per annum. Models were also built to estimate recurrent costs only, but we do not present these results, as they did not improve model fit and they underestimate spend because capital costs are a significant proportion of both actual and necessary management spending.

2.4. Management effectiveness

The most reliable management effectiveness data for different reserve types are from Madoffe and Munishi (2010), who quantified numbers of poles and trees cut per hectare (measured as stumps encountered during field-based surveys) for LAFRs, NFRs and NRs (data unavailable for NPs). We plot these values against our observed funding shortfalls for these reserve types in order to investigate whether we can expect PA effectiveness to increase under improved funding.

2.5. Cross validation and missing data

To corroborate information received from questionnaires and to fill gaps where data were missing, we used supplementary information provided in annual reports and budgets from various agencies (EAMCEF, 2008, 2009, 2010; FBD, 2008; TANAPA, 2001). We also interviewed major donors and regional forest managers to cross-validate information received from district-level managers and conservators. Where data were unavailable for 2008/9 ($n = 1$), we used the previous year's figures (2007/2008) and adjusted for inflation to 2008/2009 (Index Mundi, 2010).

In most cases, managers were uncomfortable estimating staff salaries. Therefore, for those management groups where we had sufficient data ($n = 11$) we regressed total salary expenditure

Table 1
A classification of management costs. In the Eastern Arc Mountains, each of these costs may be funded from local government, national government or donor agencies.

Cost type		Description	Examples
Recurrent expenditure	Salaries	Predictable and regular costs of employing staff	Salaries for permanent staff
	Operating costs	Other predictable and regular costs of running the reserve as it is	Forest monitoring, forest protection, equipment repairs, fuel, casual labour, research and staff training
Capital expenditure		Cost of upgrading/purchasing equipment or facilities Typically for larger amounts, and often irregular	Investing in buildings, facilities or equipment for staff or local communities
Establishment costs		Costs involved in setting up a new reserve (or transiting from one status to another)	Costs of stakeholder meetings, legal costs of gazettement, costs of boundary marking, costs of preparing management plan and capital costs during reserve establishment phase

against staff number ($\log_{10}$(salaries) = 6.776 + 0.634 $*$ $\log_{10}$(staff number)); $n = 11$; $r^2_{adj} = 0.88$; $p < 0.001$). The reason for using this equation, rather than some average measure of wage is because even the smallest departments had a district manager, but as staff number increased, so the number of staff in lower levels of the hierarchy (and receiving lower pay) increased. We used this equation to estimate total salary expenditure for reserves where staff number was known but data on salaries were unavailable.

2.6. Data analysis

2.6.1. GIS data

Spatially explicit modelling and analyses required us to extract predictor variables using the reserve boundary shapefiles, so could only be conducted on reserve groups for which GIS data were available (Table 2). Of 482 reserves for which we had data, 146 were listed in the World Database on Protected Areas (WDPA; IUCN, 2010) and had Geographical Information System (GIS) data associated with them. For actual spend we had 50 reserve groups, of which 23 had complete GIS data associated with them. For necessary spend, the data were aggregated further to 29 reserve groups, for only 13 of which could we acquire GIS information. All GIS processing was conducted in ArcGIS 9.3 (Environmental Systems Research Institute, 2009).

2.6.2. Variables

The response variables (actual spend ha^{-1} y^{-1} and necessary spend ha^{-1} y^{-1}) were transformed for analysis using Box–Cox transformation to give approximately normally distributed residuals (actual spend: Box–Cox parameter $\lambda = 0.25$; necessary spend: Box–Cox parameter $\lambda = 0$, which is equivalent to the natural log of necessary spend).

Spend can be expected to be influenced by reserve attributes, socio-economic factors and environmental variables (Table 2). The reserve characteristics examined were PA type, number of PAs in the reserve group and total combined area of the reserve group because management systems (and therefore spend) vary between reserve types and because larger groups (in number or size) may be able to utilise equipment, such as vehicles, more efficiently. To measure accessibility of reserves, which is hypothesised to positively correlate with management cost due to the necessity for mitigation of increased human impact (Bruner et al., 2004; Frazee et al., 2003; Nelson and Chomitz, 2009), we used mean terrain ruggedness using a Vector Ruggedness Measure (VRM; see methods in Sappington et al., 2007) and median population density within the PA (see supplementary material in Platts et al., 2011).

We also hypothesised that pressure exerted from outside the boundaries of the PA could have an effect on the amount of funding that is actually spent and/or necessary. We looked at three ways to measure this pressure: the percentage of human-dominated land cover within a 5 km buffer of the reserves, mean population density around the reserves (within a series of buffers at 5 km, 10 km, 15 km, 20 km, 25 km, 30 km and 40 km) and population pressure around the reserves. This final measure was included because treating the whole of the human population within a buffer area as exerting a uniform effect on conservation costs seemed unrealistic so we developed a measure of "population pressure" based on Platts (2011).

The population pressure measure we used assumes that populations impact neighbouring areas to an extent that depends on their distance from them (Walsh et al., 2003). Therefore, population pressure for point i should take into account the population at i and also the remote populations, j, in the landscape around it. The pressure of remote populations (in people equivalents, p.e.) should be inversely weighted by distance, so that more distant populations exert less pressure than those that are nearer (Walsh et al., 2001). In order to make the calculation of population pressure computationally tractable, we decreased the resolution of the population density layer from 1 km^2 to 25 km^2. We proposed that the distance decay function of the weight applied to population should follow a half-normal distribution, as we expect nearby populations to exhibit highest pressure, which decreases rapidly once the distance to the PA is beyond walking distance. Thus, population pressure in cell i is given by:

$$pressure_i = \sum_{j=1}^{n} p_j \times \exp(-(d_{ij}/\sigma)^2)$$

where p_j is the population at remote cell j, d_{ij} is the Euclidean distance between focal cell i and remote cell j, n is the number of cells within 200 km of the focal cell and σ is a parameter that determines the shape of the distance decay function (Fig. 2). Summation is over all n cells in the vicinity of cell i, with n being chosen so that the contribution to pressure of the most distant cells from i was vanishingly small. We created a range of population pressure layers, each with a different σ value, for the entire EAM landscape. We then used these to build a series of simple linear regression models of actual and necessary management spend, from which we chose the population pressure layer which gave the best model fit (lowest AIC$_c$). We then ran the same process to select the buffer size at which our population density layer gave the best model fit, so that for both population pressure and population density we used just one layer each in our subsequent model construction.

Table 2
Predictor variables used to construct models of management expenditure per year. Variables were taken from the questionnaire survey or extracted using GIS tools.

Variable name	Source	Description
Reserve type	Questionnaire survey	Category of reserve: Local Authority Forest Reserve (LAFR), National Forest Reserve (NFR), Nature Reserve (NR) or National Park (NP)
Number of PAs	Questionnaire survey	Number of reserves in group
Total area (ha)	Questionnaire survey	Total area of reserve group
Terrain Ruggedness (VRM)	GIS variable[a]	Mean Vector Ruggedness Measure (VRM) or terrain ruggedness[b] of reserve group
Human use (%)	GIS variable[a]	Percentage of land within a five km buffer of reserve under human dominated land use type (cultivation, urban and disturbed habitats)
Median population within PA	GIS variable[a]	The median population within the reserve group
Population density (people/km^2)	GIS variable[a]	Mean population density within 5, 10, 15, 20, 25, 30 and 40 km buffer of reserve (number of people per km^2)
Population pressure (p.e./km^2)	GIS variable[a]	Mean population pressure of all cells within PA boundary (in person equivalents per km^2)

[a] These GIS layers were generated as part of the Valuing the Arc project (Burgess et al., 2009).
[b] See Sappington et al. (2007) for methods and description of this variable.

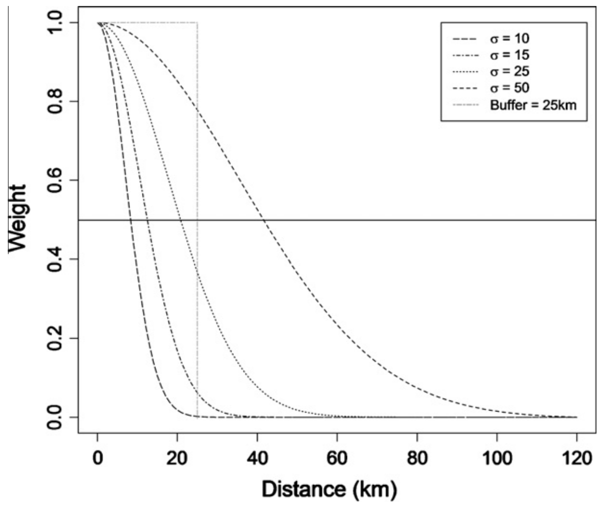

Fig. 2. Population pressure is hypothesised to impact a particular point in space according to some distance-weighted function. We used a half-normal curve, as we expect nearby populations to exhibit highest pressure, which decreases rapidly once the distance to the PA is beyond walking distance. Modifying the σ value changes the shape of the curve. Higher σ values give greater weight to relatively distant populations, while smaller σ values capture only the pressure of more proximate populations. The point at which the line crosses the horizontal solid black line indicates the distance at which a population's impact is reduced by half: for a σ value of 50, the impact decreases by 50% at around 45 km, whereas for a σ value of 10, the impact is reduced by 50% within around 8 km. The dashed grey line shows how the fixed buffer approach (for a buffer of 25 km) apportions population pressure to a reserve; all of the population within 25 km is hypothesised to exert an equal pressure.

Actual and necessary spend $ha^{-1}\,y^{-1}$ were then modelled in relation to population pressure or density and our other explanatory variables (Table 2; due to colinearity, we did not use both population density and population pressure in the same models, but analysed them separately). Using all combinations of the predictor variables (no interactions), each model was then ranked using AIC_c values.

2.6.3. Modelling

In building models, we adopted an information-theoretic approach, using AIC_c to measure goodness of fit (due to small sample sizes; Burnham and Anderson, 2002). We followed the methods of Grueber et al. (2011) to generate a set of models based on predictor variables selected because of a priori hypotheses or because they had previously been found to be associated with actual or necessary management costs. We tested all possible combinations of these variables and present those with a change in AIC_c of less than 4 ($AIC_c - AIC_{c,min} = \Delta_i < 4$) and, from these, we estimated an average model using the zero-method (Burnham and Anderson, 2002; Grueber et al., 2011). All statistical analyses were conducted in R (R Development Core Team, 2009).

3. Results

3.1. Management costs in the EAM

Across the EAM, 55% of actual spending on PA management (including reserve establishment costs) was on recurrent expenditure (20% salaries and 35% operating costs). Capital expenditure accounted for 21% of expenditure and was non-normally distributed across reserves, being present in only 20 reserve groups (out of 50 for which we obtained data), where it varied in magnitude from 1% to 510% of annual recurrent expenditure. The median total expenditure per unit area was 2.3 USD $ha^{-1}\,y^{-1}$ (mean = 6.1 USD $ha^{-1}\,y^{-1}$; IQR = 1–6 USD $ha^{-1}\,y^{-1}$; $n = 50$), while the median amount of money reported as being necessary for a PA to achieve all its objectives was 8.3 USD $ha^{-1}\,y^{-1}$ (mean = 19.7 USD $ha^{-1}\,y^{-1}$; IQR = 5–17 USD $ha^{-1}\,y^{-1}$; $n = 29$).

3.2. Spatially explicit model of actual spend

The population pressure layer for which we obtained the highest goodness of fit in simple linear models of actual spend was with a σ value of 25, in which population pressure declines by 50% over 20 km and down to zero over 60 km (Fig. 2). For population density, the best buffer size for modelling actual spend was 20 km.

The best set of models of actual spend that included population pressure as a predictor in the global model contained population pressure, reserve type, median population density within the reserves, VRM and number of PAs in the reserve group (Table 3a). These final models explained 69–78% of the variation in the response variable and an average model was derived from this subset of models with $\Delta_i < 4$:

$$((\text{Actual spend}^{0.25} - 1)/0.25) = b + 3.22 \times 10^{-5} * pp25 + 0.0213 * medpop - 0.0997 * no.PA - 34.9 * VRM$$

where b is the intercept, which is specific to each reserve type (LAFR: $b = -0.295$; NFR: $b = 1.52$; NR: $b = 3.77$; NP: $b = 4.6$); pp25 is population pressure calculated with a sigma value of 25 ($\sigma = 25$); no.PA is the number of PAs in the reserve group; medpop is the median population density within the reserve group; and VRM is the terrain ruggedness index. Using population pressure resulted in better model fit than using population density within a buffer (Table 3).

3.3. Spatially explicit model of necessary spend

The median proportion of necessary spend that is received, was just 0.31 (mean = 0.43; IQR = 0.16–0.42; $n = 29$), and this could be used with our model of actual spend to predict total necessary spend across the study area. However, this shortfall varies spatially, so that when we used actual spend to predict necessary spend in a general linear model, it was a poor predictor, accounting for only 4% of the variation. Therefore, the idea that variation in necessary spend can be estimated using multipliers and modelled actual spend is not supported by the data.

Instead, as with actual spend, we generated spatially explicit models of variation in necessary spend as a function of geographic and socio-economic variables. We first chose which population pressure and population density layers (each calculated using a different σ value or buffer size, respectively) to use. Once again, population pressure with a sigma value of 25 (Fig. 2) maximised goodness of fit, while the best fixed-buffer population density layer was 5 km.

Using population pressure ($\sigma = 25$; Table 4a) gave better models than when population density within a fixed buffer was used (Table 4b). Alongside population pressure, the best models of necessary spend contained number of PAs in reserve group, total area and VRM (Table 4a) and the average model for the subset with $\Delta_i < 4$ is calculated as:

$$\text{Ln(necessary spend)} = 9.24 + 5.6 \times 10^{-6} * pp25 - 6.91 \times 10^{-2} * \text{no.PA} - 2.67 \times 10^{-6} * \text{tot\_ha} - 4.61 * VRM$$

where tot_ha is the total area (in hectares) of the reserve group and other variables are as given above.

In using this model to make spatially explicit predictions of the spend needed per ha for PAs anywhere in the study region,

Table 3
a) Actual expenditure per hectare per year modelled with population pressure as an explanatory variable (though not forced in). b) Actual expenditure per hectare per year modelled with population density within a fixed buffer (rather than population pressure) as an explanatory variable (though not forced in). Note that the best model from the set that includes population density within a fixed buffer has a change in AIC_c value (Δ_i) of 1.4, when compared to the best model from the set that uses population pressure (Table 3a).

Intercept	Population pressure[a]	Type[c] NFR	NR	NP	Median population	Number of PAs	VRM	Log(L)	K	AIC_c	Δ_i	w_i	r^2_{adj}	n
(a)														
−1.110	3.23×10^{-5}	3.97	6.45	7.49	0.027			−43.68	7	108.8	0	0.39	0.72	23
−0.864	3.22×10^{-5}	4.92	6.46	7.50	0.028	−0.249		−41.89	8	110.1	1.2	0.21	0.75	23
0.114	3.27×10^{-5}	4.13	6.97	7.71	0.021		−66.8	−42.36	8	111.0	2.2	0.13	0.74	23
0.597	3.28×10^{-5}	5.23	7.06	7.75	0.021	−0.281	−78.0	−39.75	9	111.4	2.5	0.11	0.78	23
1.334	3.08×10^{-5}	4.47	7.21	7.46			−107.0	−45.07	7	111.6	2.8	0.1	0.69	23
1.827	3.08×10^{-5}	5.55	7.31	7.50		−0.274	−118.6	−43.16	8	112.6	3.8	0.06	0.72	23
(RVI)[b]	(1.00)		(1.00)		(0.84)	(0.38)	(0.40)							
Intercept	Population density[d]	Total area	Human use		Median population	Number of PAs	VRM	Log(L)	K	AIC_c	Δ_i	w_i	r^2_{adj}	n
(b)														
−0.4582	3.117					−0.383	−139.0	−48.34		110.2	0.00	0.51	0.43	23
−2.2980	3.060						−99.91	−51.70		113.6	3.42	0.09	0.53	23
−2.8920	2.681							-53.29		113.8	3.64	0.08	0.49	23
−1.7540	2.617−					−0.272		−51.81		113.8	3.64	0.08	0.53	23
−0.6796	3.154		5.92×10^{-3}			−0.384	−142.4	−48.30		113.9	3.65	0.08	0.61	23
−0.5342	3.130	1.18×10^{-6}				−0.386	−139.3	−48.33		113.9	3.70	0.08	0.61	23
−0.4237	3.129				-1.32×10^{-3}	−0.382	−142.1	−48.33		113.9	3.71	0.08	0.61	23
(RVI)[b]	(1.00)	(0.08)	(0.08)		(0.08)	(0.83)	(0.84)							

[a] Population pressure calculated using a sigma value of 25 ($\sigma = 25$).
[b] Relative Variable Importance (RVI).
[c] Coefficients for National Forest Reserve (NFR), Nature Reserve (NR) and National Park (NP) compared to Local Authority Forest reserve (LAFR).
[d] Mean population density within a 20 km buffer.

Table 4
a) Necessary expenditure per hectare per year modelled with population pressure as an explanatory variable (though not forced in). b) Necessary expenditure per hectare per year modelled with population density within a fixed buffer (rather than population pressure) as an explanatory variable (though not forced in). Note that the best model from the set that includes population density within a fixed buffer has a change in AIC_c value of greater than two ($\Delta_i = 2.3$), when compared to the model set that uses population pressure.

Intercept	Population pressure[a]	Number of PAs	Total area	VRM	Log(L)	K	AIC_c	Δ_i	w_i	r^2_{adj}	n
(a)											
8.315	9.038×10^{-6}				−15.87	3	40.4	0	0.4	0.40	13
8.930	7.329×10^{-6}	−0.119			−14.57	3	42.1	1.7	0.17	0.46	13
10.530		−0.169	-9.212×10^{-6}		−14.86	4	42.7	2.3	0.13	0.43	13
10.040			-1.028×10^{-5}		−17.46	3	43.6	3.2	0.08	0.23	13
10.150		−0.189			−17.46	3	43.6	3.2	0.08	0.23	13
8.838	9.434×10^{-6}			−23.99	−15.32	4	43.6	3.2	0.08	0.39	13
11.720		−0.198	-1.065×10^{-5}	−42.60	−12.76	5	44.1	3.7	0.06	0.54	13
(RVI)[b]	(0.65)	(0.44)	(0.27)	(0.14)							
Intercept	Population density[c]	Number of PAs	Total area	VRM	Log(L)	K	AIC_c	Δ_i	w_i	r^2_{adj}	n
(b)											
10.530		−0.169	-9.212×10^{-6}		14.86	4	42.7	0	0.2	0.43	13
8.562	0.00962				17.40	3	43.5	0.7	0.14	0.24	13
10.040			-1.028×10^{-5}		17.46	3	43.6	0.9	0.13	0.23	13
10.150		−0.189			17.46	3	43.6	0.9	0.13	0.23	13
11.720		−0.198	-1.065×10^{-5}	−42.6	12.76	5	44.1	1.4	0.10	0.54	13
9.537					19.72	2	44.6	1.9	0.08		13
(RVI)[b]	(0.18)	(0.56)	(0.56)	(0.13)							

[a] Population pressure calculated using a sigma value of 25 ($\sigma = 25$).
[b] Relative Variable Importance (RVI).
[c] Mean population density within a 5 km buffer.

we were careful to keep the predictions at similar scales to the analysis. Therefore, we mapped necessary spend $ha^{-1}\ y^{-1}$ at the median reserve size for our study area (9 km^2), having also verified that the size frequency distribution of reserves used in our analysis were representative of all reserves in the EAM districts (Fig. 1a). We then controlled for the effect of reserve group size by setting this parameter to be equal to one (i.e. only one reserve per reserve group) in our modelled surface (Fig. 1b). Once mapped, the effect of population across our study area becomes very clear; the most populous areas are the most costly to conserve effectively (Fig. 3).

Out of 23 reserves, 20 showed a funding shortfall (observed actual spend was less than modelled necessary spend). One NR received approximately the same amount as their modelled necessary spend, while one NR and one NFR received approximately 50% more than our model estimated was necessary.

3.4. Effectiveness

Finally, we plotted level of disturbance for LAFRs ($n = 3$), NFRs ($n = 11$) and NR ($n = 1$), measured as number of poles and trees cut per ha, against observed shortfall (Fig. 4; Madoffe and Munishi,

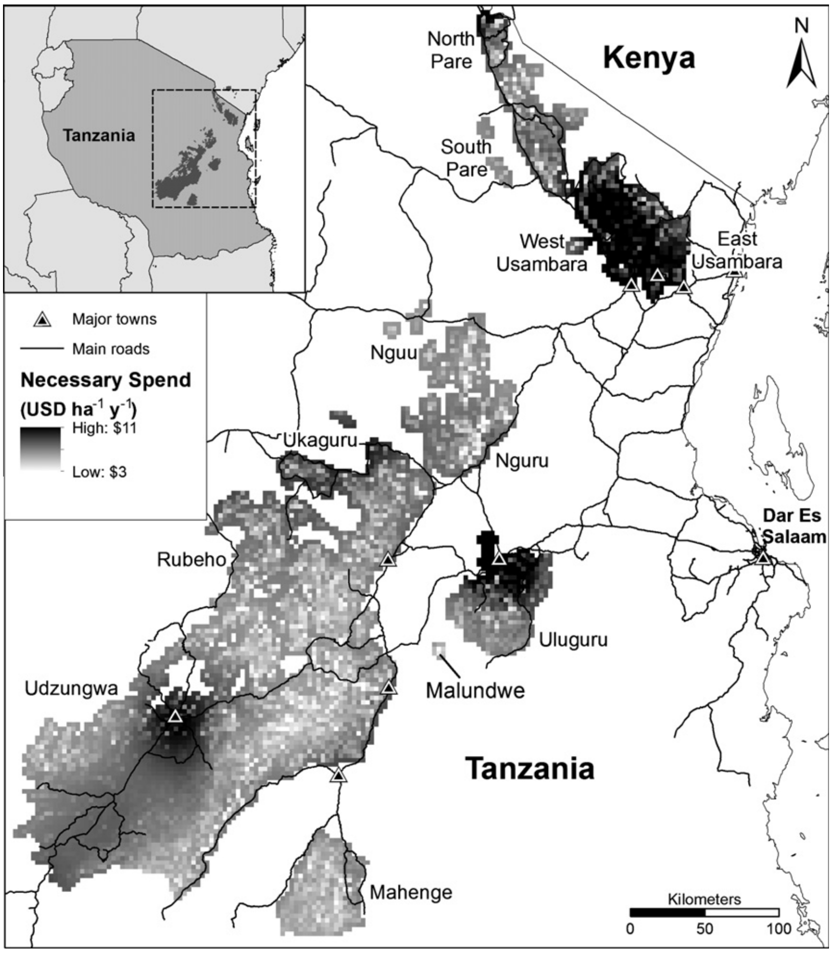

Fig. 3. Map showing spatial variation in modelled necessary spend per hectare per year for managing protected areas across the EAM. Spend per hectare varies from 3 to 11 USD y^{-1}. Major towns around the EAM (all with populations of over 20,000 people) are also marked.

2010). Disturbance appears to decrease with increased funding, with the marginal effect of spend decreasing as funding increases There are no comparable data for disturbance in NPs; however, Caro et al (1999) found mammal densities within Tanzanian NPs to be higher than those in other reserve types. A hypothetical fourth point is also shown for necessary spend and assumes full funding would largely eliminate disturbance. It seems likely that the marginal utility of spend per hectare will decrease as the actual amount spent approaches the necessary spend. This relationship highlights the importance of modelling both actual and necessary spend to identify where the shortfalls are greatest and to ensure that planners are able to estimate the true costs of effective extensions to the reserve network.

4. Discussion

The median actual spend across all sites in the EAM was 2.3 USD ha^{-1} y^{-1} (IQR = 1–6 USD ha^{-1} y^{-1}) and is around one third of the 7.7 USD ha^{-1} y^{-1} that was spent in Tanzanian NPs (TANAPA, 2009; data from 2007/2008, adjusted for inflation to 2009). However, Udzungwa Mountain NP, the only NP within the EAM, received a similar amount of funding as NRs. On the other hand, the median necessary spend reported by managers was 8.3 USD ha^{-1} y^{-1} (IQR: 5–17 USD ha^{-1} y^{-1}), which is slightly higher than the 7.7 USD currently spent in Tanzanian NPs. However, although these estimates of necessary spend may appear high, both their median and interquartile range are well within the

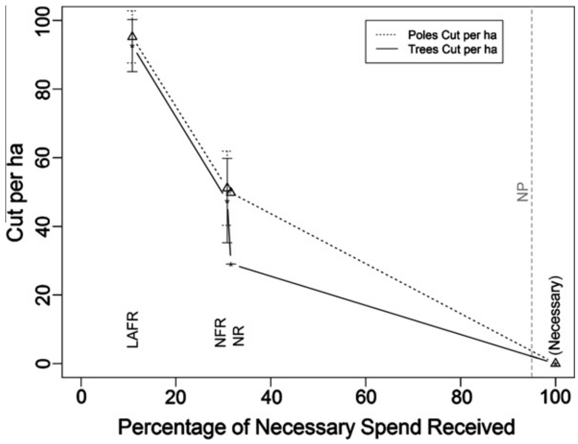

Fig. 4. Levels of disturbance from forest surveys of observed number of poles and trees cut per ha (±1 SE; Madoffe and Munishi, 2010) for Local Authority Forest Reserves (LAFRs; $n = 3$), National Forest Reserves (NFRs; $n = 11$) and Nature Reserves (NR; n = 1) plotted against observed actual median funding shortfalls (percentage of necessary spend that is received) from our survey data for the same reserve types (LAFRs: $n = 4$; median = 10%, mean = 19%, IQR = 9–20%; NFRs: $n = 6$; median = 31%, mean = 33%, IQR = 26–40%; NRs: $n = 7$; median = 32%, mean = 73%, IQR = 26–96%). Disturbance appears to decrease with increased spending. National Park (NP) shortfall (95%, $n = 1$, dashed grey line) is plotted. No comparison is available to plot disturbance for NPs, but Caro et al. (1999) found NPs to be more effective than other reserve types. Necessary spend (i.e. 100%) is also plotted against a disturbance level of zero.

range of 1.6–62 USD ha^{-1} reported from PAs in areas of high human population density in developing countries by Balmford et al. (2003), lending them further credibility.

Population density was reported to predict conservation spending by Balmford et al (2003; $r^2 = 0.36$, $n = 139$, $P < 0.001$). For both necessary spend and actual spend, population pressure was better at predicting observed values than were other measures of human pressure, such as land use conversion (Frazee et al., 2003) or population density within a fixed buffer of the reserve. The best population pressure predictor for both actual and necessary spend had a σ-value of 25, under which, pressure decays by half over a distance of around 20 km and to zero by 60 km (Fig. 2).

The positive exponential relationship between actual or necessary spend and population pressure could be a product of the way in which managers respond to high local pressure by increasing management effort (Nelson and Chomitz, 2009). On the other hand, actual spend is not only influenced by decisions based on threat levels, but also by opportunity; more populous areas may have a higher chance of receiving funding. However, this does not explain so well the finding that necessary spend increases with population pressure. To our knowledge, no other studies have investigated in detail the distance over which population exerts an effect and its correlation with PA management spending, despite it being an intuitive determinant of expenditure. These results are informative not only in maximising the proportion of variation explained by the models, but also in shedding light on the distance over which local human populations impact reserves in the EAM. Although it could be argued that the higher funding in areas of high population pressure is a result of greater stimulus or increased ability to raise funds, we think that pressure is more likely to drive the increased spending, particularly as the distance over which populations exert pressure (Fig. 2: $\sigma = 25$) is consistent for both actual and necessary spend and is similar to that found in other studies of resource use in the EAM (Green et al., in preparation).

Terrain ruggedness appeared to be negatively correlated with actual and necessary PA management costs. We hypothesise that the most rugged areas are the least accessible and least vulnerable to extractive resource use, so mitigating the effect of humans and resulting in decreased actual and necessary management costs.

Nevertheless, this analysis provides a realistic framework for estimating the actual and necessary costs of management in a complex system with complex funding pathways.

Several studies have shown particularly strong negative relationships between spend per unit area and total area (Balmford et al., 2003, 2004; Bruner et al., 2004; Frazee et al., 2003; McCrea-Strub et al., 2010) and we also found total area of reserve group to be a useful explanatory variable for necessary spend. This could be due to decreased costs of controlling unauthorised ingress, which is expected to scale in direct proportion to the length of the perimeter. Furthermore, increased total area of the reserve group could also lead to greater economies of scale and decreased costs per unit area. This effect may also exist for actual spend but is difficult to detect, as individual reserve attributes are masked by the unavoidable aggregation of PAs into reserve groups for analysis. We also found that the number of PAs in a reserve group was important in predicting both actual and necessary spend. As the number increased, spend per unit area decreased. This is to be expected due to streamlining of the administrative side of operations (offices, management salaries) and pooling of resources (vehicles and equipment).

PA type was a significant predictor in our model of actual spend. LAFRs (under local government) receive least funding, while NFRs and then NRs (both under central government) receive more and NPs (under TANAPA) receive most. This order roughly corresponds to their protected area categories (IUCN, 2001), with higher category reserves currently receiving more funding. LAFRs in the EAM generally have no IUCN category assigned as they are generally not of particular biodiversity importance and managed for resource extraction. Meanwhile many NFRs in the EAM have been coded as category IV PAs and NRs classified as category II PAs – the same as Tanzanian NPs (Burgess and Rodgers, 2004; Forestry and Beekeeping Division, 2007). Although our analysis of PA spending and forest condition (Fig. 4) is both speculative and rough, it does suggest that management effectiveness of Tanzania's PAs could be expected to improve under an adequately funded system. Obviously, differences in performance are not all down to funding. Governance is also likely to play a major role. This may explain the difference between our NFRs and NRs, which are modelled as having similar level of funding currently, yet NRs have a lower number of trees cut (Fig. 4).

This work contributes significantly to our understanding of the funding shortfalls in the current PA network while also providing information that can help to identify areas where we might maximise efficiency of effective conservation under future networks. We can also begin to think about the distribution of these costs – 22% of recurrent and capital costs are funded by non-governmental organisations (largely internationally funded), while 73% is from central government and 5% from local government. Furthermore, our model of necessary spend can be used to estimate likely costs under future scenarios of population growth and migration (Platts, 2011 develops models of future population pressure under different scenarios). This information is in a format that can be readily used by those working with systematic conservation planning in the region, while the simple message that, where possible, avoiding areas of high population pressure will keep costs down can also be applied very simply.

Models to predict necessary spend were less robust than that for actual spend, reflecting the smaller sample size and the errors associated with the unavoidably subjective assessment of how much money effective conservation would require. In addition, the grouping of reserves led to the analysis being conducted at smaller sample sizes, and diluted the effect of individual reserve attributes. This reduces the power to see smaller but still important effects. However, despite these difficulties, our models explained 39% of variation in necessary spend (weighted average; Table 4a). Although published global and international models are available, these are unlikely to perform so well at sub-national scales or for this type of reserve system (Balmford et al., 2003, 2004; Bruner et al., 2004; Frazee et al., 2003; Moore et al., 2004).

Frazee et al. (2003) suggested that biodiversity hotspots "must be bargains indeed" for conservation investment. This work goes some way towards enumerating exactly what this bargain might look like in the EAM. The current system of PAs, as recognised and mapped by the WDPA (IUCN, 2010), covers 17% of the EAM (861,254 ha) and our estimates of necessary spend predict that this could be effectively protected at a cost of 6.5 million USD y^{-1}. Although not an insignificant sum, it can be put into context by comparing it with Tanzania's military expenditure in 2008/9 of 225 million USD or to the 50 million USD received by TANAPA in tourism revenue alone in 2007/2008 (SIPRI, 2010; TANAPA, 2009). So, with the important caveats that management cost is only one part of the total cost of conservation (as part of the Valuing the Arc project [http://www.valuingthearc.org], we are quantifying indirect costs of conservation in the EAM: damage by wild animals and opportunity costs) and that there are more PAs not captured in the WDPA – particularly those under community based natural resource management (Burgess and Rodgers, 2004), conserving the EAM is not necessarily expensive. Just 3% of the military budget or 13% of the revenue generated by tourism to Tanzania's NPs could cover the management costs of effective conservation across 17% of one of the biologically richest mountain systems on the planet.

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