Incorporating Costs and Processes into Systematic Conservation Planning in a Biodiversity Hotspot

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Declaration

This dissertation is the result of my own work and includes nothing which is the outcome of work done in collaboration except where specifically indicated in the text. It does not exceed 60,000 words in length and no part has been submitted for another degree or diploma.

Signed: ……………………….

Jonathan Green

Given inadequate budgets with which to stem the rapid destruction of biodiversity, conservationists must set clear priorities for action. Systematic Conservation Planning (SCP) is an approach that uses spatially explicit data to identify areas that meet conservation targets efficiently, usually focusing on species' representation. Only rarely is the long-term persistence of species taken into account and the costs of conservation are usually ignored. I use the Eastern Arc Mountains of Tanzania as a study area to develop novel methods for creating and integrating the necessary data to fill these gaps in a developing country context. These mountains exhibit exceptional biodiversity but are also highly imperilled.

I describe the biological data that I assembled for use in a series of SCP analyses. Fine-scale distribution models for species were mapped for over 500 animal and plant species of conservation concern. I then mapped Ecological and Evolutionary Processes (EEPs), which are crucial to species' persistence and contribute to healthy ecosystem functioning. My analyses show how the inclusion of biological processes can significantly alter priorities when compared to prioritisation using information on species' presence alone. Despite their importance, EEPs are often excluded from SCP. This is largely due to the difficulties involved in expressing them quantitatively and in optimising reserve networks to represent them at a minimum cost. This reluctance should be challenged, otherwise reserve networks will, over time, lose those elements of biodiversity that they were established to conserve.

I also investigate conservation costs. Despite chronic underfunding for conservation and the recognition that funds must be invested wisely, few data on the costs of conservation are available at the spatial scales needed to inform local site management. I present methods for estimating and mapping protected area management costs, wildlife damage cost and the opportunity costs of conservation. Costs are highest in densely populated and cultivated areas, particularly in the north, whereas large areas of the more remote mountain blocs in the south show lower costs.

Integrating these data into SCP demonstrates that using real cost data (rather than assuming that cost per unit area is homogenous) alters priorities and increases the efficiency of conservation within the Eastern Arc. Importantly, the efficiency savings realised through using cost, rather than area, to prioritise conservation efforts were found to be most pronounced when budgets were limited so that not all conservation targets could be met.

Acknowledgements

First of all, I would like to thank my supervisors - Andrew Balmford for his energy and enthusiasm and Kerry Turner for his imperturbability and calm. They formed a conscientious and supportive ministering team and I am sure that they are as relieved as I am that this is all over. I have learnt a lot through observing them and through their teaching and I will always be grateful to their generosity of time and spirit which benefited me so much. I am also grateful to my PhD advisors David Aldridge and Rhys Green.

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Last of all, I thank Kate. She has done all of the above and more and even when I have been a miserable git, she has, mostly, responded with patience and kindness. She is a joy to be with and I cannot thank her enough.

Acronyms and abbreviations

AIC_c:	Akaike's Information Criterion with a correction for finite sample sizes
AO:	Area of Occurrence
AUC:	Area Under the Curve
BLM:	Boundary Length Modifier
BBC:	British Broadcasting Corporation
CBNRM:	Community-Based Natural Resource Management
CFPF:	Conservation Feature Penalty Factor
CR:	Critical (threat list status)
DEM:	Digital Elevation Model
E:	Endemic
EAM:	Eastern Arc Mountains
EAMCEF:	Eastern Arc Mountains Conservation Endowment Fund
EEP:	Ecological and Evolutionary Process
EN:	Endangered (threat list status)
EO:	Extent of Occurrence
ESH:	Extent of Suitable Habitat
FAO:	Food and Agriculture Organization
FBD:	Forestry and Beekeeping Division
GAA:	Global Amphibian Assessment
GIS:	Geographic Information System
GMA:	Global Mammal Assessment
GR:	Game Reserve
ha:	Hectare
HSI:	Habitat Suitability Index
IQR:	Inter-Quartile Range
IUCN:	International Union for the Conservation of Nature
JFM:	Joint Forestry Management
kg:	Kilogramme
km:	Kilometre
LAFR:	Local Authority Forest Reserve
LC:	Least Concern (threat list status)
m:	Metre
MNRT:	Ministry for Natural Resources and Tourism
NE:	Near-Endemic

NFR:	National Forest Reserve
NP:	National Park
NPV:	Net Present Value
nr:	Not recognised (threat list status)
NR:	Nature Reserve
NT:	Near-Threatened (threat list status)
PU:	Planning Unit
ROC:	Receiver Operating Characteristic
RR:	Restricted Range
RVI:	Relative Variable Importance
SCP:	Systematic Conservation Planning
s.d.:	Standard deviation
SPF:	Species' Penalty Factor
TANAPA:	Tanzania National Parks
TZS:	Tanzania Shillings
UMNP:	Udzungwa Mountains National Park
USD:	United States Dollars
VFR:	Village Land Forest Reserve
VRM:	Vector Ruggedness Measure
VU:	Vulnerable (threat list status)
WDPA:	World Database on Protected Areas
WMA:	Wildlife Management Area
y:	Year

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1. Introduction

"*I should say no more fertile soil could be found in the world, and it will, I am sure, produce every tropical plant…On my return next summer I should be happy to welcome a scientific botanist as my guest, and should feel well repaid if he would teach us how to turn the vegetable wealth of the country to account.*"

Reverend J. P. Farler, speaking of the Usambara Mountains, Tanzania (1879)

1.1. Systematic conservation planning

1.1.1 Development of conservation planning tools

Faced with the rapid and seemingly inexorable destruction of biodiversity and inadequate budgets with which to stem its loss (Bruner et al. 2004), conservationists must set clear spatial priorities for action. Yet historically, most reserves have been established in places unsuitable for mainstream economic activity, rather than in the most rewarding areas for conservation (Pressey 1994; Margules and Pressey 2000; Carwardine et al. 2007). Termed "*ad hoc* reservation" (Pressey 1994), this legacy persists through the necessary inertia that exists in the creation and degazettement of protected areas. As a result, current protected area networks are generally inadequate in their representation of biodiversity at both global and local scales (Jennings 2000; Rouget et al. 2003a; De Klerk et al. 2004; Fjeldså et al. 2004; Rodrigues et al. 2004b; Rodrigues et al. 2004a; Burgess et al. 2005; Langhammer et al. 2007; Beresford et al. 2011a).

To address these inefficiencies biologists have developed Systematic Conservation Planning (SCP), which aims to identify areas that collectively and efficiently meet conservation targets, such as species' representation (Margules and Pressey 2000; Sarkar et al. 2006). The fundamental tenets of SCP are efficiency, transparency and accountability (Margules and Pressey 2000). The first step is to decide upon the conservation features to be used in the analyses and to set targets for their representation within a hypothetical network of reserves. Data on the spatial distribution of these conservation features (usually species or habitat types are chosen as surrogates for total biodiversity) and on the spatial distribution of conservation costs are then collected. Using optimisation techniques, the chosen SCP software is then employed to identify the near-cheapest network for meeting the representation targets of these conservation features (referred to as a "minimum set" approach). The software can also be used to maximise the number of conservation features whose targets are met for a pre-determined budget or area constraint (a "maximum coverage" approach; Moilanen et al. 2009b).

1.1.2 Current limitations to Systematic Conservation Planning

Originally devised as an objective decision aid for protected area network design and largely based on species' distributions, technological improvements have enabled SCP's ascent to ever-more complex analyses (Moilanen et al. 2009a); SCP software is now able to optimise for hundreds of conservation features in thousands of planning units and techniques for including connectivity between species and between conservation areas have been developed (Ardron et al. 2008; Watts et al. 2009). Furthermore, recent improvements in the geographic and taxonomic coverage of species' distribution data and concurrent improvements in their availability and resolution through digitization of resources and compilation of vast databases have facilitated biologically realistic conservation planning analyses (Elith et al. 2006; Rondinini et al. 2006; Boitani et al. 2011). Nevertheless, there are still gaps: in how far SCP efforts target long-term species' persistence through incorporation of Ecological and Evolutionary Processes (EEPs), and how far they incorporate socio-economic factors such as the costs of conservation action (Fjeldså et al. 1997; Balmford et al. 1998; Balmford et al. 2000; Naidoo and Adamowicz 2005; Naidoo et al. 2006; Pressey et al. 2007; Ferrier and Wintle 2009; Klein et al. 2009; Moilanen et al. 2009a; Wilson et al. 2009).

EEPs include phenomena such as long-term environmental stability, species' movements and species' interactions. These processes are rarely considered explicitly in SCP analyses, which is partly because there are few quantitative data about the spatial requirements for such processes to operate. Even when these data are available, optimising for efficient conservation of EEPs is computationally difficult when the process operates across several planning units, and when the absence of a single spatial link between those planning units can prevent the process from occurring. In this thesis, I consider eleven EEPs that are important to species' persistence and I incorporate nine (those for which spatial data exist and which can be influenced by a regional conservation plan) into my analyses.

Systematic conservation plans attempt to allocate scarce resources to achieve specific objectives and, as such, represent a classic economics problem (Polasky et al. 2001; Morrison and Boyce 2009); however, surprisingly few prioritisation exercises include data on conservation cost. More often, area is used as a surrogate for cost, making the assumption that cost per unit area is homogenous across the landscape. That costs do vary spatially is evident and is the basis for real estate trading and Ricardo's (1821) law of rent, which states that land value increases according to the relative economic advantages of its situation or productivity. Moreover, the magnitude of variation in the costs of conservation may be more than the variation in biodiversity benefits, so taking costs into account can profoundly alter conservation priorities (Naidoo et al. 2006). It has also been found that when several spatial options exist for conservation of a biodiversity feature, the inclusion of socio-economic data can prove particularly decisive in developing spatial priorities (Ardron et al. 2008). Developing spatially explicit maps of conservation cost enables more efficient reserve network design (Polasky 2008), demonstrates where costs are borne (Balmford and Whitten 2003), and aids implementation (Knight et al. 2006b). In this thesis, three types of conservation cost are considered: management costs of protected areas, opportunity costs of conservation and wildlife damage costs. Management costs are important because these are often borne by the government – the key player in conservation policy decisions.

Opportunity costs and damage costs are also a crucial ethical consideration for conservation planning, as they are largely borne locally and frequently by those least able to afford it (Balmford and Whitten 2003). Inclusion of information on all of these costs during the planning stage will help identify and, potentially, avoid burdens on local communities as well as increasing the efficiency and effectiveness of a protected area network.

Although SCP studies have begun to consider EEPs (Burgess et al. 2006; Nicholson et al. 2006; Rouget et al. 2006), and conservation costs (Balmford et al. 2000; Balmford et al. 2003; Moore et al. 2004; Naidoo et al. 2006; Polasky 2008), this work is still in its infancy, especially in developing countries where data are often scarce. In this thesis I develop methods to incorporate considerations of long-term species' persistence and conservation cost into SCP in a developing country context. Given the shortfall in conservation funding in most countries and the coincidence of poverty and biodiversity at a global scale, such an approach is of potentially widespread importance (Bruner et al. 2004; Fisher and Christopher 2007).

1.1.3 Software and data considerations

In this section, I describe some of the key features of Marxan (the SCP programme used throughout my analyses), the metrics used to measure priority, the program settings used and the specific data that I incorporated in my analyses. In conservation planning, an important consideration is the size of the Planning Units (PUs) that are used. These are the units of selection that are used in SCP and they represent the scale at which decisions are made to include or exclude parcels of land. Some studies use uniform squares or hexagons which can be removed or added from a hypothetical reserve network, while others use natural features, such as river catchments (Klein et al. 2009; Nhancale and Smith 2011). Most incorporate information on governance or political boundaries, such as protected areas and country borders. In the following chapters I use three different planning unit scenarios (Figure 1.1):

1. Using uniform squares as PUs (9 km^2).
2. Inclusion of the current system of protected areas (IUCN 2010a), surrounded by the grid of squares described in PU scenario one.
3. The same PUs as in scenario two, except that the current system of protected areas is fixed and cannot be removed from the solution. Thus, areas that are most complementary to the current system are selected.

The third scenario is the most realistic and is the approach taken most often by conservation planners; however, scenarios one and two also provide useful information to decision makers. They provide information on the minimum set of areas that would represent all targets adequately (scenario one) and on the relative contribution of specific protected areas to conservation targets (scenario two). In deciding the size of the squares used in the scenarios (for unprotected areas only in scenarios two and three), I considered several factors. PU size has implications for the viability of species' populations 'conserved' within it, the likelihood of a unit being able to capture particular biological processes, and the feasibility and efficiency with which the PU can be managed. Smaller PUs can result in selection of areas too small to sustain viable populations or biological processes, and they can also lead to the design of a reserve system that is too disaggregated to be managed efficiently (Possingham et al. 2000). On the other hand, using PUs that are too large can produce less efficient protected area networks, as priority PUs are likely to include larger areas of land of low conservation value (Nhancale and Smith 2011). Throughout these analyses I used square PUs of 9 km^2 (3 km by 3 km). Units of this size could probably hold viable populations for most species (except for some larger mammals, birds and trees) without sacrificing efficiency. In addition, this is the median size of state-managed protected areas in EAM districts (Figure 1.2). It therefore seems to be an appropriate scale at which to consider modifications to the existing reserve network (see also chapters three and four).

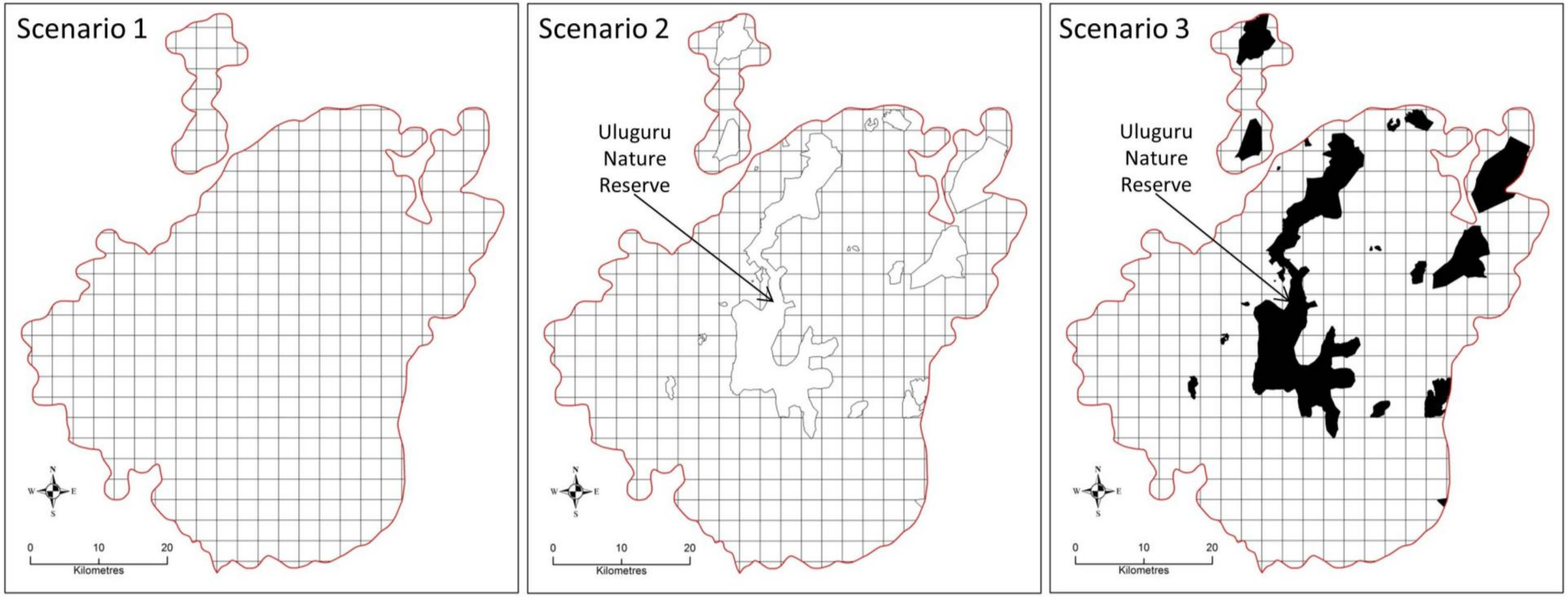


Figure 1.1. Planning unit scenarios illustrated for the Uluguru Mountains (outlined in red). The three different types of planning units used in analyses are shown. In scenario one (left hand panel), the entire area is divided according to a square grid (9 km^2). In scenario two (middle panel), current protected areas are also included as planning units (surrounded by the grid of squares described in scenario one. In this scenario, protected areas can be removed from the solution if they do not contribute to meeting conservation targets. Scenario three (right hand panel) uses the same planning unit design as scenario two; however, in scenario three, protected areas cannot be removed from the solution and are always in the final reserve network.

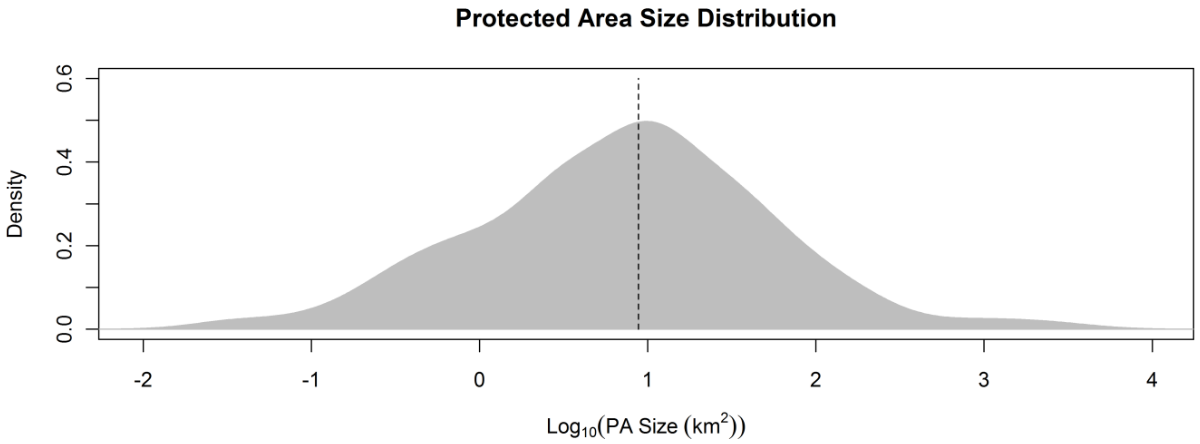


Figure 1.2. Kernel density plot of state-owned protected area size in the Eastern Arc Mountains. Median size is 8.8 km^2 (dashed line).

SCP is based upon the costs and benefits that would arise from a particular conservation action in each planning unit. In the majority of studies, conservation action is often equated with the establishment of protected areas (Ferrier and Wintle 2009); however, this is not a prescription of the approach. In the following analyses, I use protected area establishment as an example. Nonetheless, in some situations a different type of intervention may afford conservation targets equal or greater protection under a more equitable or efficient model of governance.

SCP can be based on a multitude of different data, depending on the objective of the stakeholders and planners. Common examples are to maximise representation of habitat diversity, species diversity or ecosystem service provisioning whilst minimising area or cost (e.g. Cowling and Pressey 2001; Ceballos et al. 2005; Rondinini et al. 2005; Chan et al. 2006; Ardron et al. 2008; Egoh et al. 2010). A summary of the data that I used and the way in which I incorporated them into SCP is shown in Figure 1.3. The species and EEPs (conservation benefits; green box) found within each planning unit (which can include information on the protected area network; brown box) is calculated first (pre-processing; red box). The costs of site acquisition (opportunity costs) and management (both site costs; left hand panel of orange box) are then estimated for each planning unit (pre-processing; red box). These data, describing the acquisition and management costs and the total benefits of including each site in a hypothetical reserve network, are then entered into Marxan, the priority setting algorithm (priority setting; black box). The planning units layer is also used to derive a boundary length file, into which information on damage costs is incorporated (boundary costs; right hand panel of orange box). The resulting boundary cost file is then entered into the priority setting algorithm, modified by a Boundary Length Modifier (BLM), a simple multiplier that enables the weight of damage costs to be altered, relative to the site costs. Targets for conservation features (i.e. species and EEPs) and the penalties for missing these targets are then set, along with the settings that determine how the algorithms run (program settings; blue box). The penalties (known as Species' Penalty Factors or SPFs) are a cost which is added to the total cost of the solution whenever a conservation feature is not represented within the reserve system (they are sometimes called Conservation Feature Penalty Factors or CFPFs to make the point that it is not just species' representation that can be used as targets). Marxan will try to minimise these penalties by meeting the conservation targets set by the user. It is possible to assign different penalties to different targets to reflect their importance; however, throughout my analyses all conservation targets received the same SPF. Targets for each species (or conservation feature) are defined by the operator and define the area of a species' range that should be incorporated into the reserve network. Throughout these analyses, Marxan was used to assign spatial priorities (black box in Figure 1.3). Marxan works by using a simulated annealing algorithm to add or remove planning units at random from a hypothetical reserve network (each selection or removal is one iteration). At each iteration the cost of the solution is calculated as the conservation costs (management, opportunity and damage costs) plus any penalties (the SPF for any conservation features that are not adequately represented). The basic calculation used by Marxan is:

$$Solution\ Cost = \sum_{PU} Cost + BLM \sum_{PU} Boundary + \sum_{targets} SPF$$

Equation 1.1.

where Cost is the cost of a planning unit and Boundary is the length of the external boundary of a planning unit under the current configuration. These are both summed across every planning unit (PU). BLM is the Boundary Length Modifier which determines the relative importance of minimising the boundary length compared to minimising the Cost of PUs. For every conservation feature whose representation target is not met, the Species' Penalty Factors (SPFs) are summed (Ball and Possingham 2000).

The basic premise is that when the new solution offers an improvement (a decrease in cost), the change is always accepted, otherwise it should be rejected. In order to prevent the solution from becoming trapped at local optima, the algorithm will occasionally accept a solution worse than the current one. This is termed a "temperature decrease" and the number of times that the algorithm does this is set by the user. The likelihood of improvements (i.e. decreases in cost) being accepted is always one. The likelihood of other changes (i.e. increases in cost or "temperature decreases") being accepted is higher at the beginning of the process than at the end. This feature enables the algorithm to avoid becoming trapped at local optima but, as the run progresses, move towards a solution that is, globally, near-optimal. One million iterations were performed in each run and either 100 or 1000 runs were made (stated in the individual chapter methods). This approach offers two metrics of priority: the ‘best’ solution shows which planning units were included in the cheapest solution of all the runs, while irreplaceability is the number of times (usually measured as a percentage) that a planning unit is in the final solution of each run and is a useful metric to gauge the uniqueness of the biota (or other conservation features) of a planning unit (Wilson et al. 2009). Once efficient spatial priorities have been derived, these can be compared to threats to inform decisions about the temporal scheduling of priorities.

All geographical information systems (GIS) analyses presented in this thesis were conducted in ArcGIS v9.3 and v10 (Environmental Systems Research Institute 2009), and GIS layers were projected to UTM zone 37S. All statistical analyses were conducted in R (R Development Core Team 2009).

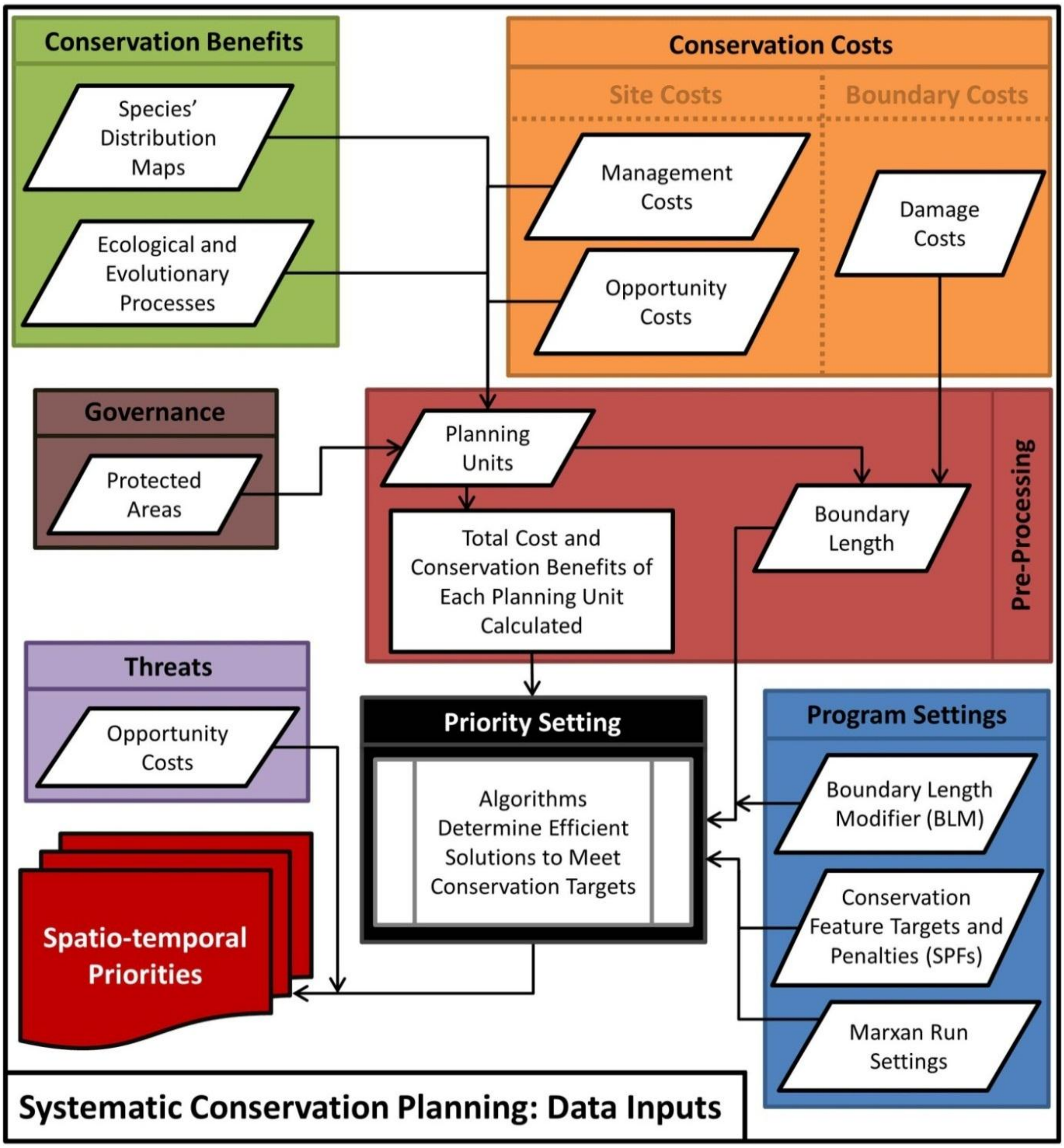


Figure 1.3. Data inputs to Systematic Conservation Planning (SCP). The types of inputs that can be used in an SCP exercise are shown (coloured boxes). In addition, the schematic shows, in white boxes, the specific data that are derived over the following chapters, the way in which they are incorporated into SCP and the steps taken to identify priorities.

1.2. Context

1.2.1 The Eastern Arc Mountains

The Eastern Arc Mountains (EAM) are a chain of mountains stretching from the Taita Hills in the south of Kenya through the east of Tanzania to the Udzungwa mountains in south-central Tanzania (Figures 1.4, 1.5; Burgess et al. 2007c; Platts et al. 2011). The forests on these mountains, reaching altitudes of just over 2,600 metres and representing remnants of a once vast forest ecosystem that was contiguous with the forests of Central Africa, are noted for their exceptionally high biodiversity (Lovett 1985; Burgess et al. 1998; Burgess et al. 2004b; Burgess et al. 2007c). New species are regularly described from the EAM (Stanley et al. 2005; e.g. Couvreur and Luke 2010; Loader et al. 2010; Loader et al. 2011), including two of the BBC's 'top ten' newly described species of the last decade – the grey-faced sengi (*Rhyncocyon udzungwensis*) and the kipunji (*Rungwecebus kipunji*), a new genus of primate (Davenport et al. 2006; Rovero et al. 2008; BBC 2010). These biologically diverse forest remnants have persisted due to the high orographic rainfall that the 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 (Mumbi et al. 2008). Since the Tertiary, as Africa gradually dried, the surrounding low-lying areas became savannah leaving the EAM as a crucial refuge for many species (Pócs 1998). The mountains host important biological processes, including seasonal and diurnal species' movements, long-term bio-climatic stability and speciation. These processes are both a cause, and an emergent property, of the mountains' extraordinary and widely recognised biodiversity and endemism (e.g. Lovett 1998; Brooks et al. 2001; Brooks et al. 2002; Burgess et al. 2007c). In concert with exceptional levels of habitat loss, this biodiversity has led to the identification of the EAM as part of the Eastern Afromontane biodiversity hotspot and their recent nomination as a World Heritage Site (Myers et al. 2000; Brooks et al. 2005; MNRT 2010). Recent work suggests that at least 76% of plant species in Tanzania are undescribed (Joppa et al. 2011).

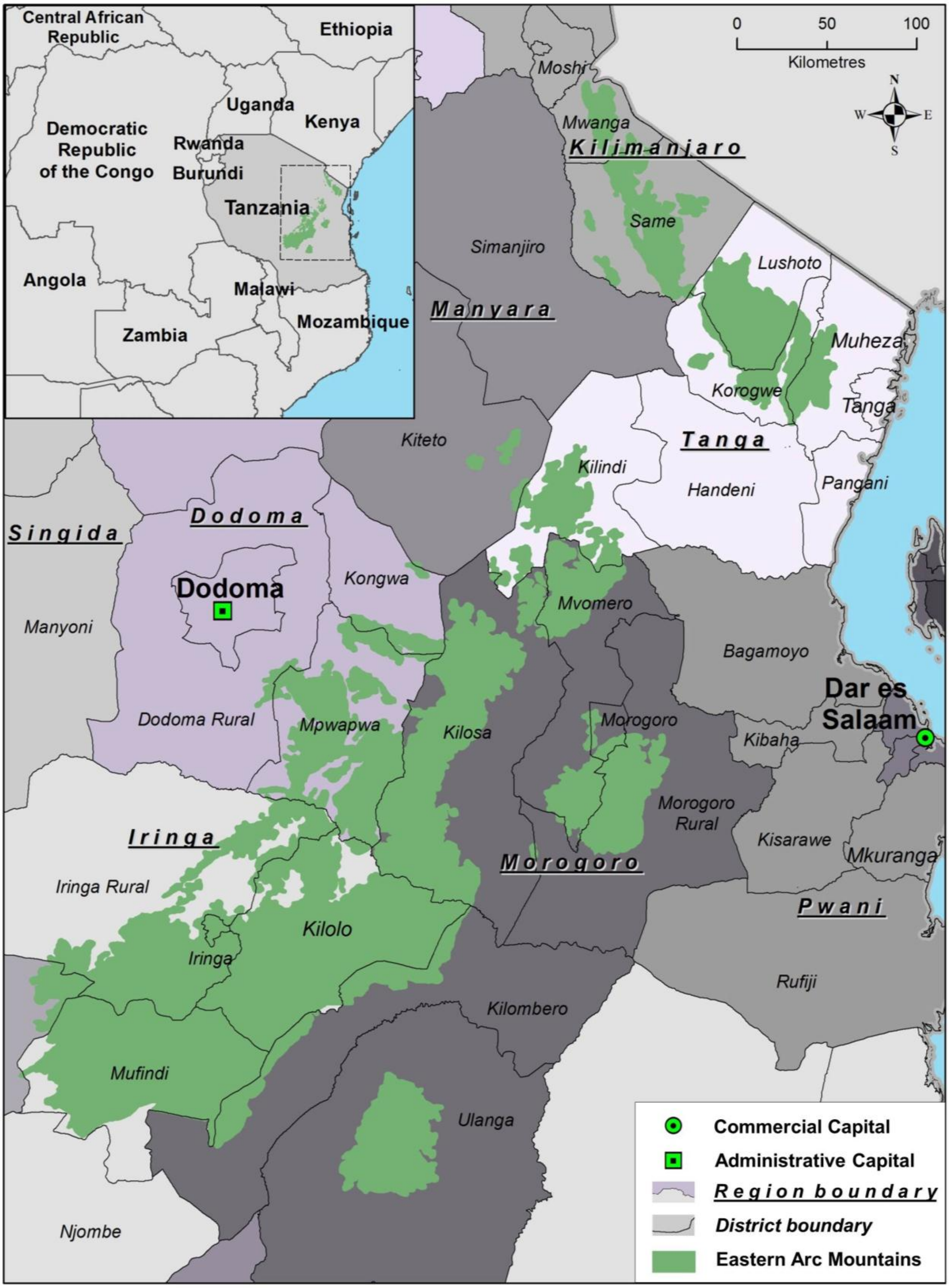


Figure 1.4. Geo-political map of the Eastern Arc Mountains (EAM) within Tanzania. The EAM cover significant proportions of 17 districts (in italics), falling within five regions (underlined). <u>Dodoma Region</u>: *Mpwapwa*; <u>Iringa Region</u>: *Iringa Rural*, *Iringa Urban*, *Kilolo* and *Mufindi*; <u>Kilimanjaro Region</u>: *Mwanga* and *Same*; <u>Morogoro Region</u>: *Kilombero*, *Kilosa*, *Morogoro Rural*, *Morogoro Urban*, *Mvomero* and *Ulanga*; <u>Tanga Region</u>: *Kilindi*, *Korogwe*, *Lushoto* and *Muheza*.

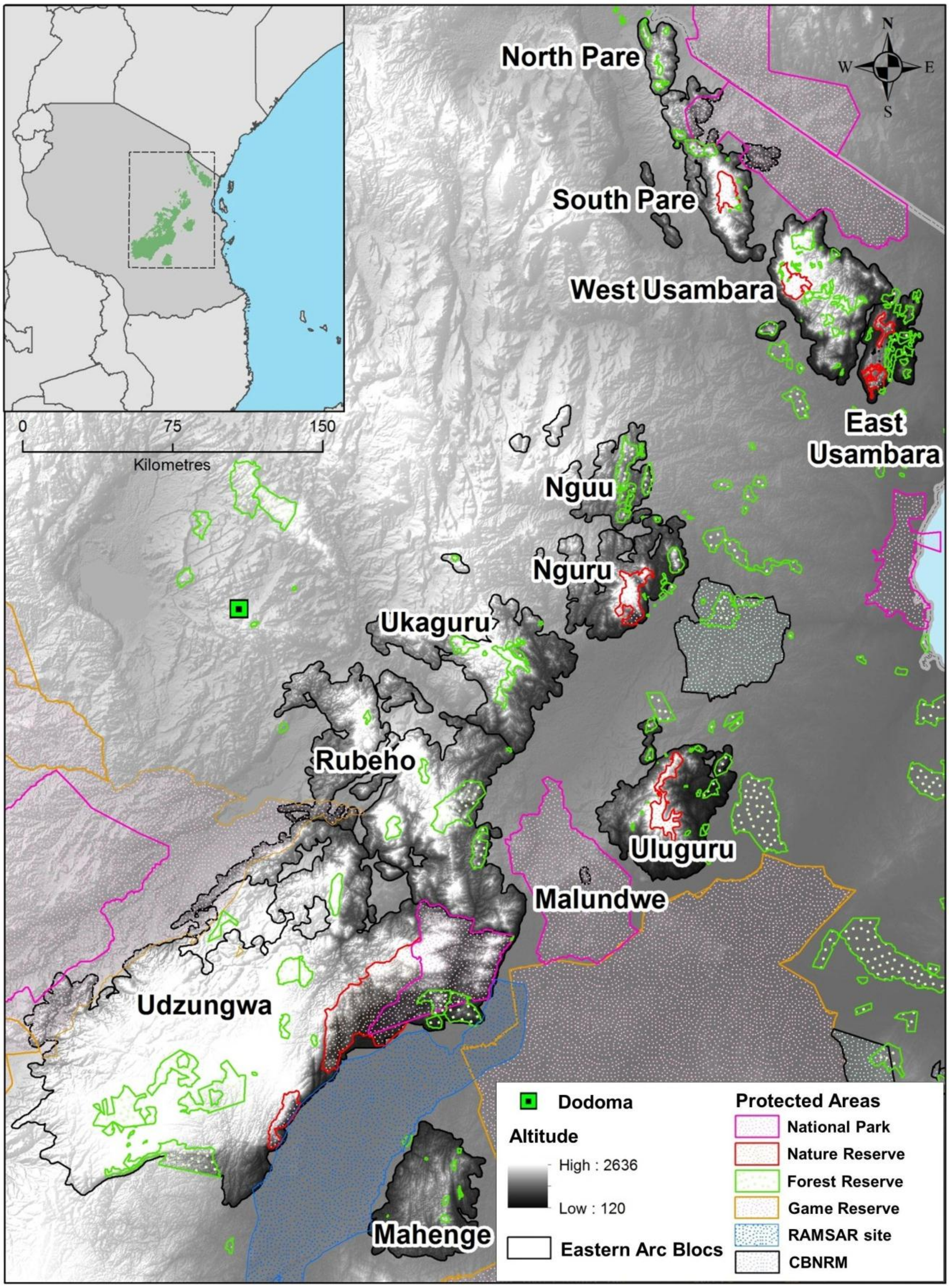


Figure 1.5. Mountain blocs, reaching altitudes of over 2600 metres, and protected areas of the Eastern Arc Mountains (EAM). State-owned protected areas fall under the control of three agencies within the EAM: National Parks are managed by the Tanzania National Park Authority (TANAPA); Nature Reserves and some forest reserves are managed by Central Government (Forestry and Beekeeping Division), with other forest reserves managed by local governments; Game Reserves are managed by Central Government (Wildlife Division). In addition, Community-based natural resource management has increased in recent years under management of village governments with assistance from local government. Lastly, the Kilombero Valley, which falls between the Udzungwa and Mahenge blocs, is a wetland of international importance and is designated as a Ramsar site.

1.2.2 History and current status of protected areas in the Eastern Arc

In the EAM, many forest reserves were declared under the German colonial administration in the late 19th Century (Burgess et al. 2007a). There was a steady increase in the area under protection during British rule from the early 20th Century until after the Second World War, when the area under protection increased by at least an order of magnitude, before resuming again its steady increase in the 1960s (Neumann 1992; Burgess et al. 2007a). Since the mid-1990s, there has been a focus on Community-Based Natural Resource Management (CBNRM), which has led to a large increase in two new categories of reserve over the last decade: Village Land Forest Reserves (VFRs) and Wildlife Management Areas (WMAs; Nelson et al. 2007; Blomley et al. 2008). In addition, some state-owned reserves now have areas that are under Joint Forest Management (JFM), in which control is shared between the government and village councils. State-owned protected areas in the Eastern Arc fall under the control of four agencies: Tanzania National Parks (TANAPA) manage all National Parks (NPs) in Tanzania, including Mikumi NP and Udzungwa Mountain NP within the EAM; Nature Reserves (NRs) and National Forest Reserves (NFRs; also called catchment forest reserves) are managed by the Central Government, under the Forestry and Beekeeping Division, while Game Reserves (GRs) are managed by the Central Government's Wildlife Division. Local governments manage Local Authority Forest Reserves (LAFRs). The median size of these state-owned reserves within the districts of the EAM is 8.8 km^2 (Figures 1.2 and 1.5).

In Tanzania reserves have been established for numerous reasons. Apart from designating protected areas for their high biodiversity value, areas have been protected for their high hunting value (e.g. Selous GR), low commercial value (e.g. Mikumi NP, of low value to pastoralists because of tsetse flies) and high ecosystem service value (e.g. National Forest Reserves, often established to conserve water flow regulation). Given that the reserve network has been developed over more than a century and that both the authorities making the decisions, and their motivations for reserve creation, have changed during this period, it is wise to now take stock of the current situation (Nelson et al. 2007). Finely resolved data are now available, or can be derived, to make a spatially explicit, EAM-wide assessment of the degree to which this reserve network meets biodiversity conservation targets. This is particularly pertinent in an area where there are significant, increasing anthropogenic pressures (Lovett 1985; Balmford et al. 2001; Liu et al. 2008) yet where there are major gaps in the PA system for threatened plants and vertebrates (De Klerk et al. 2004; Fjeldså et al. 2004; Rodrigues et al. 2004a; Burgess et al. 2005; Schmitt et al. 2009).

1.2.3 Threats to the Eastern Arc

Of a group of 25 of the most imperilled terrestrial areas in the world, the EAM are one of three identified as being least able to afford further deforestation (Brooks et al. 2002). Deforestation is largely driven by small-scale farming and charcoal production and only around 34% of the area's original land cover remains (Brooks et al. 2002; Burgess et al. 2002a; Newmark 2002; Ahrends et al. 2010; Fisher 2010). Approximately 46% of the EAM is under cultivation and the mountains have long been considered amongst the most valuable lands in Tanzania (Farler 1879). Human population pressure (described in more detail in chapter four) and growth are high in the region – particularly in large towns and cities such as Dar es Salaam, Ifakara, Iringa, Morogoro and Tanga – and this exerts further pressure on the mountains' ecosystems (Cincotta et al. 2000; United Nations 2011; Figure 1.6). These factors threaten not only the mountains' species, but also the biological processes that operate within them (Newmark 2002). Losing these may have severe consequences for the communities that depend upon the ecosystem services that they provide; water flow regulation, carbon sequestration, non-timber forest products, charcoal, firewood and tourism all represent valuable assets that are realised by local and global communities (Burgess et al. 2007c; Burgess et al. 2009).

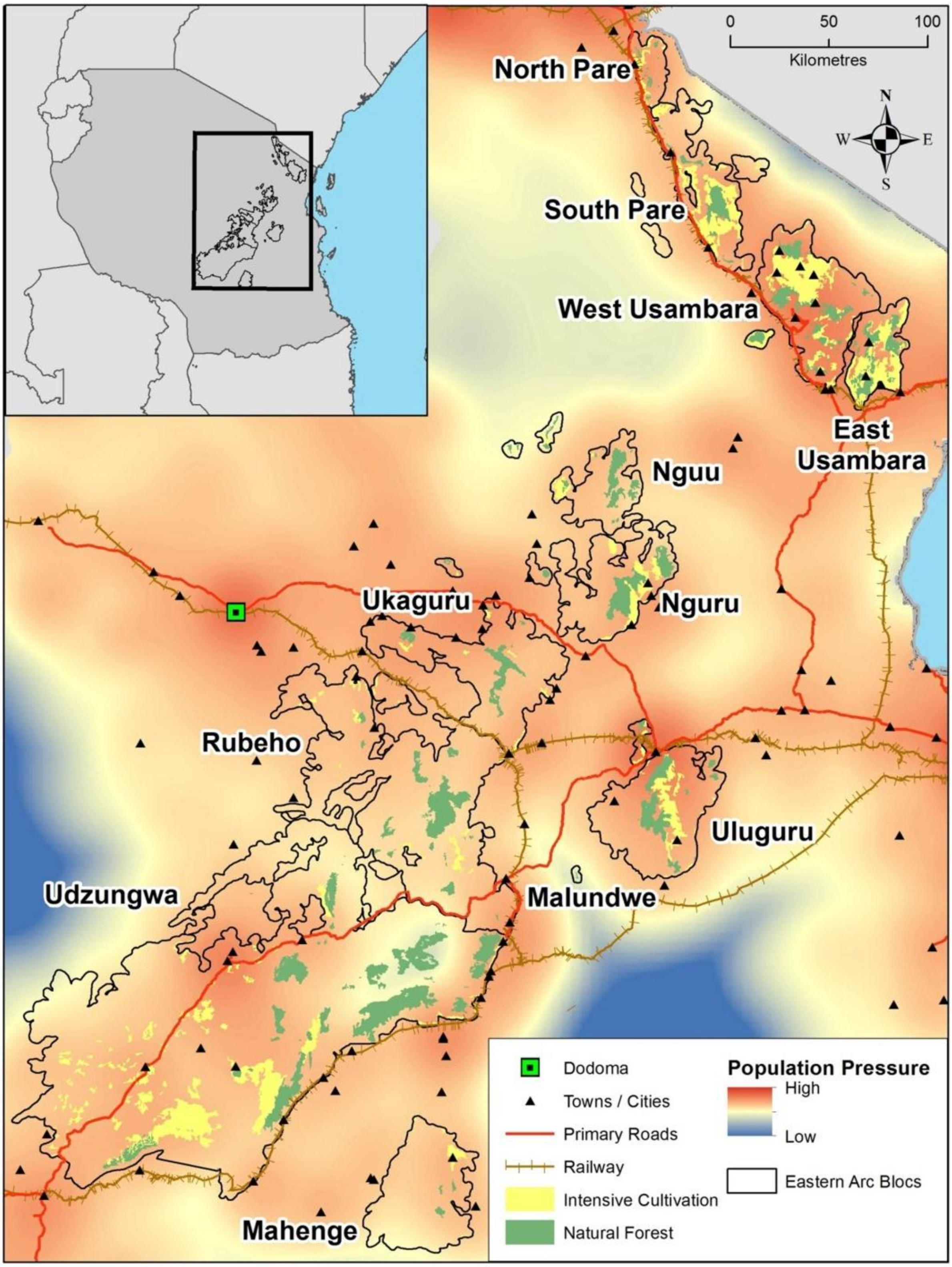


Figure 1.6. Anthropogenic pressures in the Eastern Arc Mountains (EAM). Human population pressure is high around the EAM and accessibility is facilitated by road and rail infrastructure. Intensive small-holder cultivation is a major driver of deforestation in the EAM.

Conflict between biodiversity conservation and economic development in the EAM is likely to increase under predictions of social, economic and climatic change: Tanzania's population is expected to triple over the next 40 years (United Nations 2011), it is likely to become a hotspot of under-nutrition (Liu et al. 2008) and, while climate change is expected to cause a decrease in crop yields over large areas of Tanzania, productivity in highland areas, such as the EAM, is expected to increase (Thornton et al. 2009). Thus the conflict arising from positive correlations between human populations and biodiversity is likely to worsen (Balmford et al. 2001; Burgess et al. 2007b; Fjeldså and Burgess 2008; Platts et al. 2011) and the use of finely resolved data to identify areas where conflicts between conservation and development goals are expected to be lower is crucial. The EAM are also a useful test system for developing these methods because (as in many other tropical regions) data on biodiversity value and conservation cost are scarce. Hence, modelling is integral to the approach I have adopted in my work.

Over next four chapters I investigate how best to create the data layers to conduct an SCP analysis that will consider long-term species' persistence and the costs of conservation alongside biodiversity patterns. In chapter two, I describe my work to collate information on species' diversity for species of conservation concern. Then in chapter three I investigate which Ecological and Evolutionary Processes (EEPs) are most important to the long-term persistence of EAM species, and where these occur. In chapter four, I develop a model to estimate the cost of effective protected area management and then in chapter five I consider how best to measure the indirect costs of conservation: wildlife damage costs and the cost of foregone opportunities to local communities. In chapter six, I then explore how the current PA system performs and how incorporation of these various data sets influences conservation priorities across the EAM. Finally in chapter seven key points from each of the previous chapters and recurring themes are discussed.

2. Species' patterns

"The jungle teems, but in a manner mostly beyond the reach of the human senses"

Edward O. Wilson, The Diversity of Life (1994)

2.1. Introduction

2.1.1 The Eastern Arc Mountains

Running from the south-east of Kenya to southern Tanzania, the Eastern Arc Mountains (EAM) are a chain of disjunct mountain blocs, which share common climatic characteristics and biogeography (Lovett 1985; see chapter one for detailed description). The EAM exhibit extraordinarily high biodiversity (Burgess et al. 1998; Burgess et al. 2007c) and new species are regularly described from the region (e.g. Menegon et al. 2002; Doggart et al. 2006; Mariaux and Tilbury 2006; Menegon et al. 2008; Rovero et al. 2008; Loader et al. 2010; Loader et al. 2011). This exceptional biodiversity and endemism, in concert with exceptional levels of historical habitat loss, have led to their identification as a biodiversity hotspot (Lovett 1998; Myers et al. 2000; Brooks et al. 2001; Brooks et al. 2002; Mittermeier et al. 2004; Burgess et al. 2007c). Species inventories at coarse scales, such as lists of species within protected areas, have underpinned classifications such as this; however, fine-scale species' distribution data are much needed for an EAM-wide Systematic Conservation Planning (SCP) prioritisation of on-the-ground conservation within the EAM (Burgess et al. 2002b). Despite this need, collecting and analysing such data can be a time-consuming and expensive exercise.

2.1.1.1 Resolution and scale of species' distribution maps

At its crudest, a species' distribution map is an Extent of Occurrence (EO), which describes the limits to its range (Gaston 1991). An EO should include all occurrences of a species within it, but will also contain many areas which are not used by the species (commission errors). At the other extreme, point occurrences describe specific locations where the species is known to occur (often such data are linked to museum specimens), and so, in theory, avoid commission errors. Species will, however, occur unrecorded in many places away from the observed point localities (omission errors). Both EO data and point locality records are widely available, but their high levels of commission and omission errors, respectively, limit their usefulness to conservation planners. Various modelling approaches have been used to move such species' distribution estimations from either end of this spectrum towards an Area of Occupancy (AO) map – the area that is actually used by a population. One popular approach is to link specimen point locality records with climatic and geographical information in a regression-based model to infer a bioclimatic envelope within which the species is expected to occur (e.g. Peterson 2001). Alternatively, others have begun at the opposite end of the spectrum, using the EO as their starting point and extracting from within it only the most suitable habitat, based on an expert-derived Habitat Suitability Index (HSI; Rondinini et al. 2005; Boitani et al. 2008; Beresford et al. 2011b;

Boitani et al. 2011; Rondinini et al. 2011). This approach uses information on a species' habitat preferences (from the HSI) and constrains this to the species' known geographic range (EO) to generate an Extent of Suitable Habitat (ESH) model (da Fonseca et al. 2000; Rondinini et al. 2005). In reality most modelling approaches will use both point locality data and EO data - for example, regression-based models might be restricted to the known EO, while ESH models might use information from point locality records to provide information on habitat preferences.

In SCP, use of ESH models (rather than EOs) will help avoid commission errors and the costly mistake of assuming a species is conserved when in reality it may not actually be within the area estimated (Midgley et al. 2003; Rondinini et al. 2005). Although using coarse-scale EO data may be a pragmatic response to the urgent need for conservation strategy, refinements to such data by modelling the ESH is a cost-effective way to dramatically reduce the coarseness of distribution maps, improving the accuracy with which they represent a species' AO (Larsen and Rahbek 2003). Given the trade-off between delaying conservation action whilst ever more-refined data are compiled and using less accurate data to take less informed conservation action before it is too late, this technique is an efficient way with which to improve the state of knowledge efficiently and quickly (Grantham et al. 2009).

2.1.1.2 Surrogacy

The use of surrogates in conservation planning is unavoidable, as the true level of biodiversity can never been known. Conservationists must try to use surrogate taxa that will correlate as closely as possible with actual biodiversity and surrogates are best between taxa with similar ecological requirements (Mortelliti et al. 2009). Several authors argue for as comprehensive a group of surrogate taxa as possible to best represent the true biodiversity of a site (Margules et al. 2002; Pressey 2004; Larsen et al. 2012), while a review by Rodrigues and Brooks (2007) supports the use of cross-taxon surrogates (rather than using environmental surrogacy) and emphasises the importance of using species distribution models (rather than point localities) to set priorities.

2.1.2 Previous studies and aims

In the EAM, conservation planning has generally relied upon species inventories and evidence of species' occurrences in specific locations (e.g. Jones et al. 2007; MNRT 2010). With resources scarce, this has been a pragmatic solution, yet, as a result, no EAM-wide conservation plan for multiple taxa has been conducted using spatially explicit distribution data. This fact is particularly pertinent in an area where there are significant, increasing anthropogenic pressures (Lovett 1985; Balmford et al. 2001; Fjeldså 2007; Fjeldså and Burgess 2008; Liu et al. 2008; Ahrends et al. 2010), but where, at an international scale, there are major gaps in the protected area system for threatened plants and vertebrates (De Klerk et al. 2004; Fjeldså et al. 2004; Rodrigues et al. 2004b; Rodrigues et al. 2004a; Burgess et al. 2005; Fjeldså 2007; Beresford et al. 2011b). Concurrence of high biodiversity and anthropogenic value, at the resolution of a one-degree grid, means that higher resolution data must be used to identify areas where conservation conflicts can be minimised and conservation efforts implemented most efficiently (Burgess et al. 2002b; Fjeldså and Tushabe 2005; Fjeldså 2007; Fjeldså and Burgess 2008).

For species of conservation concern (section 2.2.2), I compile published ESH models for amphibian, mammal and plant distributions and I develop ESH models for birds and chameleons. I also investigate the effect of geographical and taxonomic bias on modelled patterns of species' distributions. Using these data, I mapped species richness and then used the SCP software, Marxan, to develop spatial priorities for conservation that meet representation targets for each species within a minimum sized protected area network. These analyses do not account for the biological processes that sustain biodiversity in the long-term (Wilson et al. 2009) and nor do they include information on the costs of conservation, which are crucial for efficient conservation planning (Polasky 2008). These data are detailed in chapters three to five and their integration into SCP analyses described in chapter six.

2.2. Methods

2.2.1 Species data

I was opportunistic about which taxa to include in this study; a taxon was chosen if there were data available at the necessary resolution or if such data could be created through collaboration with experts. I was able to collate or assemble datasets for amphibians, birds, chameleons, mammals and plants (Table 2.1).

Table 2.1. Species' distribution model sources. For amphibians, mammals and plants, models derived by other authors were used. The source of models, Extent of Occurrence (EO) data and Habitat Suitability Index (HSI) scores are given. HSI scores were based on information about species' habitat preferences for particular environmental variables (column two). In collaboration with experts, EO and HSI data were used to develop Extent of Suitable Habitat (ESH) models for birds and chameleons.

	Taxon	Environmental Variables Used	Model Source	EO Source	HSI Scores
Published	Amphibians	Altitude, land cover, distance to water	1	2	-
	Mammals	Altitude, land cover, distance to water	3,4	2	-
	Plants	Mean annual temperature, temperature seasonality, annual rainfall, annual moisture index, dry season water stress and land cover	5	EO delimited by mountain blocs[6]	-
New	Birds	Altitude, land cover	This study	EO based on $\frac{1}{4}$ degree grid cells[7, 8, 9] or BirdLife[10]	11, 12
	Chameleons	Altitude, land cover	This study	Field Guide[13]	14

Sources: 1. Ficetola et al. (in prep.); **2**. IUCN (2010b); **3**. Rondinini et al. (2005); **4**. Rondinini et al. (in prep.); **5**. Platts (2012); **6**. Platts et al. (2011); **7**. Fjeldså (2007); **8**. Fjelsdå et al. (2010); **9**. Fjeldså and Tushabe (2005); **10**. BirdLife International (2008); **11**. J. Fjeldså (pers. comm.); **12**. L. Hansen (pers. comm.); **13**. Spawls et al. (2004); **14**. K. Howell (pers. comm.).

2.2.2 Species of conservation concern

I targeted species of conservation concern by only including in the analyses Threatened and Near-Threatened species, restricted-range species and endemic or near-endemic species (see Appendix A). Using species of conservation concern to direct conservation priorities is supported by a study by Drummond et al. (2010) in Indonesia, which found that conservation planning based only on threatened mammal distributions identified networks of reserves that sufficiently represent over 90% of non-threatened mammals species too. The definitions used to define species of conservation concern are described below; together they identified 504 species for inclusion in these analyses (57 amphibians, 76 birds, 14 chameleons, 41 mammals and 316 plants).

2.2.2.1 Threatened and near-threatened species

The first criterion for inclusion was threat status (IUCN 2001). All Near-Threatened, Vulnerable, Endangered and Critically Endangered species were included. Species-specific threat status was taken from IUCN (2010b) for amphibians and mammals and BirdLife International (2008) for birds. Neither chameleons nor plants have been fully assessed (although see Gereau (unpublished data) for work in progress on plants), so this information was unavailable for these taxa.

2.2.2.2 Restricted-range species

Range size was my second criterion for inclusion of a species. Restricted-range species have been shown to be at disproportionate risk of extinction and their global status is more affected by any given local threat (Purvis et al. 2000; Sodhi et al. 2008). Their inclusion is further justified by the coincidence of small range size and local rarity and the constraints on spatial configurations that their inclusion in a conservation plan dictates (Rodrigues et al. 2004b; Ceballos et al. 2005; Langhammer et al. 2007; Nicholson et al. 2009; Larsen et al. 2012).

To capture the range size that represents the most vulnerable species, different thresholds should be calculated for different taxa, as some species groups require larger areas on average (Gaston 1996). Stattersfield et al. (1998) define birds as restricted-range if their EO is less than 50,000 km^2, while Ceballos et al. (2005) give 24,000 km^2 as the lower quartile of mammals' EOs and the threshold for definition of a restricted-range mammal species. For amphibians, birds and mammals, where global datasets detailing the EOs for the majority of species exist, I calculate the lower quartile of species' ranges and use this to define the restricted-range threshold. I use 300 km^2 for amphibians (first quartile < 284 km2; IUCN 2010b), 82,000 km^2 for birds (first quartile < 81,734 km^2; Orme et al. 2006) and 22,000 km^2 for mammals (first quartile < 21,604 km2; IUCN 2010b). For chameleons, I classify species as restricted-range if their limited distribution is noted as a potential threat to their continued persistence by Spawls et al. (2004). For plants, comprehensive global distribution maps are unavailable, so range size could not be used as a criterion for their inclusion.

2.2.2.3 Endemic and near-endemic species

The final criterion for inclusion was for species that had been identified as endemic or near-endemic to the EAM. Near-endemic vertebrates are defined by Burgess et al. (2007c) as those that are only found in the Eastern Arc ecoregion and in one or more of the Northern Inhambane–Zanzibar Coastal Forest Mosaic, the Southern Rift Montane Forest-Grassland Mosaic and the East African Montane Forests (Burgess et al. 2004a). By definition, the EAM are a core part of these species' ranges and the chance of continued survival for these species is poor or nil if they are lost from here. Furthermore, Meuser et al. (2009) found that, out of a range of tools to prioritise taxa for conservation, endemism has most public support. Data on endemism were taken from Burgess et al. (2007c) for amphibians, birds and chameleons, while plant endemism was based on Gereau et al. (in prep.) and Platts (2012).

2.2.3 Published species data

Data for amphibians were obtained through the Global Amphibian Assessment (GAA; IUCN 2010b) and work by Ficetola et al. (in prep.). Elephant data are from the African Elephant

Database (Blanc et al. 2007), while other mammal distributions are from the Global Mammal Assessment (GMA; IUCN 2010b) and work by Rondinini et al. (2005; in prep.). Except for the elephant data, these amphibian and mammal models couple EO maps with known habitat preferences for land cover, altitude and distance to water to derive ESH models (Table 2.1). Plant distributions were provided by Platts (2012). These models were derived in a similar way to amphibian and mammal models (Table 2.1), but used point locality data to determine species' habitat preferences. The point locality data were used to extract information for six variables (mean annual temperature, temperature seasonality, annual rainfall, annual moisture index, dry season water stress and land cover), which are known to be important to the distribution of plants in the EAM (Platts et al. 2008; Platts et al. 2010). Only areas that fall within each of the six environmental envelopes that encompass these point localities were included as suitable habitat. These models were then constrained to just the mountain blocs in which they are reported. This was done for all plant species that are endemic to the mountain blocs (Platts et al. 2011; Gereau unpublished data) and for which there are records from two or more 1 km^2 pixels in the EAM (316 species). For all taxa, the information used represents the best available data for the region.

2.2.4 New models of extent of suitable habitat

I used a similar protocol to that used by Rondinini et al. (2005) to derive species' distribution maps for birds and chameleons. I developed EO maps for bird species from a quarter-degree resolution dataset on East African birds (Fjeldså and Tushabe 2005; Fjeldså 2007; Fjeldså et al. 2010). Where species were missing from this dataset, I used EOs published by BirdLife International (2008). For chameleons, distribution maps from the region's definitive field guide (Spawls et al. 2004) were digitised to produce EOs. Experts then determined habitat suitability scores for each land cover class (bushland, bushland with scattered crops, closed woodland, cultivation (including rice, rubber, sisal, sugarcane, tea and teak plantations), forest, forest mosaic, grassland, grassland with scattered crops, open woodland, permanent swamp, plantation forest, woodland with scattered crops) and each altitudinal band (≤ 300 masl; 300 – 500 masl; 500 – 1000 masl; 1000 – 1500 masl; 1500 – 2000 masl; 2000 – 2500 masl; 2500 – 3000 masl, ≥ 3000 masl) for every bird (J. Fjeldså, pers. comm.; L. Hansen, pers. comm.) and chameleon species (K. Howell, pers. comm.) of conservation concern. Distribution maps showing the modelled ESH were then validated by the experts. Each habitat type and altitude band was scored between zero (unsuitable) and five (ideal) for every species. The two scored layers were then overlain and the lowest score taken as the HSI for that location. Thus, the models assumed no compensatory relationship between altitude and land cover (Burgman et al. 2001); low suitability of one environmental variable was not offset by high suitability for the other.

2.2.5 Mapping species' distributions

The ordinal scale predictions of ESH models were converted to binary presence / absence maps by selecting the most suitable habitat within a species' range to represent presence (sensu Drummond et al. 2010). Thus, for amphibians and mammals, areas that were described as "high suitability" were assumed to indicate species' presence. Where there was no high suitability habitat available within the EO, I used the "low suitability" habitat to classify presence (one amphibian species). For seven amphibian species, their entire EO was classed as unsuitable. These species all had extremely small range sizes (EO size: median = 12 km^2) and the lack of suitable habitat represents misclassifications in the land cover layer. As I was unable to generate ESH models, I used the EOs to define species' presence. For birds and chameleons, the same logic was applied by classifying the most suitable habitat (HSI ≥ 4) as the AO. For eleven species of bird, there were no areas of high habitat suitability within the EAM. Therefore, the most suitable habitat within the EO was used for these species (for five species I used HSI ≥ 3 and for two species I used HSI ≥ 2). Plant models were provided as binary grids (Platts 2012), while elephant presence was identified by using the areas where the species is confirmed as present (Blanc et al. 2007).

As well as mapping species richness, an index of range-size rarity (also known as endemism richness) was mapped at the resolution of nine square kilometres to provide a continuous mapped surface that combines species richness with a measure of endemism. Range-size rarity was calculated as the sum of the inverse of the range size of every species present in a nine square kilometre cell and then the absolute values were rescaled to an index of between zero and one (Kier and Barthlott 2001).

2.2.6 Sampling bias

To assess the effect of taxonomic bias, richness for each of the five taxa was mapped alongside total richness, in order that the contribution of each taxon could be visualised. In addition, an index of richness for each taxon was mapped by dividing the richness of every pixel by the maximum richness for that taxon. Summing these for all five layers creates a map in which the contribution of each taxon is equal. Comparison of this richness index with the raw total richness enables examination of the effect of taxonomic group size. To investigate the effect of geographical sampling bias on distribution models, information on sampling effort was taken from Platts et al. (2010). This is based on total numbers of plant records per mountain bloc, which serve as a proxy for sampling effort (the number of records should capture information on number and length of surveys, as well as man hours spent surveying).

2.2.7 Reserve selection and targets

The conservation planning software Marxan v1.8 (Ball and Possingham 2000) was used to run the SCP analyses, with the following settings: algorithm, simulated annealing; number of simulations, 1000; number of iterations per simulation 1,000,000; number of temperature decreases per simulation, 10,000; choice of initial temperature and cooling, adaptive. More information on how Marxan generates spatial priorities and the settings and planning units used is given in chapter one. Setting targets for conservation is a crucial step in conservation planning and can influence solutions dramatically. Often a percentage of species' ranges are used and, although arbitrary, this is a pragmatic and transparent solution where data on minimum viable populations and species-specific home range sizes or population dynamics are unavailable (Rodrigues et al. 2004b; Ardron et al. 2008; Drummond et al. 2010). Drummond et al. (2010) use a target of 30% of the species' AO, while Rodrigues (2004b) use a target of 100% of a species' EO for those whose EO is less than 1,000 km^2 and 10% for those whose EO is greater than 250,000 km^2 with intermediate range sizes receiving targets interpolated between 10% and 100%. Rondinini et al. (2005) use a similar logic, giving a targets of 100% to species with an AO of less than 1,000 km^2, 10% to species with an AO of greater than 10,000 km^2 and 1,000 km^2 to species whose AO falls between these extremes. Although Rodrigues et al. (2004b) and Rondinini et al. (2005) use a similar threshold for their smallest ranging species, those of the latter study are based on AO, whereas those of the former study are based on EO. In this study, targets followed a similar logic, but, as they are based on AO rather than EO, I reduced the thresholds by an order of magnitude. Thus, species with an AO of less than 100 km^2 received a target of 100%, those with an AO of 100 km^2 to 1,000 km^2 received a target of 100 km^2 and those with an AO of greater than 1,000 km^2 received a target of 10% (Figure 2.1). I tested the targets used by Rondinini et al. (2005; 100% for species with an AO of less than 1,000 km^2, 10% for species with an AO of greater than 10,000 km^2 and 1,000 km^2 for species whose AO falls between these extremes), but this resulted in a very inflexible solution, as most species required all of their AO to be conserved (Figure 2.1). The thresholds at which a species' target is met was set at 95% and conservation feature penalty factors (the penalty applied to the cost of the solution if a conservation target is not met) of 0.1, 1, 10 and 100 were tested (see chapter one). Setting the penalty factor to ten ensured that all species met their targets and this penalty factor was used throughout. Irreplaceability is a measure of the importance of a planning unit for meeting conservation targets and was calculated as the percentage of times (out of 1000 runs) that a planning unit was included in a hypothetical reserve network. Analyses were run in three ways: First, using square planning units of nine square kilometres, which included no information on current protection status. Second, a gap analysis was conducted, in which current protected areas were included to identify the areas most complementary to the existing reserve network. Third, protected areas were retained as contiguous planning units, but they were not necessarily kept in the final solution if they did not contribute sufficiently to the conservation targets (i.e. the entire protected area was either included or excluded from the solution). This provides information on each reserve's irreplaceability.

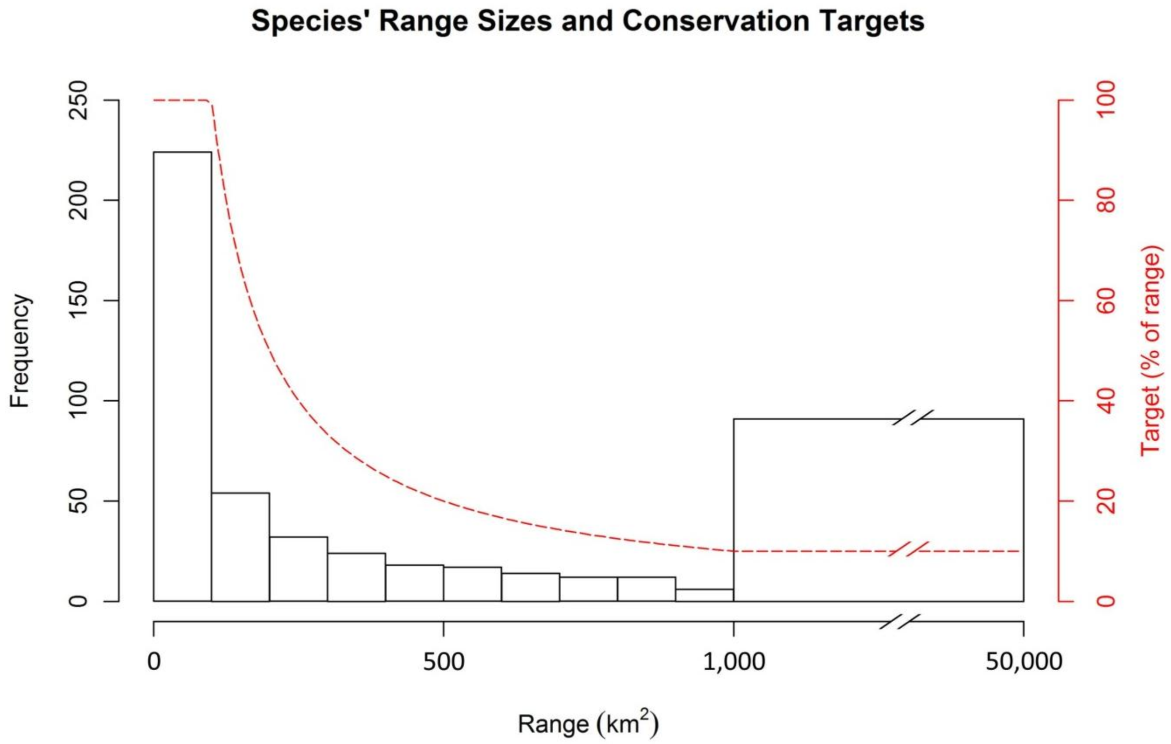


Figure 2.1. Histogram showing the frequency distribution of range sizes for species used in these analyses. The target (as a percentage of the range size) is also plotted (dashed red line; right hand y-axis).

2.2.8 Non-species data

Rather than using data on the cost of conservation, I used planning unit area. The conservation planning software, therefore, attempted to maximise biodiversity for a minimum area. In developing a useful conservation plan, the analysis should also incorporate spatially explicit information on the ability of a protected area network to ensure the long-term persistence of species and data on the actual cost of a reserve network. Whilst recognising the importance of these data (explored in detail in chapters three to six), here I wanted to explore priorities for conservation based solely on information on species' patterns. Therefore, no attempt was made to include data on biological processes that promote long-term species' persistence or on costs. This extends to the fact that I did not include a boundary length modifier, which would encourage aggregation of the reserve network solution and should act both to promote more viable species' populations and to decrease management costs per unit area (Possingham et al. 2000).

Data on the current protected area system were taken from the World Database on Protected Areas (WDPA; IUCN 2010a), modified to include recently designated nature reserves (MNRT 2010). The land cover map and digital elevation model are products of the Valuing the Arc project (Burgess et al. 2009) and are described in Platts et al. (2011).

2.3. Results

2.3.1 Refinement of species' distribution maps

Where data on the size of EOs were available (all amphibians, birds and mammals), the models suggested that a species occupies around 23% of its EO (mean ± 1 s.d. = 24 ± 30.5%, n = 174). The refinements to a species' EO were most pronounced in birds, where the average modelled ESH represented 5% of a species' EO (mean ± 1 s.d. = 5 ± 10%, n = 76), while the average ESH of amphibians and mammals represented 55% and 16% of species' EOs respectively (amphibian: mean ± 1 s.d. = 55 ± 29.8%, n = 57; mammal: mean ± 1 s.d. = 16 ± 20.3%, n = 41).

2.3.2 Species richness

Species richness was generally highest in the East Usambara, West Usambara, Uluguru, and Nguru Mountains and on the eastern flanks of the Udzungwa Mountains (Figure 2.2). Mammals of conservation concern, with at least one species present throughout the Arc, are noticeably less restricted in their distributions than other taxa. However, the patterns shown exhibit a degree of taxonomic bias, due to the fact that some groups are better represented than others. For instance, chameleon richness is based upon 14 species (Figure 2.2d), while plants are based upon 316 species (Figure 2.2f). Therefore, I compared total richness with an index of richness where each taxon contributes equally (Figure 2.3). Although the forests are given higher priority in the unweighted map (due to the influence of plants, which are confined to forest), the pattern remains remarkably similar.

In addition, the data are geographically biased by the fact that some areas have been more extensively surveyed, resulting in a more complete description of their biodiversity (Platts et al. 2010; Ahrends et al. 2011). Using data from Platts et al. (2010) on botanical surveying effort, I plotted number of species of conservation concern against sampling effort (Figure 2.4). There is a clear and significant positive relationship between a taxon's richness in a mountain bloc and the number of botanical surveys that have been conducted there. This is true for plants (R^2_{adj} = 0.71; Figure 2.3b, P = 3.3*10-4; n = 12 blocs; Platts et al. 2010), but also holds for other taxa, for which botanical surveying effort is a less direct proxy (amphibians: $R^2_{adj} = 0.66$, $P = 8.1*10^{-4}$; birds: $R^2_{adj} = 0.78$, $P = 8.9*10^{-5}$; chameleons: $R^2_{adj} = 0.37$, $P = 0.021$; mammals: $R^2_{adj} = 0.44$, $P = 0.011$; n = 12 for all tests; Figure 2.4b). The degree of to which the species in a mountain bloc are found nowhere else (single-bloc endemics) is a crucial measure of the blocs' importance as, for these species, spatial options for their conservation lie entirely within that bloc. Table 2.2. shows how information on endemism can influence priorities. For example, the East and West Usambara Mountains show similar richness, but the East Usambaras have over twice the number of single-bloc endemics. I used a range-size rarity index to map endemism richness across the EAM at a resolution of nine square kilometres (Figure 2.3c; Kier and Barthlott 2001). This map retains broadly similar patterns to those of species richness, but areas in the Mahenge Mountains and in the west of the Rubeho, Udzungwa and Ukaguru Mountains exhibit higher priority than when richness alone is used (Figure 2.3b).

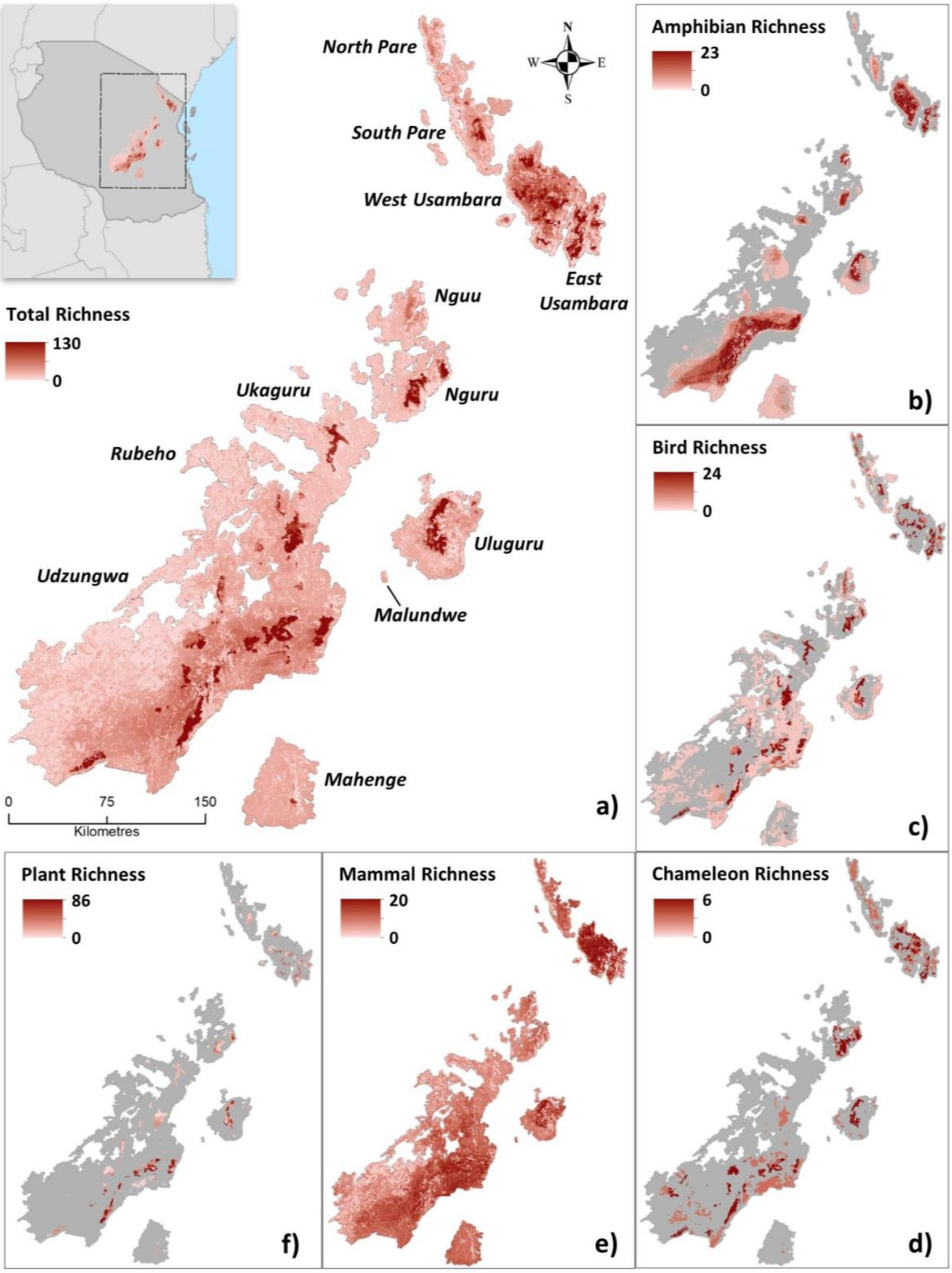


Figure 2.2. Distribution models for 504 species of conservation concern are used to map species richness in the Eastern Arc (a). Darker red indicates greater numbers of species and mountain bloc names are labelled in bold. Modelled richness is also shown for each taxonomic group (b-f). Grey areas within the mountain blocs indicate that no species of conservation concern are modelled as present for that taxon.

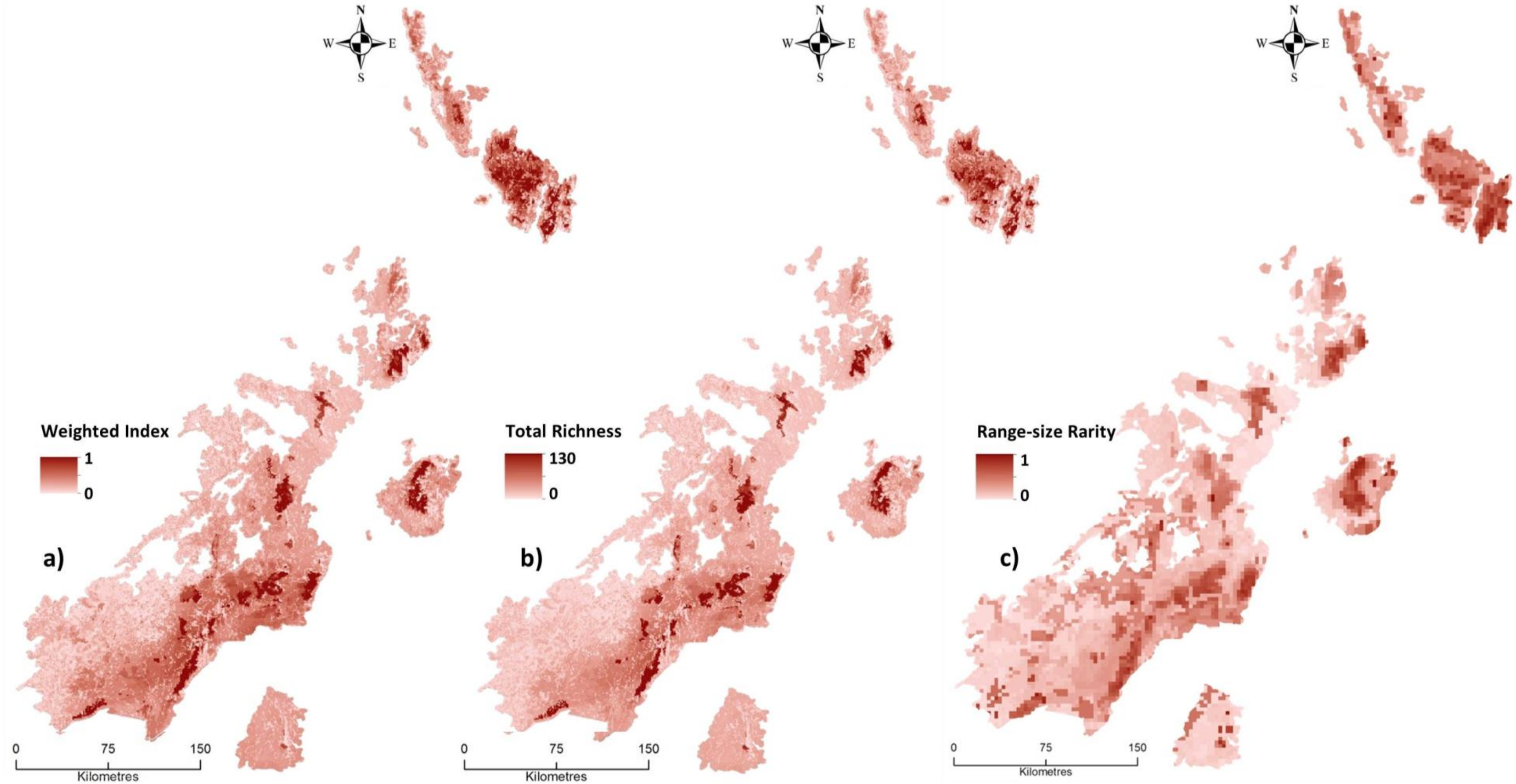


Figure 2.3. The effect of taxon size and endemism upon species richness patterns in the Eastern Arc Mountains. Although there is a large difference between the number of species represented by each taxonomic group (e.g. chameleons = 14 and plants = 316), when each taxon is given equal weighting (a), the pattern remains very similar to that obtained when unweighted richness is mapped (b). Forests show greater richness in the unweighted map due to the fact that plants, which are the largest group, are confined to forest. When each taxon is weighted equally, woodlands and grasslands show increased priority. An index of range-size rarity is also mapped (c). This measure couples species richness with endemism to demonstrate irreplaceability and emphasises the importance of areas in the Mahenge Mountains and in the west of the Rubeho, Udzungwa and Ukaguru Mountains, despite these areas not showing relatively low species richness.

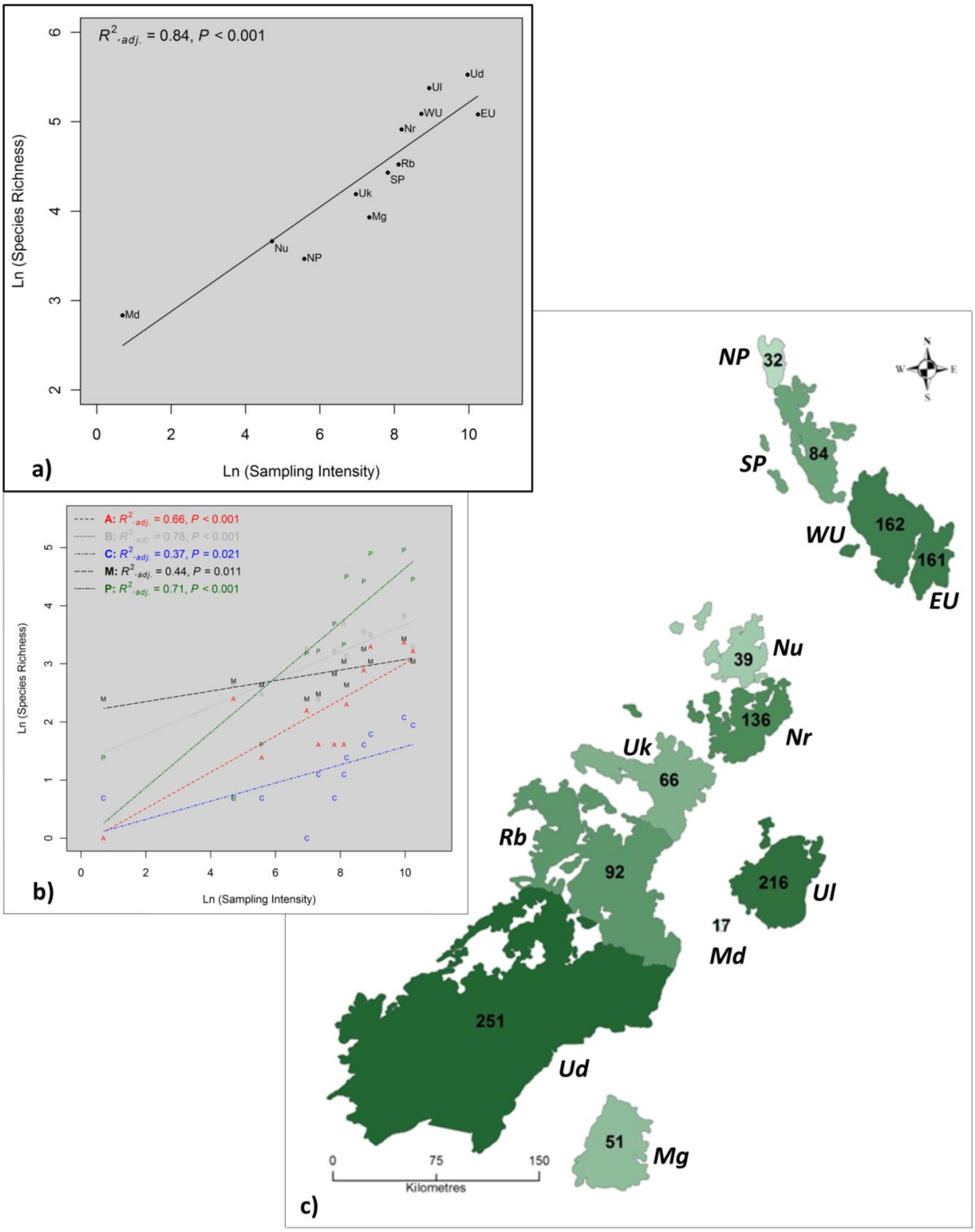


Figure 2.4. Relationship between modelled species richness and sampling effort. The natural logs of total number of species of conservation concern (a) and of number of species of conservation concern for each taxon (b) are plotted against the natural log of sampling effort per mountain bloc. The number of species of conservation concern per mountain bloc is also mapped (c), with darker green indicating higher richness (absolute number given in bold). Mountain bloc abbreviations (graph a and map c): EU, East Usambara; Mg, Mahenge; Md, Malundwe; NP, North Pare; Nr, Nguru; NU, Nguu; Rb, Rubeho; SP, South Pare; Ud, Udzungwa; Uk, Ukaguru; Ul, Uluguru; WU, West Usambara. Taxon abbreviations (graph b): A, Amphibian; B, Bird; C, Chameleon; M, Mammal; P, Plant.

Table 2.2. Species richness and endemism, summarised for each mountain bloc, shows the relative biodiversity importance and uniqueness of each.

	Species Richness						Single-bloc Endemics					
Mountain Bloc	**Total**	**Amphibians**	**Birds**	**Chameleons**	**Mammals**	**Plants**	**Total**	**Amphibians**	**Birds**	**Chameleons**	**Mammals**	**Plants**
North Pare	32	3	11	1	13	4	0	0	0	0	0	0
South Pare	84	4	24	1	16	39	8	0	2	0	0	6
West Usambara	162	17	34	4	25	82	17	3	0	0	1	13
East Usambara	161	24	26	6	20	85	37	5	2	2	1	27
Nguu	39	10	13	1	14	1	0	0	0	0	0	0
Nguru	136	9	22	3	13	89	12	0	0	0	0	12
Ukaguru	66	8	25	0	10	23	9	4	3	0	0	2
Uluguru	216	26	32	5	20	133	54	5	6	1	2	40
Malundwe	17	0	3	1	10	3	0	0	0	0	0	0
Rubeho	92	4	39	2	20	27	4	1	1	0	0	2
Udzungwa	251	28	45	7	30	141	60	9	5	3	6	37
Mahenge	51	4	10	2	11	24	4	0	1	0	0	3

2.3.3 Current protection

Under the current 10,540 km^2 system of reserves (which cover 21% of the 50,800 km^2 of the EAM), a mean of 66% of species' modelled AOs are conserved (median = 75%, *n* = 504; Appendix A), and a mean of 225% of their targets are conserved (median = 103%, *n* = 504; Appendix A). The targets for 224 species (44%), however, are not met in the current reserve system and, for 56 species (11%), less than 50% of their target is included.

2.3.4 Reserve selection

Initially, the SCP analysis was run using square planning units of nine square kilometres. This analysis did not include protected areas as planning units and was, therefore, not constrained by the current reserve network. The mean solution from 1,000 runs identified 10,473 km^2 which met the conservation targets for all 504 species (mean area ± 1 s.d. = 10,473 ± 31 km^2, *n* = 1000). As expected, the Usambara, Uluguru and Udzungwa Mountains show high irreplaceability (Figure 2.5a). Furthermore, the Mahenge, Nguru, Pare and Ukaguru Mountains also all exhibit areas of high irreplaceability, which is strongly influenced by endemism, as well as by total richness (Figure 2.3c and Table 2.2).

The second analysis used protected areas as planning units, but allowed them to be removed from the solution. This resulted in a larger reserve network (mean area ± 1 s.d. = 15,762 ± 26 km^2, *n* = 1,000) and helped identify the contribution of individual reserves to meeting the conservation targets of species used in these analyses (Figure 2.5b).

In implementation of any conservation planning exercise, areas identified as high priority are most likely to be added to an existing reserve network, rather than wholly replacing the current network. The final analysis was, therefore, constrained to always include current protected areas. This allows the software to identify the areas most complementary to the current system of reserves (Figure 2.5c). This resulted in an expansion of the current network by 63% from 10,540 km^2 to 17,203 km^2 (mean ± 1 s.d. = 17,203 ± 33 km^2, *n* = 1,000) to meet the targets for all 504 species. The analysis highlights the importance of currently unprotected areas in the West Usambara and Udzungwa Mountains, which both demonstrate high irreplaceability, yet are not currently within the reserve network.

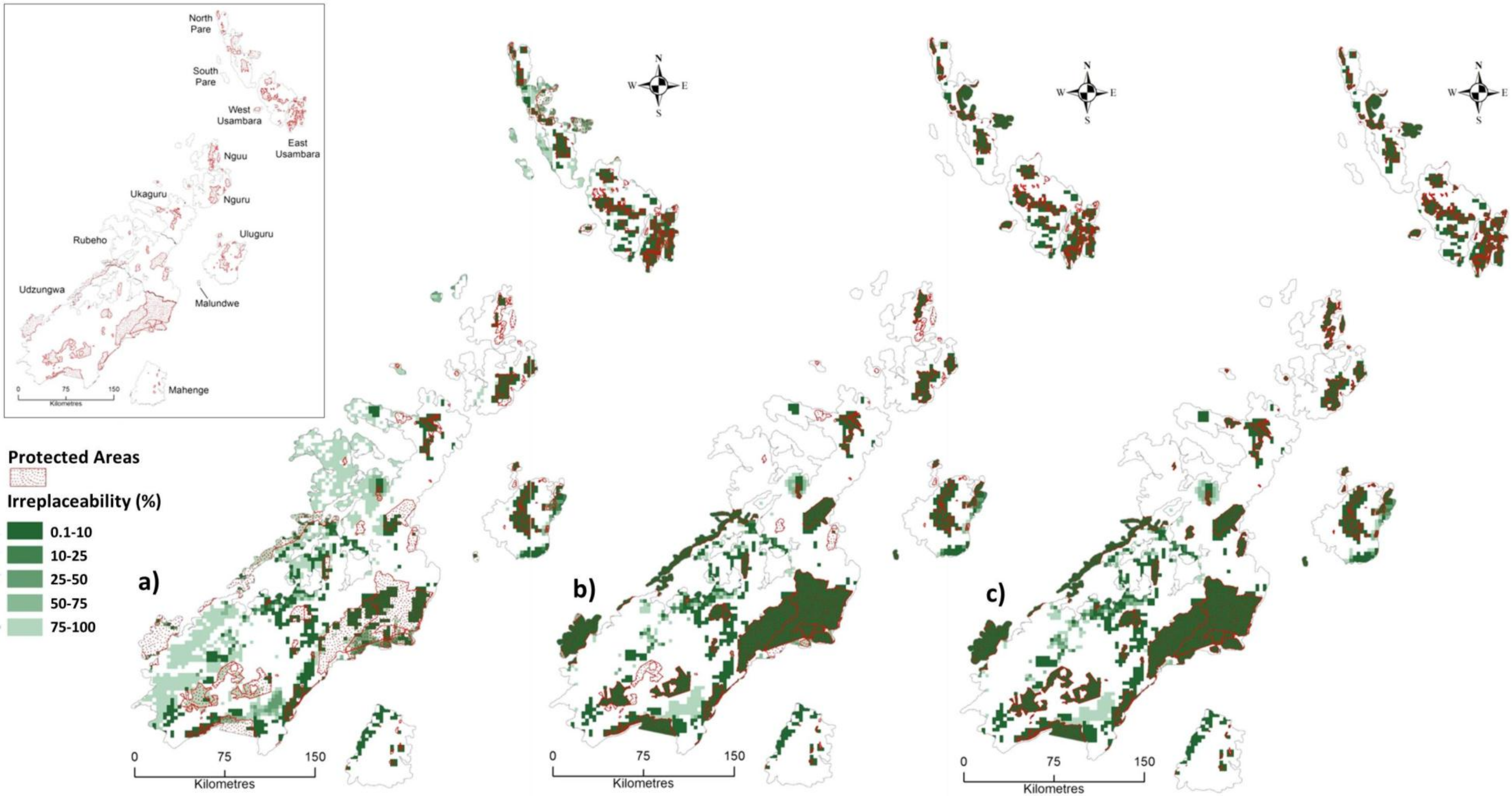


Figure 2.5. Irreplaceability in the Eastern Arc Mountains. The unconstrained solution shows the selection frequency of 9 km^2 planning units, for which all units could be selected or removed by the programme (a). In the second scenario (b), current protected areas are used as planning units but can be removed from the solution. This helps highlight the relative importance of individual reserves within the current reserve network. Lastly, a constrained solution is also presented (c), in which currently protected areas are locked into the solution and the programme identifies those planning units that are most complementary to the current reserve network.

2.4. Discussion

2.4.1 Species' distributions

Species richness and range-size rarity maps (Figure 2.2) clearly illustrate the spatial variation in biodiversity value found across the EAM. Moist winds coming from the Indian Ocean and depositing their moisture onto the easterly slopes of the mountain blocs support the area's characteristic ecoclimatic stability and biodiversity (Fjeldså and Lovett 1997a; Platts et al. 2010). This pattern is apparent in these data – the most comprehensive set that have been compiled at such fine resolution. The Uluguru and Usambara Mountains, which lie to the east show particularly high diversity, along with the eastern flanks of the Udzungwa and Nguru Mountains. Apart from species richness, range-size rarity and irreplaceability have both been mapped, providing useful measures for conservationists assessing the uniqueness of the area's biota. Although the Nguu Mountains show relatively high diversity for amphibians, these species are not unique to that mountain bloc, so conservation priority that accounts for irreplaceability is shifted to other planning units which contain high diversity of other taxa too.

The current reserve network captures many of the highly irreplaceable areas in the EAM (Figure 2.5), reflecting the fact that much conservation planning, to date, has been based on information on species' patterns (Lovett and Moyer 1992; Burgess et al. 2007c; MNRT 2010). In complementing the existing reserve network, efforts should focus on the West Usambara and Udzungwa Mountains, which show many currently unprotected areas of high irreplaceability. Although the average size of the reserve network when unconstrained by current protected areas is just 69% of the solution when currently protected areas are forced in (Figure 2.5a and c), this is not necessarily an inefficiency of the current reserve network. The current system caters for species besides those which are targeted here and species' distributions have not been the only consideration when this network of protected areas was created. An important goal of conservationists working in the EAM has been to link protected areas via corridors (World Bank 2006; Jones et al. 2007; Epps et al. 2008; Jones et al. 2009a; Jones et al. 2009b). Although these corridors may not represent core habitat for threatened species, their function is to promote long-term persistence through enabling species dispersal and migration. Therefore, the apparent inefficiency of the current system, when compared to the unconstrained solution is, in part, an artefact of the intentionally limited scope of the present analysis.

The priority set identified here is broadly similar to that identified in previous studies at lower resolutions and highlights the importance of the Uluguru, Udzungwa and Usambara Mountains (Fjeldså and Tushabe 2005; Fjeldså 2007; Fjeldså et al. 2010). However, the high resolution data compiled here also indicate the importance of the Pare, Nguru and Ukaguru Mountains, which was not so clear in previous studies (Figures 2.2., 2.3, and 2.5. and Table 2.2). Although these previous studies, at coarser resolutions, have helped identify important areas for conservation, they have not, to date, been sufficiently resolved to guide conservation implementation at a local level.

2.4.2 Caveats

2.4.2.1 Sample bias

Species distribution data can be spatially, temporally or taxonomically biased and are often biased in all three aspects (Rondinini et al. 2006). This can be seen in the data used here: only five taxonomic groups are used due to data availability and much of the data for the Usambara and Uluguru Mountains were collected pre-1980, while the Udzungwa Mountains have been surveyed more recently (Platts et al. 2010). Several of the mountain blocs exhibit apparently low species' richness (e.g. Nguu; Figure 2.4, Table 2.2) and irreplaceability scores, yet this is likely to be largely due to the fact that the bloc is undersampled (Platts et al. 2010). In particular, the lack of threatened chameleons in the Ukaguru and northern Udzungwa Mountains (Figure 2.2d) is likely to be down to gaps in our knowledge, rather than a lack of threatened species (J. Fjeldså, pers. comm.). However, such an effect is difficult to prove and the link between survey effort and richness (Figure 2.4) could, in fact, be driven by initial surveys of some areas finding few species or less interesting ecosystems that were consequently less intensively surveyed than areas that showed greater promise during early surveys and received greater subsequent interest (Fjeldså 2003; Fjeldså and Tushabe 2005).

Using predicted, rather than observed, occurrence data reduces spatial effects of sampling bias to some extent (Platts et al. 2010). All possible taxa are used in this assessment, but it was not possible to account for the bias towards larger species. However, using as many taxa as possible is the best way to approximate the true biodiversity of an area and minimise the way in which taxonomic bias spatially influences priorities for conservation (Margules et al. 2002; Rodrigues and Brooks 2007; Platts et al. 2010). The models used are also dependent upon the quality of the variables from which they are inferred. Some taxa are likely to be able to utilise habitat patches that are too small to be identified by a regional land cover map or elevation model used (Rondinini et al. 2006). This is particularly the case for amphibians, for which the modelled AO will likely have higher omission errors than for other taxa. However, this trade-off tends towards a conservative conservation plan and increases the confidence that a species will actually occur where it is modelled (Rondinini et al. 2005; Rondinini et al. 2006).

2.4.2.2 Targets

The choice of targets is a subjective, albeit transparent, element of SCP. The targets chosen here reflect those used commonly in the literature (e.g. Rondinini et al. 2005). Although targets are a fundamental driver of the SCP software, Marxan, other authors have noted that solutions are often only marginally sensitive to their size. This is because most species are either common and, therefore, included incidentally when planning units are selected for other species, or species are range-restricted and all of their distribution is conserved when a planning unit is selected for their conservation (Figure 2.1). Thus, species' representation tends to exceed targets, causing the insensitivity noted (Warman et al. 2004; Rondinini et al. 2005).

2.4.2.3 Implementation

There are several other important caveats to the conservation plan described here. These plans are based on modelled distributions and, although a conservative approach has been taken, sites should be carefully assessed to make sure that they do actually contain the species predicted to occur there. Species of conservation concern are hardest to model as they often have few records from which to base models or expert opinion and they are often highly specialized to particular environmental conditions which may not be captured by existing environmental datasets (Jetz et al. 2004; Rondinini et al. 2005; Platts et al. 2010). The most important limitation to these analyses, however, is that only the biodiversity benefits, which form the backbone of many SCP analyses (Ardron et al. 2008), are described. The analyses specifically exclude information on the cost of the planning units as well as considerations of long-term species' persistence. I go on to explore how priorities change when these extra considerations are taken into account in subsequent chapters. Moreover, before beginning implementation, conservationists should consider threat, opportunity and willingness to conserve for the areas identified (Knight et al. 2009; Knight et al. 2011). There is no utility in reserving a site that will be lost to an immitigable threat and, equally, there may be an unforeseen opportunity to protect a site that was previously unavailable.

2.5. Summary

Systematic Conservation Planning (SCP) relies on accurate data representing species' occurrence across the landscape. Here several published datasets are combined with newly created maps to provide the first SCP exercise for the region based on fine-scale data for species of conservation concern in five taxa: amphibians, birds, chameleons, mammals and plants. Based on the data in this chapter, an expansion of the current network by 63%, particularly in the Udzungwa and West Usambara Mountains would help ensure conservation for 504 species of conservation concern. Although some obvious taxonomic and geographic biases in these data remain, it is important to remember that refinements in biodiversity data only address one side of the equation for developing efficient conservation plans and improving species' distribution models does not necessarily address the issue of ensuring long-term species' persistence. Work to improve our knowledge of poorly known or undiscovered species will continue, but the trade-off between time spent gathering more data and implementing an effective conservation plan rapidly can be better addressed by working to incorporate other important biological and socio-economic data. Such considerations are investigated in the following chapters.

3. Ecological and evolutionary processes

"What escapes the eye, however, is a far more insidious kind of extinction: the extinction of ecological interactions"

Daniel H. Janzen (1974)

3.1. Introduction

3.1.1 Ecological and evolutionary processes in conservation planning

Ecological and Evolutionary Processes (EEPs), from intra-specific interactions between individual organisms through to species' responses to macroclimatic gradients, both generate and maintain the biodiversity of the past, present and future. They include the features of an ecosystem that promote the generation and long-term persistence of species under current environmental conditions and in the face of external challenges. They all operate through both space and time, but at varying scales (Lagabrielle et al. 2009). For instance, speciation can occur over a relatively small area, but might take tens of thousands of years, while migration can cover tens of thousands of kilometres in a matter of days.

Systematic Conservation Planning (SCP) is a spatially explicit technique to identify a near-optimal reserve network configuration that meets representation targets for a given set of species or other conservation features (Margules and Pressey 2000). Conservation targets in SCP exercises tend to be based on species' distributions - finding, for example, the smallest reserve network containing each species of conservation concern. However, using pattern to determine priority sets may not be enough to ensure the long-term persistence of species and the maintenance of self-sustaining and functional ecosystems (Balmford et al. 1998; Cowling et al. 1999; Cowling and Pressey 2001; Pressey et al. 2003; Rouget et al. 2003b; Burgess et al. 2006; Rouget et al. 2006; Mouillot et al. 2008; Klein et al. 2009; Lagabrielle et al. 2009); at the very least, the spatial coincidence of pattern and process should be investigated.

SCP requires the use of parcels of land, usually called planning units (PUs), in which conservation features, and the cost of bringing them into a reserve network, are quantified (Ball et al. 2009; Wilson et al. 2009). These units are then selected at random for inclusion or exclusion from a hypothetical reserve network to make iterative improvements towards a solution that meets all the conservation targets for a near-minimum cost or area (see chapter one); alternatively maximum representation of conservation features for a fixed cost or area can be computed. As spatial data on species pattern and cost become more finely resolved, ever-smaller PUs can be used to generate more efficient reserve networks. This efficiency will often correspond to decreased total area, putting processes that operate across large areas at increased risk.

Despite the importance of EEPs to biodiversity conservation, techniques to incorporate them in SCP are not well developed and they are rarely specifically included (e.g. Ando et al. 1998; Kati et al. 2004; Naidoo and Adamowicz 2006; Wilson et al. 2009; Fuller et al. 2010).

This is due to at least four factors: First, it is difficult to identify where EEPs and their proxies, by which they can be mapped, occur. Related to this is the second factor: even when spatial proxies are identified, it is still difficult to validate subsequent maps showing important areas for EEPs because of the large temporal and spatial scales over which these processes operate. Data are generally unavailable at such large scales, so validation is rarely possible. Third, they are difficult to define quantitatively because they are rarely fully understood, they are often an emergent property of several other biological or geographical components and they are dynamic in time and space. Last, optimising the efficient conservation of EEPs is often challenging. For instance, although there is plenty of information on how corridors should be created and how their paths may be optimised (e.g. Rosenberg et al. 1997; Chetkiewicz et al. 2006; Cushman et al. 2009; Conrad et al. 2012), the problem of optimising a reserve network to include multiple connectivity pathways (each of which might have several alternative routes), whilst also optimising for biodiversity representation targets, is much more complex (although see Carroll et al. 2010).

3.1.2 Previous studies

What proxies have been incorporated in previous work to integrate EEPs into SCP? Interfaces between major habitat types have been a particular focus for EEP conservation, as the greater diversity in environmental conditions and species assemblages are expected to drive ecological diversification, so maintaining evolutionary processes (Cowling and Pressey 2001; Lagabrielle et al. 2009). These include oceanic-terrestrial interfaces, edaphic interfaces and interfaces between biomes and habitat types (Cowling et al. 1999; Cowling and Pressey 2001; Rouget et al. 2003b; Rouget et al. 2006; Lagabrielle et al. 2009). Similarly, geographic and climatic gradients are identified as important proxies for the conservation of EEPs in Africa and Europe both because steep environmental gradients are, again, expected to drive ecological diversification and for their contribution to species' ability to respond to changing environmental conditions (Cowling et al. 1999; Cowling and Pressey 2001; Rouget et al. 2003b; Rouget et al. 2006; Lagabrielle et al. 2009; Carvalho et al. 2011). Some studies have also attempted to integrate geographic features into SCP that allow for migration of nutrients, soils, species and water (Cowling et al. 1999; Cowling and Pressey 2001; Rouget et al. 2003b; Rouget et al. 2006; Klein et al. 2009; Lagabrielle et al. 2009; Carroll et al. 2010). Finally, particular continuous units, such as mega-wilderness areas and river catchments are sometimes selected for the EEPs that they are expected to host, such as species' movements, drought refugia and viable populations of large mammals (Cowling et al. 1999; Klein et al. 2009; Lagabrielle et al. 2009).

The focus on these kinds of proxies emerges from adopting a process-specific approach to EEP conservation - targeting a particular EEP and then identifying a proxy to represent it.

However, more generic tools within SCP are also available. These include using PUs that are large enough to include ‘incidentally’ the processes which are considered important (Pressey et al. 2003; Klein et al. 2009); preferentially aggregating PUs through the use of a modifier, which penalises reserve systems with higher edge/area ratios (e.g. Ball and Possingham 2000; Moilanen and Wintle 2007; Ardron et al. 2008); and adjusting the final solution so that small reserves have their area increased to meet some minimum size requirement efficiently (Smith et al. 2010). This final method differs from the previous two in that it does not result in the smallest reserves becoming uniform in size and shape, as is the case if the minimum size is specified by the size of the planning unit, and it only applies to reserves that fall below a size threshold, so does not suffer from the fact that all reserves are modified, as is the case when a modifier is used to penalise higher edge/area ratios. Such generic design criteria are favoured due to their simplicity, but they are applied to the entire set of reserves, no matter whether there is any identified need for EEP conservation in that area. To avoid this problem, and because the central tenets of SCP are to operate in an accountable, transparent and efficient manner, in this study only process-specific proxies are used.

Studies have sometimes considered anthropogenic threat alongside EEPs because they are also difficult to quantify, dynamic in time and space and can often be expected to disrupt EEPs (e.g. Rouget et al. 2006; Pressey et al. 2007; Lagabrielle et al. 2009). Threats to biodiversity can be dealt with in three ways: by targeting EEPs, such as dispersal, that allow species to respond to threats; by basing species’ prioritisation upon threatened species or other threatened conservation features; or by using cost constraints within SCP to limit exposure to threats, which works because, where options are available, the software will select areas of low cost/value, which will often correlate with low threat. Threats are not dealt with solely in this chapter, but are included in other chapters using the latter two methods: basing conservation targets upon threatened species (previous chapter) and using cost as a proxy for threat (chapters five and six).

3.1.3 Study area

The Eastern Arc Mountains (EAM) contain a wealth of rare and endemic species (Myers et al. 2000; see chapters one and two for a detailed description of their location and importance to biodiversity conservation; Mittermeier et al. 2004; Burgess et al. 2007c). This biodiversity is both a consequence and a component of the region’s EEPs. However, the mountains and surrounding lowlands are also home to many people. Coincidence of human populations with high biodiversity is perhaps no surprise (Balmford et al. 2001; Burgess et al. 2007b; Fjeldså and Burgess 2008; Platts et al. 2011); the long-term climatic stability associated with EEPs in the region may also provide societal benefits, such as more predictable water supplies, which will draw human societies to the same areas (Fjeldså and Lovett 1997b). This relationship does, however, mean that EEPs are likely to coincide with high anthropogenic pressure and merit increased attention from conservation planners.

There are many EEPs operating within the EAM, but Pressey et al. (2003) present pragmatic criteria for inclusion in conservation planning. The EEPs included should be ones that we know about, that we understand well enough to map, and that operate over the ‘meso-scale’ scale upon which we can act. For instance, pollination, especially for small species, is often included within PUs incidentally when targets for species’ representation are set (Pressey et al. 2003), thereby avoiding the need for explicit consideration. On the other hand, the moist winds that blow from the Indian Ocean depositing orographic rainfall on the EAM and generating their characteristic climatic stability (Lovett 1985) cannot be included, as they operate beyond the scale at which regional conservation plans can be expected to exert an influence. Therefore, conservation planning is best focused upon meso-scale EEPs, which can be included in regional conservation priorities, but need explicit consideration to be so (Pressey et al. 2003). Meso-scale is a relative term, identifying processes that potentially operate over areas larger than a single PU, but within the overall scope of the planning area. Hence in any given planning area using smaller PUs will generate more meso-scale EEPs which will need to be considered explicitly for inclusion. In this study PUs of 9 km^2 are used because this is the median size of protected areas in the study districts and, if effectively managed, is expected to be large enough to provide viable conservation areas. Although these relatively small PUs make EEP conservation harder, the units are at a scale relevant to the practical decision support for which the exercise is intended.

3.1.4 Ecological and evolutionary processes in the Eastern Arc

Because SCP is an inherently transparent discipline, the assumptions about which processes are included and how they are mapped should be clearly laid out. Consultation with experts working in conservation within the EAM and a review of the literature led to the identification of 11 EEPs that are considered important within the mountains. These EEPs, and their proxies, are summarised on an approximately temporal scale in Table 3.1 and are described in detail here.

Table 3.1. The Ecological and Evolutionary Processes (EEPs) expected to occur in the Eastern Arc Mountains and the proxies which might be used to map them spatially. In square brackets are those processes which enable mitigation of human-induced threats. In bold are the EEPs and proxies addressed in this exercise. Processes are listed approximately on a temporal scale, with those occurring over the shortest timescale first and those occurring over geological timescales last.

Process	Spatial Proxies	Notes
Diurnal altitudinal migration/movement	**Forested lowland / upland gradients**	Birds utilise forested altitudinal gradients daily (J. Fjeldså pers. comm.).
	Riverine corridors	Birds move along riverine corridors daily (Lagabrielle et al. 2009).
[Resilience to nest predation]	Areas of low fragmentation	Fragmentation, via an edge-effect, could raise levels of nest predation lowering bird survival. However, there are inconsistent findings globally and within the EAM (Carlson and Hartman 2001; Lahti 2001; Newmark and Stanley 2011).
Pollination/seed dispersal	**Forested lowland / upland gradients**	Pollinators/dispersers follow asynchronous flowering across altitudinal gradients (Lagabrielle et al. 2009).
	Minimum patch size	Minimum patch should be > 30 ha; in larger planning units, the process will be conserved incidentally (Cordeiro and Howe 2001; Cordeiro and Howe 2003).
	Areas of low human population density	Bushmeat hunting is highest in areas of high population pressure (Nielsen 2006; Henschel et al. 2011), and this affects higher trophic levels and seed dispersal (Vanthomme et al. 2010; Henschel et al. 2011).
Seasonal migration	**Large mammal corridors between major habitats**	Corridors are fundamental to large mammal persistence and to maintaining intermediate levels of disturbance generated in the habitats that they utilise. The corridors have been mapped in detail (Jones et al. 2007; Epps et al. 2008).
	Forested lowland / upland gradients	Birds undertake seasonal altitudinal migration through forests (Stuart 1983; Burgess and Mlingwa 2000; Newmark 2002 p137; Lagabrielle et al. 2009).
Disturbance	**Elephant corridors between major habitats**	Current routes between major protected areas have been described in detail (Jones et al. 2007; Epps et al. 2008; Jones et al. 2009b). Elephant presence maintains disturbance and succession regimes and promotes niche diversity (Pringle 2008; Whyte et al. 2008). Their distribution has been mapped throughout Africa (Blanc et al. 2007).
[Resilience to fire]	Areas of low fragmentation	Edges dry out faster (due to more wind) and are more prone to accidental and deliberate fires (S. Madoffe pers. comm.; R. Temu pers. comm.).
	Areas of high moisture availability	Within forest, moisture availability is likely to predict resilience to fire (S. Madoffe pers. comm.; R. Temu pers. comm.).
[Resilience to invasive species]	Areas more than 1.5 km from reserve boundary	The number of invasive plants declines to a background level within protected areas at distances of >1.5 km from the boundary (Foxcroft et al. 2011).
	Areas with low road density	Road density is an important predictor of plant invasions (Foxcroft et al. 2011).

[Resilience to land use change]	Riverine corridors	Birds use the steep sides of mountain riverine corridors as refuges in transformed landscapes (Lagabrielle et al. 2009).
	Areas of high topographic complexity	Areas of high topographic complexity, such as ravines, are less accessible (Lagabrielle et al. 2009).
[Ability to respond to climate change]	**Forested lowland / upland gradients**	These gradients provide corridors, through which species can rapidly disperse/migrate to more favourable climatic conditions (Raxworthy et al. 2008).
	Areas of high topographic complexity	Topographically complex areas provide refugia in which species might persist, despite surrounding climate change.
Long-term persistence	**Areas of high topographic complexity**	Topographically complex areas provide microhabitats in which species can persist (Qian and Ricklefs 2000; Hopper 2009).
	Areas of high moisture availability	Wet areas could be a useful proxy for increased environmental stability and reduced extinction risk (Fjeldså and Lovett 1997b).
	Areas of low seasonality	Low seasonality, measured as low annual temperature range, could reduce extinctions via increased environmental stability (Fjeldså and Lovett 1997b; Jetz et al. 2004). However, not clear whether patterns of high diversity due to higher speciation or lower extinction.
	Areas with high concentrations of old species	Areas with high concentrations of relict species should exhibit environmental characteristics associated with EEPs that promote long-term persistence.
Speciation	**Areas of high topographic complexity**	Greater topographic heterogeneity allows for more allopatric speciation (Fjeldså and Lovett 1997a; Qian and Ricklefs 2000).
	Areas of high moisture availability	Stable conditions promote speciation (Fjeldså and Lovett 1997b). Annual moisture index is best predictor of endemic plants (Platts et al. 2010).
	Areas of low seasonality	Stable conditions promote speciation (Fjeldså and Lovett 1997b). Low annual temperature range may contribute via increased environmental stability (Fjeldså et al. 1997; Jetz et al. 2004).
	Forested lowland / upland gradients	The environmental gradient found between lowlands and uplands is expected to promote radiative speciation (Lagabrielle et al. 2009).
	Macrohabitat interfaces	Macrohabitat interfaces promote ecological diversification (Cowling and Pressey 2001; Lagabrielle et al. 2009).
	Areas with high concentrations of young, restricted-range species	Areas with high concentrations of young, range-restricted species should exhibit environmental characteristics associated with EEPs that promote speciation.

Diurnal altitudinal movement is of particular concern for birds, which have been observed in the EAM moving along forested altitudinal gradients on a daily basis (J. Fjeldså pers. comm.). Riverine corridors may also provide a route through which birds move (Lagabrielle et al. 2009).

Resilience to nest predation, a potential threat in fragmented landscapes, is another potentially important EEP (Wilcove 1985; Lahti 2001). This edge-effect and others have led to efforts to promote reserve aggregation and maximise patch size (e.g. Moilanen and Wintle 2007; Smith et al. 2010). However, within the EAM and elsewhere, fragmentation is an inconsistent predictor of nest predation (Carlson and Hartman 2001; Lahti 2001; Lahti 2009; Newmark and Stanley 2011).

Pollination and seed dispersal are of fundamental importance to the continued persistence of EAM forests and to the species that depend upon them (Newmark 2002). Retention of vegetation across altitudinal gradients is important, as plants flower asynchronously across elevations (Lagabrielle et al. 2009). Hence fructivorous, granivorous and nectarivorous species that follow fruiting and flowering of plant species over such a gradient can access a continuous supply of resources and promote seed recruitment and fertilisation of forest flora. In addition, pollination and seed dispersal are under threat in smaller fragments and in fragments where, because of high hunting pressure (which is likely to correlate with human population density), fewer pollinators and dispersers exist to enable these processes to persist (Nielsen 2006; Vanthomme et al. 2010; Henschel et al. 2011).

Seasonal migration has been documented for two corridor types in the EAM: First, corridors enabling African wild dog, buffalo, elephant, lion and sable antelope (hereafter referred to as large mammals) to migrate between key protected areas have been mapped (Jones et al. 2007; Epps et al. 2008; Jones et al. 2009b). Corridors such as these are vital for population viability of wide-ranging species (Newmark 1991; Rosenberg et al. 1997; Cushman et al. 2009; Morrison and Boyce 2009; Newmark 2009). Second, forested altitudinal migration corridors link lower altitude forest to higher altitude forest and are expected to be crucial to the persistence of bird species which move between these areas seasonally (Stuart 1983; Burgess and Mlingwa 2000; J. Fjeldså pers. comm.; Newmark 2002).

Disturbance and subsequent succession also constitute processes upon which many other EEPs depend. In parts of the EAM elephants are a particularly important source of disturbance, leading to niche diversity and further inter-specific interactions (e.g. Pringle 2008). Elephant presence has been mapped as part of the African elephant database (Blanc et al. 2007) and the corridors they use to move between important conservation areas have been described (Jones et al. 2007; Epps et al. 2008; Jones et al. 2009b). Maintaining the links between key sites for elephants is important both to maintain the disturbance regime and to prevent the animals becoming trapped within EAM forest, which could also have negative impacts upon the ecosystem (Jones et al. 2007).

Resilience to fire in EAM forests is expected to be compromised in fragmented landscapes due to the drying out of forest edges and increased anthropogenic activity along forest boundaries (S. Madoffe pers. comm.; R. Temu pers. comm.). Resilience is also likely to be highest in the wettest areas, so moisture availability may help to identify forests under less threat.

Resilience to invasive species is important given that invasive species are an increasing threat globally (Cronk and Fuller 1995) and in the EAM (Dawson et al. 2009). Nevertheless, there are few data on the spatially explicit processes that bestow resilience on natural habitats except that areas of low road density and within core areas of reserves (over 1.5 km from an edge) show much lower levels of alien invasion by plants (Foxcroft et al. 2011).

Resilience to land use change might be conferred by the inaccessibility of steep ravines associated with riverine corridors in mountainous regions. This proxy has been used to promote long-term bird persistence elsewhere in Africa (Lagabrielle et al. 2009), but might also be used for other, particularly sedentary, species. Alternatively, a measure of topographic complexity could be used to identify such areas.

Ability to respond to climate change may rely both on the availability of climatic refugia and species' ability to reach those refugia. Forested altitudinal gradients should enable species to respond to climate change by providing a dispersal corridor to areas with the climatic conditions to which the species is adapted; therefore, it is imperative to conserve contiguous forest across such gradients (Raxworthy et al. 2008). In addition, areas of high topographic complexity might also provide micro-habitats in which species can persist (Loarie et al. 2009).

Long-term persistence can be likened to a temporal corridor, linking extant species with the distant past and future. Topographic complexity is, again, likely to be important in providing micro-habitats in which species persist (Loarie et al. 2009; Sandel et al. 2011). In addition, moisture availability in the EAM has been associated with greater species diversity, perhaps due to reduced species' extinctions (Lovett 1985; Gentry 1988; Fjeldså and Lovett 1997b). Another indicator of environmental stability is low seasonality, which can be estimated from annual temperature range. Lower seasonality (measured by mean annual temperature range) has been correlated with greater species diversity, due to either decreased extinction rates (i.e. enhanced long-term survival) or increased speciation rates (Fjeldså et al. 1997; Jetz et al. 2004; Platts et al. 2010). The presence of high concentrations of old species might highlight the areas that have provided most eco-climatic stability in the past.

Proxies for *speciation* are difficult to disentangle from those for long-term persistence, both spatially and in terms of drivers. Topographic complexity is expected to provide more opportunities for speciation (Fjeldså and Lovett 1997a; Qian and Ricklefs 2000; Jetz and Rahbek 2002), while greater environmental stability, mapped as either greater annual moisture availability or lower annual temperature range (seasonality), is also expected to promote speciation (Fjeldså and Lovett 1997b; Platts et al. 2010; Arponen 2012). In addition, macro-habitat interfaces and forested altitudinal gradients may also foster opportunities for ecological diversification and radiative speciation (Cowling and Pressey 2001; Jetz and Rahbek 2002; Lagabrielle et al. 2009). Lastly, areas that show higher concentrations of young, restricted-range species may also be the places where features that promote speciation can be conserved (Arponen 2012).

Identifying the EEPs that we know about is the first step in the criteria of Pressey et al. (2003). From this list (Table 3.1) I chose large mammal migration corridors, altitudinal migration corridors, areas of high topographic complexity, areas of high moisture availability and areas of low seasonality as proxies to represent nine meso-scale EEPs (Table 3.1, in bold). These nine EEPs are understood well enough to map, they would not necessarily be conserved under a conservation plan based on species' distributions alone, and they can plausibly be influenced by a regional conservation plan. Neither resilience to nest predation nor resilience to invasive species are understood well enough in the EAM to identify proxies that might be used to map them spatially. Using areas of low fragmentation as a spatial proxy for EEPs is better done through the use of generic rules that encourage aggregation or enforce minimum patch sizes (see section 3.1.2.). These generic rules cannot easily be applied to specific processes or to specific areas and are, therefore, less accountable. Accordingly, fragmentation measures and aggregation methods were not used as proxies for EEPs. The evidence for riverine corridors as proxies for diurnal movement and as sanctuaries from surrounding land use change is also speculative; forested altitudinal gradients and topographic complexity were judged to better represent these EEPs. Likewise, there is little evidence for the importance of macrohabitat interfaces for speciation within the EAM, other than the highland/lowland interface, which is best captured within continuous forested altitudinal gradients. Remote or inaccessible areas, such as those of low human population pressure or low road density, are more easily included by using cost constraints within SCP software to preferentially select sites away from areas of high value (see chapters four to six). Finally, the distribution of areas currently mapped as having high concentrations of old and young species is biased taxonomically and by where sampling efforts have been concentrated (see chapter two). Therefore, I use these data here only to validate other proxies.

3.2. Methods

3.2.1 Mapping the proxies of biological processes

The EAM exhibit remarkable endemism within single mountain blocs (Scharff 1992; Burgess et al. 2007c). The EEPs that help maintain these species must, therefore, be preserved within each bloc. For instance, even if the wettest areas of the region are all within one bloc, conservation of this spatial proxy for long-term persistence in only one bloc will not promote the long-term persistence of species in other blocs. In this case, the wettest areas of each bloc should be identified for EEP conservation. This is true for all the EEPs included here except large mammal corridors, which are only mapped for the southern blocs, as large mammals (elephant, buffalo and African wild dog) have long been absent from the rest of the EAM.

I have divided EEPs into two groups: spatially fixed and spatially flexible. Spatially fixed EEPs are those that must conform to a specific configuration, relative to other selected planning units. Corridors are a good example of this; a corridor must be conserved in its entirety if it is to contribute to conserving the process of migration. Spatially flexible EEPs are those for which there might be several options for their conservation and their contribution to EEP conservation targets is less dependent on their position relative to other selected planning units.

3.2.1.1 Spatially fixed proxies for process conservation

Large mammal migration corridors: Udzungwa Mountain National Park (UMNP) forms a vital link between some of the largest protected areas in Tanzania, including the Selous Game Reserve, Ruaha National Park and Mikumi National Park (Jones et al. 2007; Epps et al. 2008). The corridors linking UMNP to other important protected areas have been identified through interviews with locals and field-based surveys for dung and other signs of large mammals (Jones et al. 2007). The parts of the corridor that fall outside of the EAM, between UMNP and other national parks and game reserves extend beyond the study area, so cannot be considered here; however, these sections will also require conservation action (see Jones et al. 2007 for details). Apart from the populations utilising these corridors in the Mahenge, Rubeho and Udzungwa blocs, large mammals have been hunted out from the EAM (N. Burgess pers. comm.). Thus, these large, remote areas and the corridors that link them are the last opportunity for conservation of large mammal movements in the EAM (Jones et al. 2009a; Mduma et al. 2010).

Forested altitudinal gradients: Within each mountain bloc, the altitudinal range of each continuous forest patch was calculated. This was done by using GIS tools to select lowland forest, sub-montane forest, montane forest and upper-montane forest and reclassifying them as one land cover type before converting this 'forest' layer into a polygon. The zonal statistics tools in ArcMap (Environmental Systems Research Institute 2009) were then used to return the minimum and maximum elevation within each forest patch. In each mountain bloc, the patch with the largest range was identified and selected. In addition, any other forests within the same bloc were selected if they covered three or more types of forest (of lowland, sub-montane, montane and upper-montane) or if they had the same variety of forest types as the previously-identified 'best' forest within the bloc. In total, 25 forest patches were identified across the 12 mountain blocs. These forest patches show the greatest within-bloc potential for conservation of contiguous habitat over an altitudinal gradient. From within each patch, I then selected the PU with the highest elevation as well as all PUs within 10% of the altitude of the lowest PU within the patch. The shortest path (through continuous forest) between the highest PU and the closest low PU was then identified and mapped to give a single altitudinal gradient for each of the 25 forest patches chosen.

3.2.1.2 Spatially flexible proxies for process conservation

For the following three proxies - areas of high topographic complexity, high moisture availability and low seasonality - only forested areas were considered suitable for conservation of the EEPs which they represent. Therefore, PUs with less than 0.5 km^2 of forest were excluded from this exercise.

Topographic complexity: Deriving a vector ruggedness measure from a digital elevation model was achieved using methods from Sappington et al. (2007). Mean topographic complexity was computed for each PU and the most rugged 10% of PUs within each mountain bloc were selected as potential areas for conservation of this proxy.

Moisture availability: Moisture availability was assessed using an annual moisture index (described in Platts et al. 2010). This measure was found to be the best predictor of endemic plant distributions in the EAM (Platts et al. 2010); therefore, it is likely to be a useful proxy of areas of high speciation and/or long-term species' persistence. Again, moisture availability was calculated for each PU and the wettest 10% of PUs within each mountain bloc were selected as areas in which this proxy could be conserved.

Seasonality: Areas of low seasonality were identified by mapping annual temperature range (see Hijmans et al. 2005; Platts et al. 2010). Within each mountain bloc, the 10% of PUs with the lowest temperature range were identified as areas of low seasonality that could be prioritised for EEP conservation.

Experts agreed that these proxies provide reasonable spatial representation of the areas where these EEPs take place (J. Fjeldså pers. comm.; R. Temu, pers. comm.). In addition, 14 old bird species (those that evolved before the mid-Miocene climatic optimum, 18-16 MYA) and 37 young, restricted-range bird species (those that evolved since the Pleistocene epoch, 0.01-2.6 MYA, and whose global range is less than 82,000 km^2) were identified and mapped as in chapter two (J. Fjeldså pers. comm.; Fjeldså and Lovett 1997a; Appendix B). Spearman's rank correlations were then used to assess how well each of the three measures for topographic complexity, moisture availability and seasonality correlated with the distributions of old and young species.

3.2.2 Spatial data

Altitudinal data were from a global digital elevation model (Jarvis et al. 2008; Platts et al. 2011), from which topographic complexity was also derived (Sappington et al. 2007). Annual temperature range (seasonality) and annual moisture index were derived from WorldClim (Hijmans et al. 2005; Platts et al. 2010). EAM boundaries were from Platts et al. (2011) and elephant distribution data were from the African elephant database (Blanc et al. 2007). Other species' distribution data are described in chapter two.

3.2.3 Systematic conservation planning

These mapped proxies have different degrees of flexibility, resulting in different levels of optimisation that can be applied to them. Large mammal corridors are fixed; the linkages between protected areas are so few that there are no alternative options where these migrations could occur (Epps et al. 2008; Jones et al. 2009b). Contiguous forested altitudinal gradients are also relatively fixed, due to the requirement for particular configurations of PUs; PUs are required to be adjacent to each other and within contiguous forest cover to be valid. On the other hand, topographic complexity, moisture availability and seasonality have no inherent linkage requirements, so the best areas can simply be optimised using simulated annealing. Simulated annealing is an iterative process, whereby planning units are added to or removed from the reserve system if this improves the network's efficiency. Occasionally, and with decreasing frequency as the search progresses, the algorithm will allow an addition or removal from the network that decreases the overall efficiency. This feature of simulated annealing allows for rapid identification of near-optimal solutions, while ensuring that the solution does not become trapped at local optima (Moilanen and Ball 2009; see also chapter one).

Priorities for EEP conservation were, therefore, developed in stages. First, PUs contributing to large mammal migration corridors were selected, as these are inflexible (i.e. irreplaceable). Second, where not already incorporated, the priority set was expanded to include the 25 forested altitudinal gradients identified across the EAM. Third, these PUs were locked in to the solution and SCP software was used to identify the additional PUs needed to meet representation targets for topographic complexity, moisture availability and low seasonality most efficiently. Representation targets for areas of high topographic complexity, high moisture availability and low seasonality were set at 50% of each bloc's candidate sites for these proxies (which were defined as all PUs in the top 10% for the relevant proxy within each mountain bloc). This target gave flexibility to the final solution whilst still ensuring that only areas exceeding the (relatively high) 10% threshold were prioritised. This conservative threshold was chosen because of the uncertainty about the relationship between the proxies and actual EEP representation.

The SCP software, Marxan (Ball and Possingham 2000), was used to derive 1000 solutions to each of three different scenarios under a simulated annealing schedule. Each run passed 1 million iterations and aimed to meet each of the representation targets within a minimum area. Priorities were assessed both by looking at the percentage of times (out of 1000) that a PU was in the final solution (irreplaceability) and identifying the overall cheapest (smallest) solution. The first scenario was an unconstrained prioritisation, based entirely on 9 km^2 PUs (the median size of all state-owned protected areas within EAM districts). This identifies a near-optimal solution. The second scenario was constrained so that all nationally and internationally designated protected areas (according to the WDPA; IUCN 2010a) were locked in to the final solution. The final scenario also incorporated current protected areas as PUs but, in this scenario, they could be removed from the solution if they did not contribute to meeting targets for EEP conservation. This enables assessment of individual reserves' contribution to process conservation. The second scenario, gives the least flexibility and results in the largest (most costly) solutions, but it is also the most realistic, as reserves are unlikely to be removed from the network (although see Mascia and Pailler 2011). It identifies PUs that are most complementary to the current reserve network. A further advantage to this scenario is that it does not discount reserves that were gazetted for the preservation of species or conservation features not considered in these analyses, whose inclusion in the EAM conservation network may nonetheless be well justified. For all analyses, no boundary length modifier was used.

3.3. Results

3.3.1 Priorities for spatially-fixed conservation features

The EAM cover 50,800 km^2, of which 21% (10,540 km^2) currently falls within protected areas (IUCN 2010a). In total, 6% (2,775 km^2) of the area of the EAM was selected for conservation of corridors crucial to the persistence of large mammal migrations (Figure 3.1a), while the minimum area to conserve the 25 prioritised forested altitudinal gradients is 2% (1,019 km^2; Figure 3.1b). Overlap between these and large mammal corridors is low (just 4% of area selected for conservation of altitudinal gradients was also selected for conservation of large mammal corridors), so that, in total, an area of 3,749 km^2 was required for these two proxies. Under the current system of reserves, 2,625 km^2 of this area (70%) is already within protected areas, which cover just 21% of the entire EAM, suggesting that the current network is reasonably well suited for conservation of these EEPs.

As well as the need to protect remaining habitat to conserve the processes represented in forested altitudinal gradients, forest restoration may also be necessary in the Nguu, Uluguru and Mahenge mountains. In these blocs only 65-68% of the total altitudinal range of extant forest is represented within a single continuously forested patch (Appendix C).

3.3.2 Priorities for spatially-flexible conservation features

Priority areas of high topographic complexity, high moisture availability and low seasonality (Figure 3.2a-c respectively) were also mapped. There is some correlation between these variables, but it is low (Table 3.2), highlighting the usefulness of conservation planning tools, which are capable of selecting efficient solutions in which priorities between conservation features overlap. There is also some degree of overlap between the fixed and flexible priority areas with 34%, 29% and 51% of the best areas for the conservation of rugged, moist and low seasonality areas, respectively, falling within the areas selected for conservation of large mammal corridors and forested altitudinal gradients combined.

Of the most topographically complex terrain, 70% (231 km^2) is already under some form of protection. However, this is not spread proportionately between mountain blocs, so that the most rugged terrain in some blocs remains unprotected. In addition, 90% (510 km^2) of the wettest area and 83% (364 km^2) of the area with the lowest seasonality also falls within current protected areas.

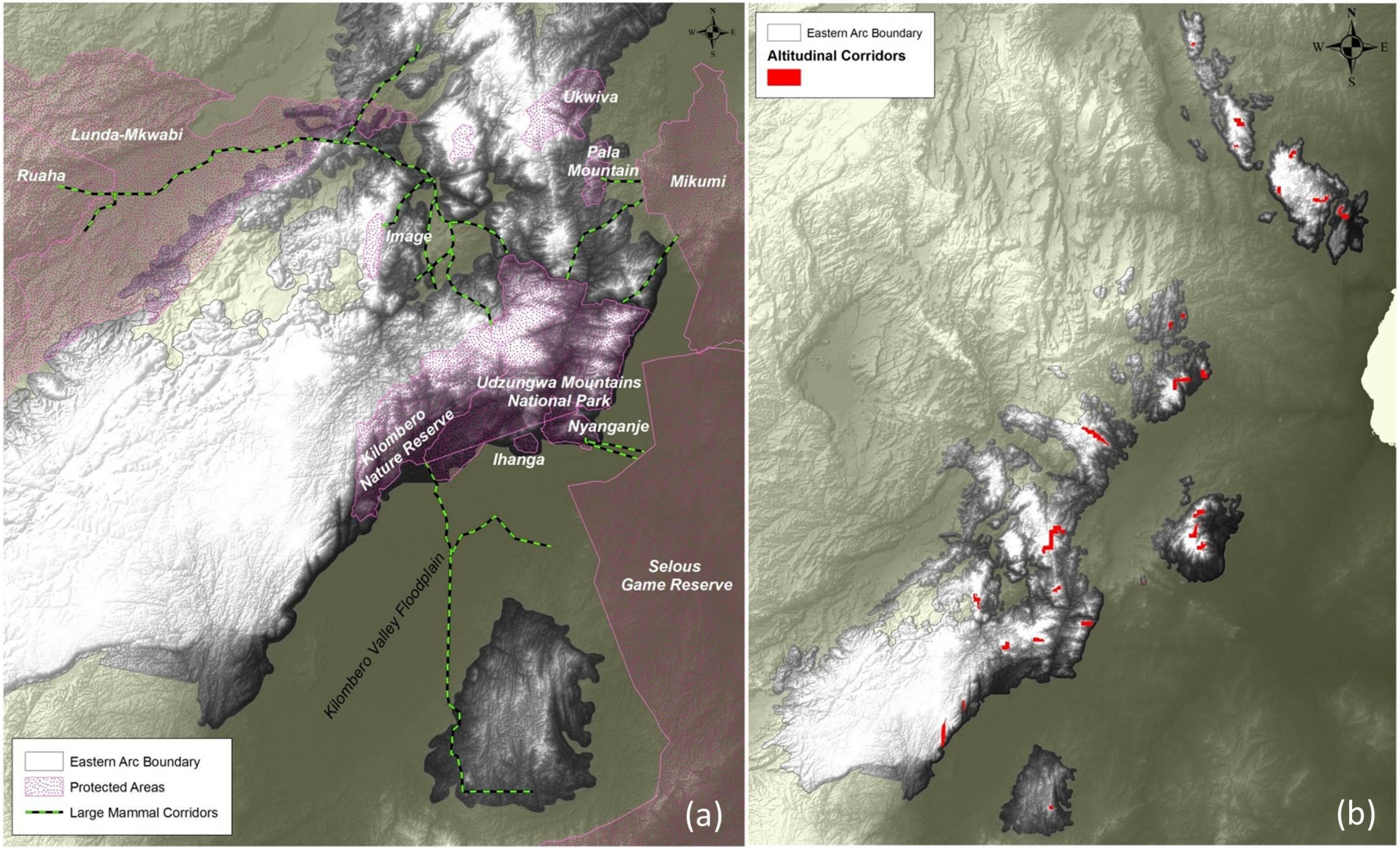


Figure 3.1. Priority areas for conservation of the last remaining large mammal corridors linking the Udzungwa Mountains National Park with other important protected areas in the region (a) and forested altitudinal gradients within each mountain bloc (b).

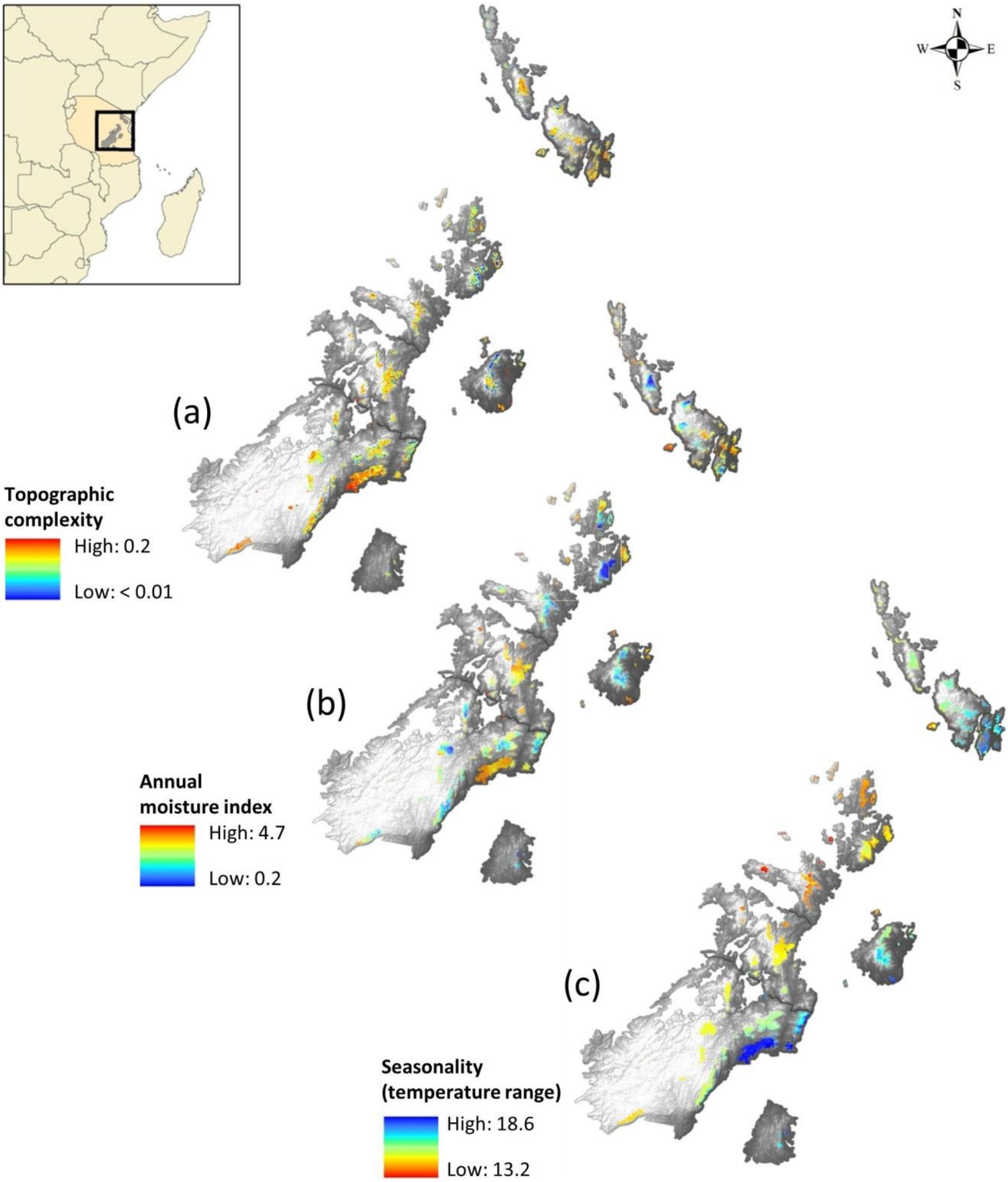


Figure 3.2. **Topographic complexity (a), annual moisture index (b) and seasonality (c) are shown for forested areas of the Eastern Arc Mountains. For each proxy, red areas represent the areas expected to be most important for conservation of the Ecological and Evolutionary Process that it represents, while blue areas are expected to be least important.**

Table 3.2. Pearson's product-moment correlation coefficient for relationships between mean topographic complexity, annual moisture index and annual temperature range of each planning unit (*n* = 967).

	Annual Moisture Index	Seasonality (Annual Temperature Range)
Topographic Complexity	**0.183**	**0.164**
Annual Moisture Index	**-**	**0.141**

3.3.3 Validation

Comparison of areas of high topographic complexity, high annual moisture index and low seasonality with the distributions of very old species and young, range-restricted species (Fjeldså and Lovett 1997a) bears out some of the assumed links between such proxies and the EEPs that generate and maintain biodiversity (Figure 3.3; Table 3.3). Nevertheless, these proxies are clearly imperfect at capturing all the factors that influence species generation and survival (Figure 3.3; Table 3.3). In part, this could be due to the fact that species' distribution maps are biased geographically, while the topographic and climatic proxies are mapped across the EAM. In addition, variation in species richness is low (old species richness range = 5; young species richness range = 15), reducing the power of the analyses to detect relationships.

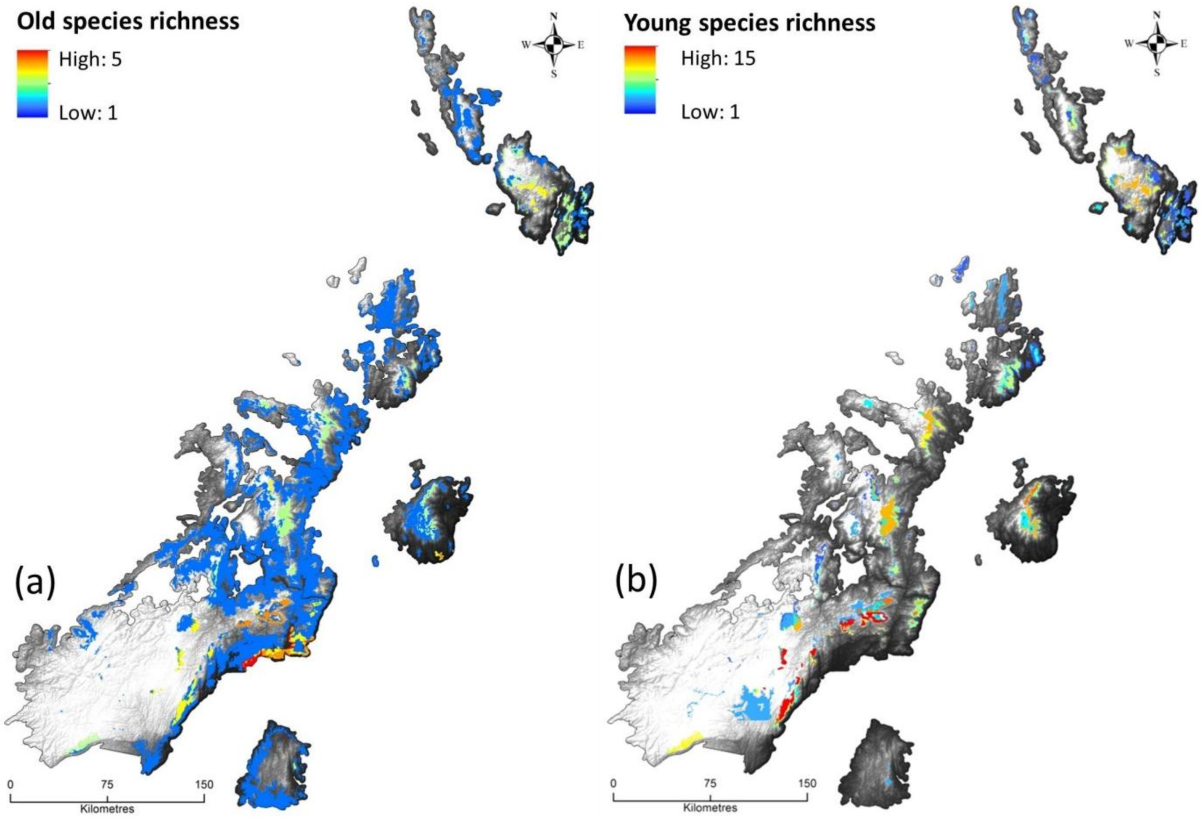


Figure 3.3. Richness of old species (a) and young, restricted range bird species (b) (see Appendix B for species lists).

Table 3.3. Spearman's Rank correlation coefficients for relationships between planning units' (*n* = 967) mean values for topographic and climatic variables and richness of old and young, restricted range bird species.

	Species Richness	
	Young Species	**Old Species**
Topographic Complexity	**0.39**	**0.19**
Annual Moisture Index	**0.12**	**-0.14**
Seasonality (Temperature Range)	**0.22**	**-0.19**

3.3.4 Systematic conservation planning for processes

The minimum area required to achieve the targets for all proxies is 4,160 km^2 (Figure 3.4a). When information on currently protected areas is included but their removal from the protected area network is allowed, the importance of different reserves can be seen – some, such as Udzungwa Mountains National Park in the east of the Udzungwa bloc, are in the final solution at every run, while others are never in the final solution (Figure 3.4b). Finally, if, as would be expected, the current protected area network (10,540 km^2) is retained, then it must be supplemented by a further 1,604 km^2 (15%) to meet these targets (Figure 3.4c).

3.3.5 Comparison with priorities for biodiversity

The importance of considering EEPs alongside biodiversity pattern can be demonstrated by considering how priorities change when EEPs are prioritised. I will examine this in more detail in chapter six. However, a quick illustration of the effect of targeting EEPs can be seen by considering the areas of low richness for species of conservation concern (whose derivation is explained in chapter two) that are included in a plan for EEP conservation (Figure 3.5). In particular, the mammal corridors cross large tracts of habitat which is low in richness of priority species, yet provides vital links between key sites for large mammals. A prioritisation exercise that only includes information on the patterns of species' distributions would evidently not be sufficient to conserve these EEPs.

3.3.6 Current representation of processes

Under the current network of reserves, EEPs are reasonably well represented. Of the targets set for each proxy, the percentage coverage within the current protected area network ranges from 64% to 100% (Table 3.4). Most poorly represented are forested altitudinal gradients (64% are included in the current reserve network), and it is for this proxy that the greatest number of EEPs have also been identified.

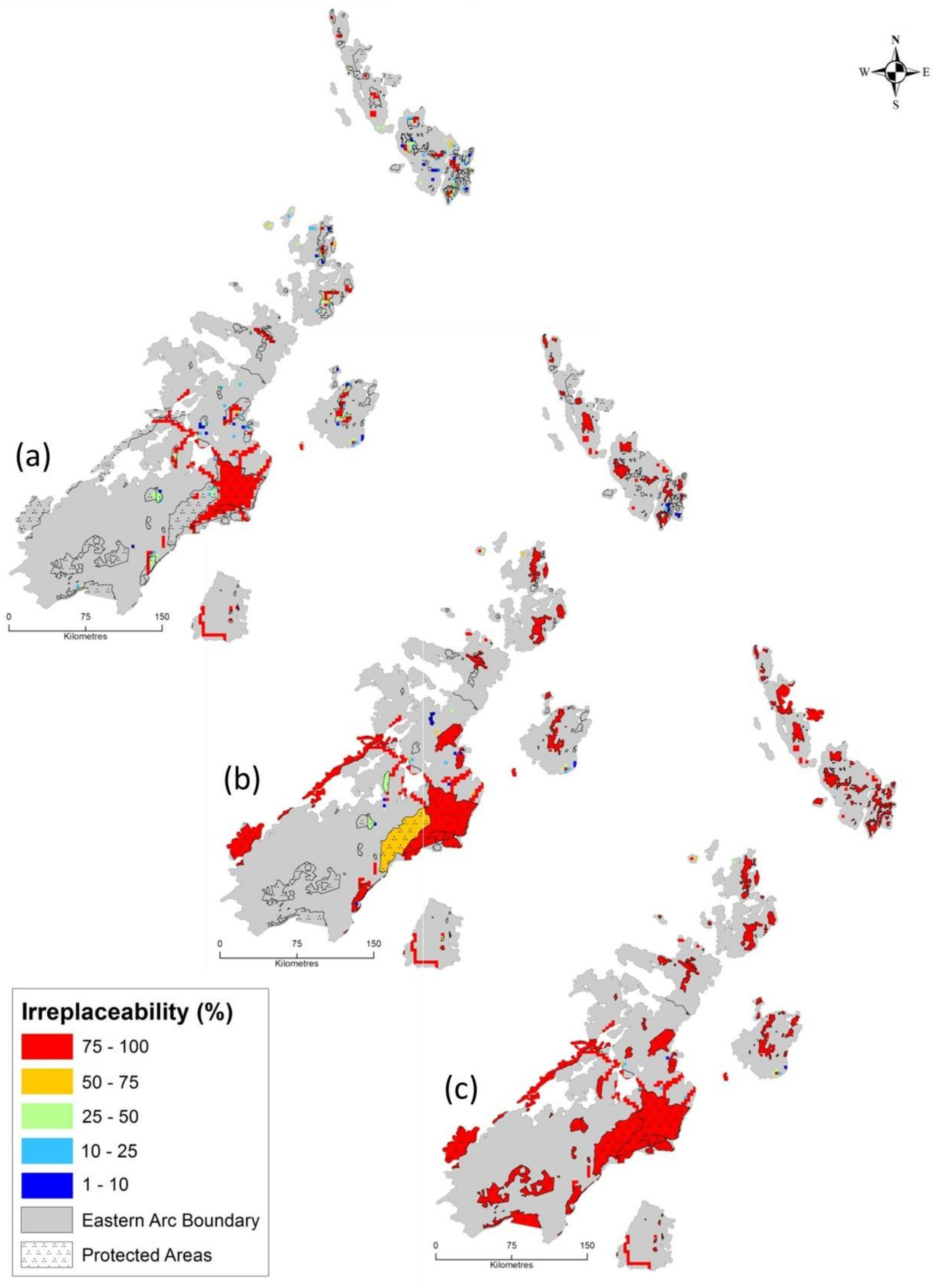


Figure 3.4. The number of times a planning unit is included in the final solution (out of 1,000 runs) indicates its importance (irreplaceability) for conserving Ecological and Evolutionary Processes (EEPs). Grey areas are within the Eastern Arc Mountains, but have an irreplaceability of zero. a) The unconstrained solution, which uses 9 km^2 planning units only. b) Current protected areas are also incorporated as planning units, but in this scenario they can be removed from the solution if they do not contribute to meeting targets for EEP conservation. This enables identification of individual reserves' contributions to the conservation of EEPs. c) Current protected areas are locked in to the final solution, so that those planning units that are most complementary to the current system are highlighted.

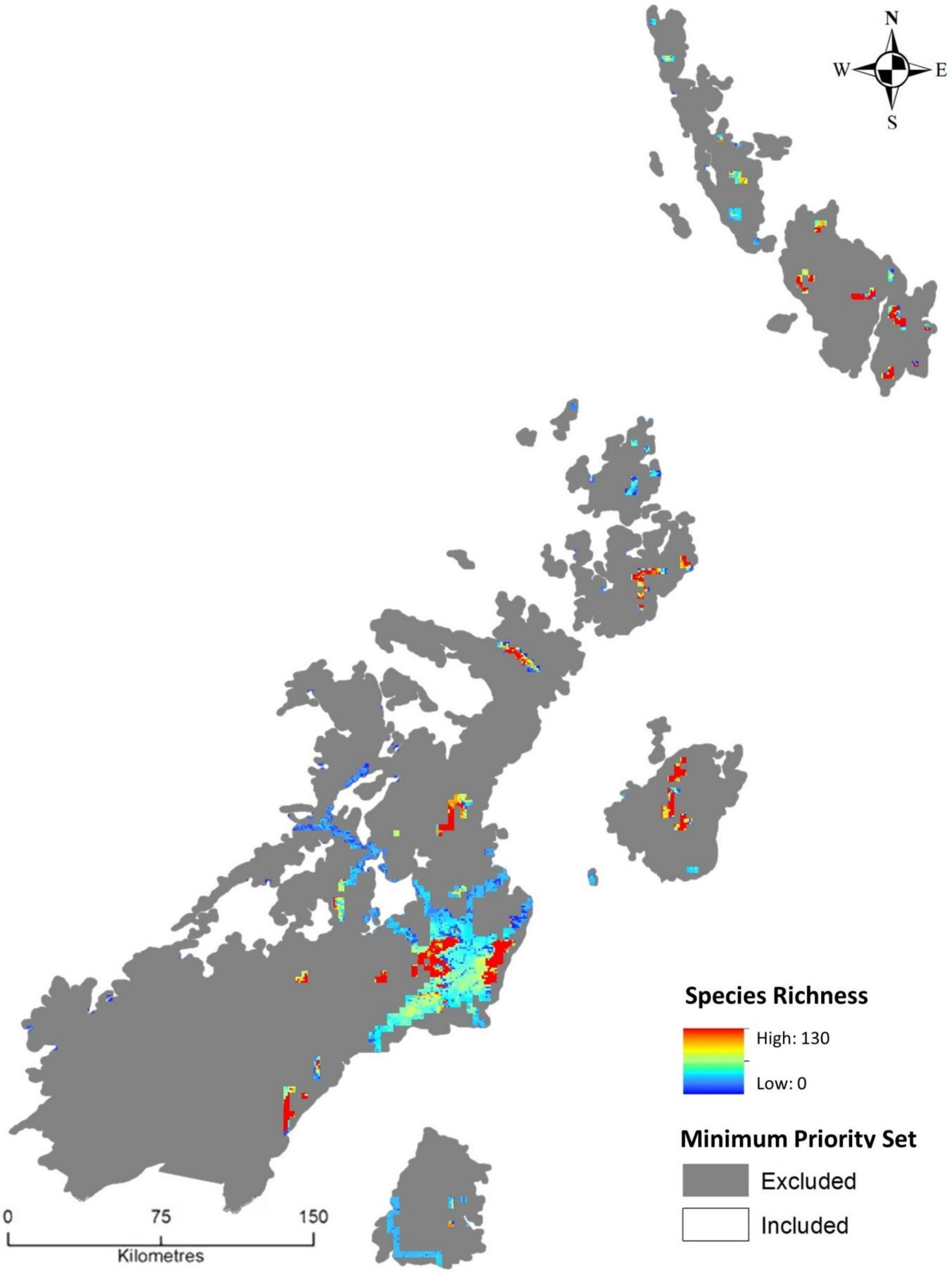


Figure 3.5. The importance of explicit consideration for conservation of Ecological and Evolutionary Processes (EEPs) is highlighted by the fact that the minimum area set (non-grey areas) includes large areas of low richness for species of conservation concern (blue). Red areas that are included in the minimum set are places where selection of planning units is likely to contribute most to meeting representation targets for both species pattern and EEPs.

Table 3.4. Coverage of proxy targets under the current protected area network and representation of Ecological and Evolutionary Processes.

Proxy	Target Coverage	Processes represented
Large mammal migration corridors	69%	- Seasonal migration - Disturbance
Forested altitudinal corridors	64%	- Diurnal altitudinal migration - Pollination/seed dispersal - Seasonal migration - Ability to respond to climate change - Speciation
Areas of high topographic complexity	99%	- Resilience to land use change - Ability to respond to climate change - Long-term persistence - Speciation
Areas of high moisture availability	100%	- Resilience to fire - Long-term persistence - Speciation
Areas of low seasonality	98%	- Long-term persistence - Speciation

3.4. Discussion

The spatial distributions of five proxies were mapped, highlighting priorities for conserving nine EEPs (of eleven identified as important within the study area). Conserving all of the fixed proxies and 50% of the best area for flexible proxies can be achieved within a minimum reserve network of a little over 4,000 km^2. However, if the current reserve network is maintained, then it would need to be increased by 15% (from 10,540 km^2 to 12,144 km^2) to include the most complementary planning units for EEP conservation. Without this expansion, large mammal migration corridors and forested altitudinal gradients are particularly at risk. Seasonal and diurnal migrations; natural disturbance and succession regimes; and dispersal of flora and fauna might cease, whilst the region's flora and fauna will also be more vulnerable to climate change. In addition, the evolutionary processes that characterise the mountains may be impaired. Importantly, SCP for processes has the ability to shift priorities for conservation compared to when just species pattern is considered. For instance, large areas of habitat that is low in diversity for species of conservation concern, yet crucial to species' persistence were needed for the maintenance of large mammal corridors. This highlights the inadequacy of using species' distribution data alone for generating conservation plans for long-term species' survival.

These analyses represent minimum targets. Forested altitudinal gradients are the least well represented of the proxies used here and should be a focus of future conservation efforts. These gradients are so fundamental to the conservation of EEPs, that restoration or expansion of forest to increase their range and number should be encouraged. In particular, the Nguu, Uluguru and Mahenge mountain blocs only have between 65 and 68% of their total forested altitudinal range represented within continuously forested patches (Appendix C). As long as low and high elevation forest remains in these blocs, there still exists an opportunity to recreate the links between them. Once either is lost, this will become much more difficult.

One of the biggest challenges of incorporating EEPs into conservation planning lies in disentangling specific processes and their spatial proxies. In fact, conservation of one EEP is very likely to conserve others; however the value of the approach outlined here is that every proxy is documented and assumptions regarding how it will represent specific EEPs are made clear. This allows costs and impacts to be attributed directly to the conservation of a particular process, whilst also giving greater confidence in capturing those processes within a conservation plan. Even when using only generic criteria such as minimum size or boundary length penalties, it is useful to consider specific EEPs under a similar framework and to identify those which are likely to benefit from the design principles.

Using topographic and climatic features to set priorities enables important areas to be mapped across the EAM and for multiple taxa. This can be achieved even when little is known about current species' distributions or their phylogenies because it is based on globally available data. Therefore, setting targets for conservation of proxies, rather than for distributions of old or young taxa, helped avoid sampling biases. Nevertheless, I was able to compare the distributions of old and young, restricted-range bird species with these priority areas, which indicated that the proxies capture some useful information on how well a site contributes to past and future long-term survival opportunities and speciation, but that the measures are imperfect.

A major impediment to the inclusion of EEPs in SCP lies in the nature of the available data, which tend to be a mix of qualitative and quantitative information, often based on expert judgement. This is juxtaposed against the discipline of SCP, which was borne of a necessity to move away from *ad hoc* reservation (Pressey 1994) and towards a more rigorous, scientifically justified (and often quantitative) approach to conservation planning. However, as Hopper (2009), quoting Ghiselin (1969), points out, the scientific method is based upon validation of premises and use of logic and "The truth does not derive from the jargon in which it is expressed". Therefore, although much of the information on EEPs is difficult to quantify and may be based on expert opinion rather than extensive databases, so long as the assumptions and premises are clearly stated and logical, making judgements about where EEPs are most likely to occur is valid and has the potential to greatly improve the practical contribution of systematic conservation plans. Given that few would deny the importance of EEPs, and given the growing evidence that they are not likely to be effectively conserved simply in pattern-derived priority networks (Cowling et al. 1999; Cowling 2003; Rouget et al. 2003b; Rouget et al. 2006; Figure 3.5), the reluctance to incorporate them into SCP should be avoided. Nonetheless, methods for their inclusion must be explicit about what processes are being conserved and how they are mapped. Assumptions and methods can then be tested and refined or rejected as knowledge is developed.

The maps presented here show priorities derived without consideration of species. This is a useful step in understanding where EEP conservation should take place, but just as prioritising solely on species' pattern does not ensure long-term persistence, neither does prioritising solely on EEPs. Thus, it is insufficient for creating operational conservation plans. In addition, this plan prioritises meeting EEP representation targets in a minimum area, whereas further analyses are needed that incorporate spatially explicit information on the costs of conservation. The following chapters discuss how to collect and integrate such information.

3.5. Summary

Ecological and Evolutionary Processes (EEPs) are fundamental to the long-term conservation of species and ecosystems. Here, several EEPs important to the biodiversity of the Eastern Arc Mountains are identified and spatial proxies for mapping areas for their maintenance are described. Five proxies, contributing to the conservation of eight EEPs, are mapped and included in a conservation prioritisation exercise. The results demonstrate how the inclusion of factors that contribute to healthy ecosystem functioning can dramatically alter priorities when compared to prioritisation using information on species' presence alone. Despite their importance, EEPs are often excluded from Systematic Conservation Planning, largely due to the difficulties involved in expressing them quantitatively and in optimising reserve networks to represent them at a minimum cost. This reluctance should be challenged, otherwise reserve networks will, over the medium to long term, lose those elements of biodiversity that they were established to conserve.

4. Management costs of protected areas

“The survival of our wildlife is a matter of grave concern to all of us in Africa. These wild creatures amid the wild places they inhabit are not only important as a source of wonder and inspiration but are an integral part of our natural resources and our future livelihood and well-being.

In accepting the trusteeship of our wildlife we solemnly declare that we will do everything in our power to make sure that our children’s grand-children will be able to enjoy this rich and precious inheritance.

The conservation of wildlife and wild places calls for specialist knowledge, trained manpower, and money, and we look to other nations to co-operate with us in this important task – the success or failure of which not only affects the continent of Africa but the rest of the world as well.”

Mwalimu Julius K. Nyerere, The Arusha Manifesto (1961)

4.1. Introduction

Systematic conservation planning (SCP) represents a classic economics problem: the allocation of scarce resources to achieve specific objectives (Polasky et al. 2001). It is surprising, then, that few prioritisation exercises address spatial variation in conservation cost. The reason is the lack of data, and an apparent reluctance to collect additional information. Most SCP studies instead use area as a proxy for cost, which makes the assumption that cost is predicted by area, rather than by geographical or socio-economic attributes of the land (Naidoo et al. 2006). However, management cost often does not scale in direct proportion to size (e.g. Frazee et al. 2003; Bruner et al. 2004), so instead explicit consideration needs to be given to investigating both the relationship between reserve size and management costs per unit area and to other potential cost predictors (such as anthropogenic pressure).

SCP was developed to ensure that biodiversity considerations were included in the design of protected area networks; however, often, biodiversity considerations now replace economic considerations. This is simply the opposite solution; rather than using the invalid (but implicit) assumption that all areas are equally valuable for biodiversity, which led to reserves being placed in cheaper locations, the paradigm has shifted to assuming that all areas are equally costly, so reserve systems are designed that collectively represent the greatest biodiversity value. Costs often vary more widely than biodiversity values yet are wholly or largely ignored; therefore, improvements in quality of spatial representation of costs of conservation and their inclusion in SCP will lead to greater gains in efficiency (i.e. biodiversity conserved per unit of investment) than would similar improvements in species distribution data (Balmford et al. 2000; Balmford et al. 2003; Moore et al. 2004; Naidoo et al. 2006; Polasky 2008).

Aside from efficiency gains that inclusion of spatially explicit cost data afford, quantification of costs enables assessment of the impact of conservation on people's livelihoods. Effective and equitable conservation requires that the cost burden is shifted away from local people and externalities are internalised through sharing the costs with the wider community of beneficiaries (Balmford and Whitten 2003; Venter et al. 2009). Developing spatially explicit maps of conservation costs will therefore enable more efficient reserve network design (Polasky 2008), demonstrate where costs are borne and by whom (Balmford and Whitten 2003), and, ultimately, aid equitable implementation (Knight et al. 2006a; Linnell et al. 2010).

I investigate both direct costs (of protected area management) and indirect costs (the opportunity costs of land under reservation and cost of wildlife damage around protected areas). This chapter describes my work to model protected area management costs in the

Eastern Arc Mountains (EAM; see chapter one), whilst I describe my work on indirect costs in chapter five. Management costs are those that are directly incurred in maintaining a system of protected areas (Dixon and Sherman 1991; James et al. 1999; Bruner et al. 2004; Morrison and Boyce 2009) and are the focus of several quantification and modelling efforts (James et al. 1999; Wilkie et al. 2001; Balmford et al. 2003; Frazee et al. 2003; Balmford et al. 2004; Burgess and Kilahama 2004; Moore et al. 2004; McCrea-Strub et al. 2010; Busch et al. 2012; Shaw et al. 2012; Wise et al. 2012). Modelling is crucial when understanding the distribution of costs can help guide efficient solutions but data are scarce.

Protected area managers in the study region commonly complain of insufficient funding to manage their reserves effectively, so this analysis explores variation both in current spending on management and in estimated necessary spend. These data, reported by protected area managers in the EAM, are modelled 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, it is possible 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). These findings can also be compared with data on pole and timber cutting in protected areas (Madoffe and Munishi 2010) to investigate whether increased spending is associated with improved management effectiveness.

Although global and international models have been constructed in previous studies (e.g. Balmford et al. 2003; Balmford et al. 2004; Bruner et al. 2004; Moore et al. 2004), 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. Howard 1995; Culverwell 1997; Blom 2004) or they were developed for very specific habitat types and require explanatory variables that do not exist in the EAM (Frazee et al. 2003). Most importantly though, none have looked at spatially explicit variation in both actual and necessary spend.

This chapter focuses in particular on my work to develop a novel measure of population pressure - a measure of human density that is sensitive to the distance people are away from a site. This measure performed substantially better than other socio-economic variables at predicting management cost.

4.2. Methods

The system of protected areas in the Eastern Arc falls under the control of three agencies (Figure 4.1): Tanzania National Parks (TANAPA) manage all National Parks (NPs) in Tanzania, including Mikumi NP and Udzungwa Mountain NP within the EAM study area; 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, the local governments manage Local Authority Forest Reserves (LAFRs) and the 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.

4.2.1 Cost data

During April-June 2010 I 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 protected area funding (James et al. 1999; Burgess and Kilahama 2004; Craigie 2010). I gathered information on the money spent on management of protected areas in the financial year 2008/9[1] from these managers, who were responsible for administering, or assisting with administering, 482 protected areas 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 (Figure 4.1); 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 termed “actual spend”) on protected area management in the EAM. An earlier study (Madoffe and Munishi 2010) found that protected area management effectiveness in the EAM varied, with only one out of 15 state-owned reserves classified as having “good” management effectiveness. Because this performance might be due to inadequate current spending on management, managers were also asked 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.

[1] The financial year in Tanzania runs from July to June

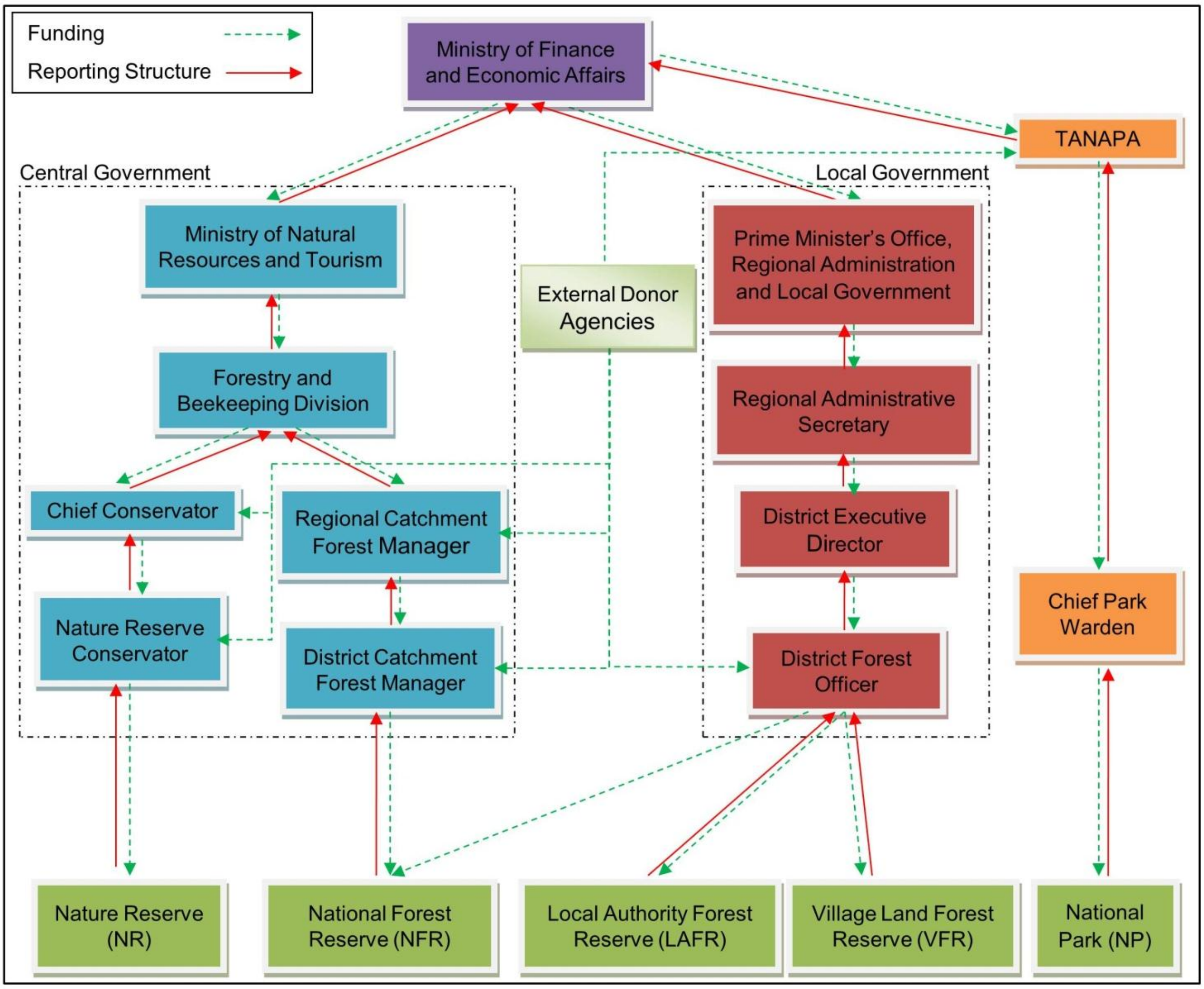


Figure 4.1. Funding and reporting structure for different protected area types (green boxes) in the Eastern Arc Mountains. Red arrows show the reporting structure through the decentralised local government (red boxes), the Central Government (blue boxes) or the Tanzania National Parks (TANAPA; orange boxes). Green arrows show common funding routes. External donor agencies generally fund conservation implementation through district and regional managers, conservators, and TANAPA head office. Reporting structure for external donors depends on the specific contract under which funding is provided.

Median reserve size (LAFRs, NFRs, NRs and NPs) within the current system is 8.8 km^2 (Figure 4.2a). Wherever possible, budgets of individual reserves were obtained 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 such cases, these aggregates were used as the units of analysis (hereafter referred to as “reserve groups”; Figure 4.2b). This lumping could hide or dilute the effects of explanatory variables – particularly protected area size, which has been shown to have a negative relationship with spend per unit area (Balmford et al. 2003; Frazee et al. 2003; Bruner et al. 2004; McCrea-Strub et al. 2010). However, joint management of reserves in groups like this is the reality for many protected area managers both in Tanzania and elsewhere, so an analysis of such units is highly relevant.

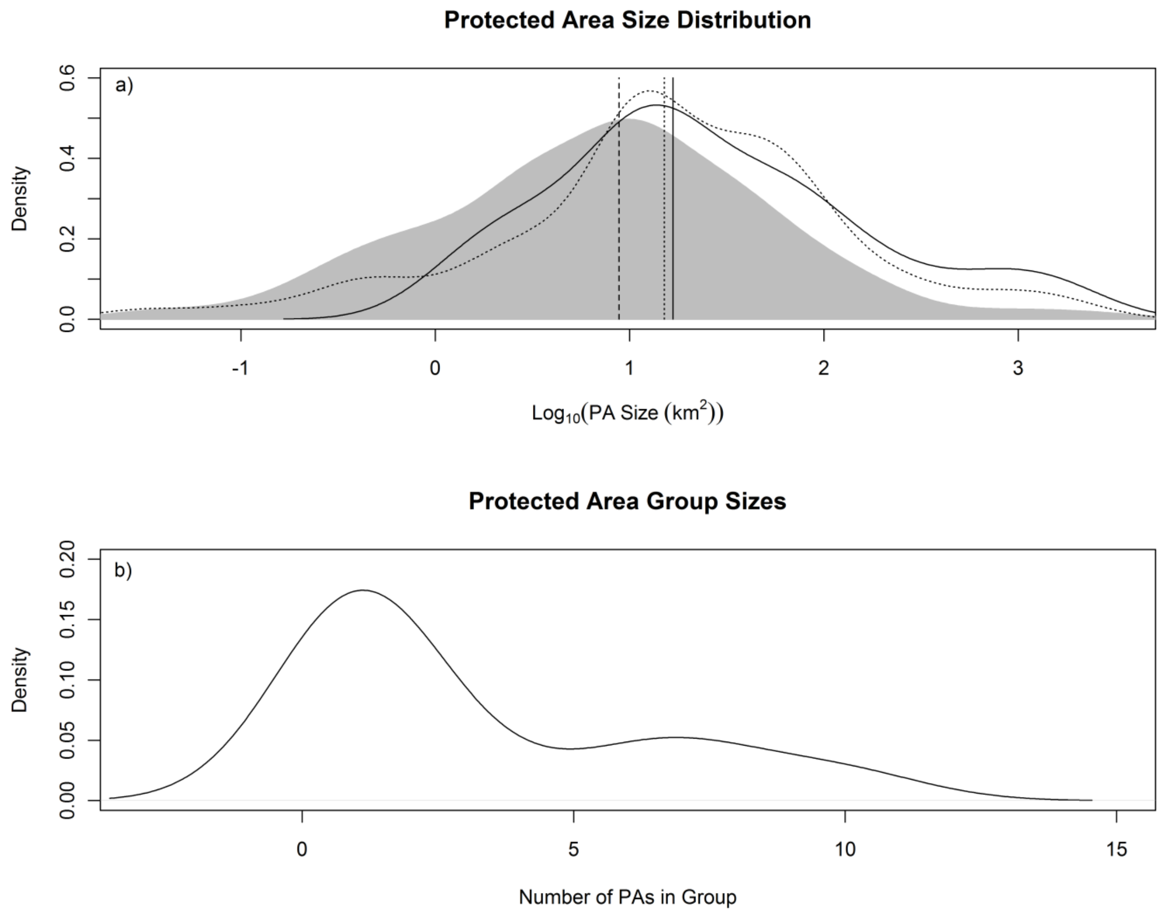


Figure 4.2. a) Kernel density plot of $\log_{10}$(protected area size) shows the median size (8.8 km^2; dashed black line) and size frequency distribution of all reserves in the Eastern Arc Mountains (shaded grey; n = 220) and the size frequency distribution and median sizes (vertical lines) for reserves used in analyses of actual spend (dotted black line; median = 15; n = 74) and necessary spend (solid black line; median = 17; n = 40). b) Kernel density plot of number of protected areas in reserve groups (median = 1; n = 24). In extrapolating models across the study area, number of protected areas in reserve group was standardised to be equal to one and total reserve area standardised to be 9 km^2.

All analyses were conducted in Tanzanian shillings per hectare per year (TZS ha^{-1} y^{-1}), but are expressed in United States Dollars per hectare per year (USD ha^{-1} y^{-1}), using an exchange rate of 1450 TZS = 1 USD. In order to compare my findings with earlier studies, I adjusted reported figures (in USD) for inflation (Index Mundi 2010)

4.2.2 Cost types

Management of protected areas is a complex process involving many kinds of outlay, with a common division made between recurrent expenditure (often calculated per annum) and capital expenditure. However, even this dichotomy is not always easily defined, so I have drawn on several previous studies (Frazee et al. 2003; Bruner et al. 2004) and my own experience of the protected area network in the EAM to develop a classification of management spending (Table 4.1). For all analyses, recurrent plus capital expenditure (but excluding protected area establishment costs) were modelled per hectare per annum.

Models were also built to estimate recurrent costs only but these results are not presented, 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.

4.2.3 Cross validation and missing data

To corroborate information received from questionnaires and to fill gaps where data were missing, supplementary information, provided in annual reports and budgets from various agencies (TANAPA 2001; EAMCEF 2008; FBD 2008; EAMCEF 2009, 2010), was used. Major donors and regional forest managers were also interviewed to cross-validate information received from district-level managers and conservators. Where data were unavailable for 2008/9 (for one nature reserve), the previous year's figures were used (2007/8) and adjusted for inflation to 2008/9 (Index Mundi 2010).

In most cases, managers were uncomfortable with estimating staff salaries. Therefore, for those management groups where there were sufficient data (n = 11) total salary expenditure was regressed against staff number ($\log_{10}(\text{salaries}) = 6.776 + 0.634 * \log_{10}(\text{staff number})$); $n = 11$; $r^2_{adj} = 0.88$; $p < 0.001$). 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. This equation was used to estimate total salary expenditure for reserves where staff number was known but data on salaries were unavailable.

Table 4.1. A classification of cost types. 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 cost of 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/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.

4.2.4 Data analysis

4.2.4.1 GIS data

Spatially explicit modelling requires extraction of predictor variables using reserve boundary shapefiles, so could only be conducted on reserve groups for which GIS data were available (Table 4.2). Of 482 reserves for which financial data were available, 146 were listed in the World Database on Protected Areas (WDPA; IUCN 2010a) and had Geographical Information System (GIS) data associated with them. For actual spend there were 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 GIS information be acquired.

4.2.4.2 Modelling

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

4.2.4.3 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 λ=0.25; necessary spend: Box-Cox parameter λ=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 4.2). The reserve characteristics examined were protected area type, number of protected areas in the reserve group and total combined area of the reserve group. Management systems (and therefore spend) vary between reserve types and 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 (Frazee et al. 2003; Bruner et al. 2004; Nelson and Chomitz 2009), mean terrain ruggedness using a Vector Ruggedness Measure (VRM; see methods in Sappington et al. 2007) and median population density within the protected area were used (see Platts 2012 for a description of the population density layer). I also hypothesised that pressure exerted from outside the boundaries of the protected area could have an effect on the amount of funding that is actually spent and/or necessary. I 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 a measure of "population pressure" was developed, based on Platts (2011).

The population pressure measure 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, the resolution of the population density layer was decreased from 1 km^2 to 25 km^2. I proposed that the distance decay function of the weight applied to population should follow a half-normal distribution, as nearby populations are expected to exhibit highest pressure, which decreases rapidly once the distance to the protected area is beyond walking distance. Thus, population pressure in cell *i* is given by:

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

Equation 4.1.

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 (Figure 4.3). 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. A range of population pressure layers was created, each with a different σ value, for the entire EAM landscape. These were then used to build a series of simple linear regression models of actual and necessary management spend, from which the population pressure layer which gave the best model fit was chosen (lowest AIC_c). The same process was then run to select the buffer size at which the population density layer gave the best model fit, so that, for both population pressure and population density, just one layer each was used in subsequent model construction.

Actual and necessary spend ha^{-1} y^{-1} were then modelled in relation to population pressure or density and other explanatory variables (Table 4.2; due to colinearity, population density and population pressure were never both in the same models, but were analysed separately). Using all combinations of the predictor variables (no interactions), each model was then ranked using AIC_c values.

Table 4.2. Predictor variables used to construct a model 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 protected areas	Questionnaire survey	Number of reserves in group.
Total area (ha)	Questionnaire survey	Total area of reserve group.
Terrain Ruggedness (VRM)	GIS variable[1]	Mean Vector Ruggedness Measure (VRM) or terrain ruggedness[2] of reserve group.
Human use (%)	GIS variable[1]	Percentage of land in 5 km buffer of reserve under human dominated land use type (cultivation, urban and disturbed habitats).
Median population within protected area	GIS variable[1]	The median population within the reserve group
Population density (people/km^2)	GIS variable[1]	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[1]	Mean population pressure of all cells within protected area boundary (in person equivalents per km^2).

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

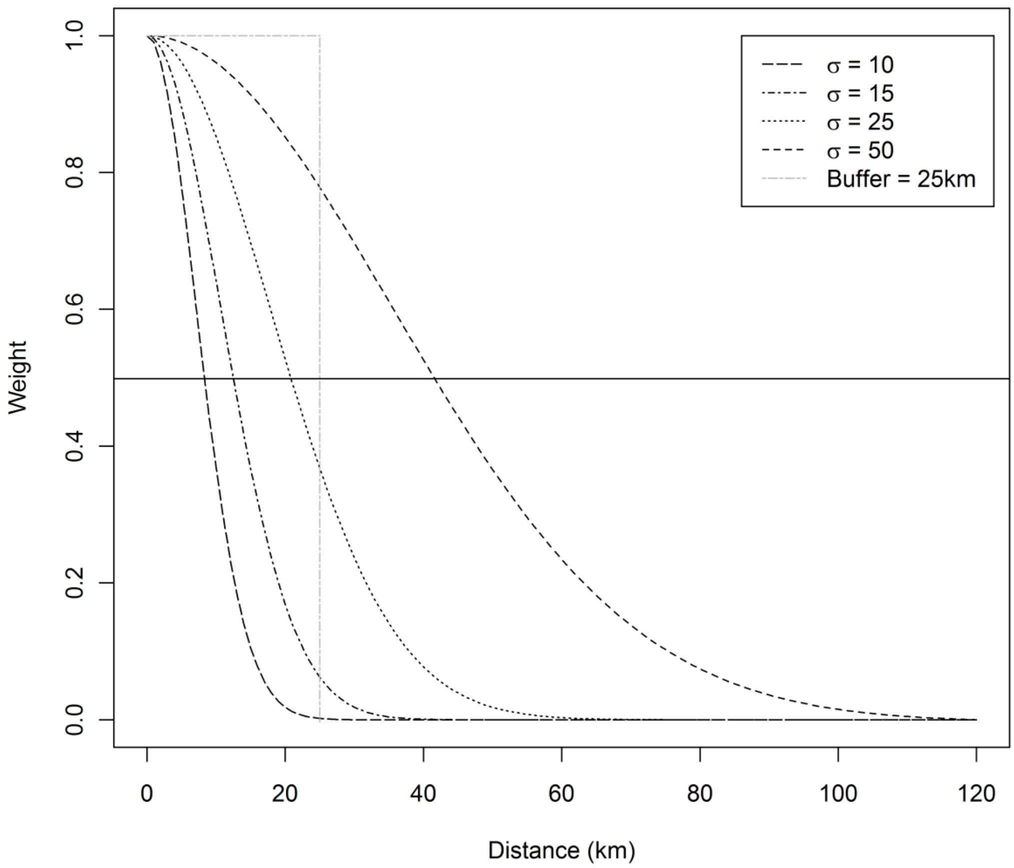


Figure 4.3. Population pressure is hypothesised to impact a particular point in space according to some distance-weighted function. Half-normal curves are used, as nearby populations are expected to exhibit highest pressure, which decreases rapidly once the distance to the protected area is beyond walking 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.

4.3. Results

4.3.1 Actual spend

Across the EAM, 55% of annual protected area spending was on recurrent expenditure (salaries and operating costs; Figure 4.4). Capital expenditure was non-normally distributed across reserves and present in only 20 reserve groups (out of 50 for which I obtained data). Where there was capital expenditure, 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}; interquartile range = 1 to 6 USD ha^{-1} y^{-1}; n = 50). The median amount of money reported as being necessary for a protected area to achieve all its management objectives was 8.3 USD ha^{-1} y^{-1} (mean = 19.7 USD ha^{-1} y^{-1}; interquartile range = 5 to 17 USD ha^{-1} y^{-1}; n = 29).

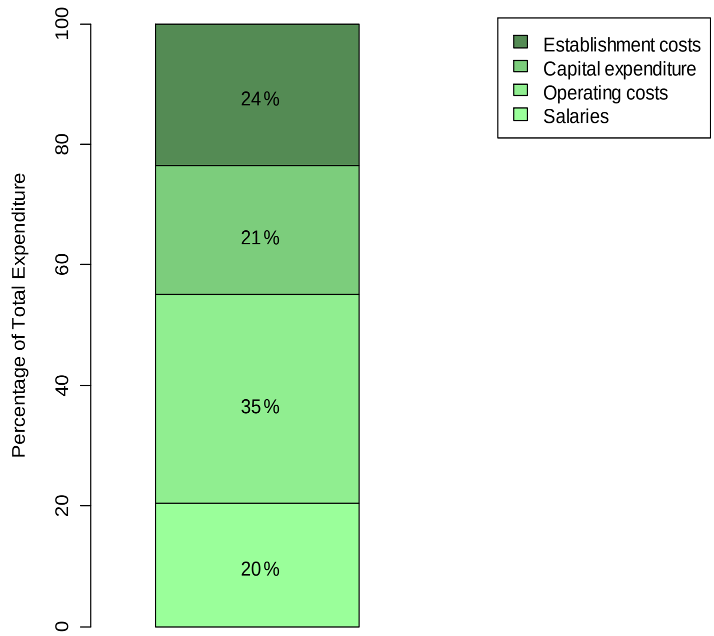


Figure 4.4. Actual protected area spending by category for all expenditure in the EAM. Percentages are shown for expenditure on salaries, operating costs, capital expenditure and reserve establishment costs. Recurrent expenditure (salaries and operational expenditure) made up 55% of all spending.

4.3.2 Spatially explicit model of actual spend

The population pressure layer for which was obtained the highest goodness of fit in simple linear models of actual spend had a σ value of 25, in which population pressure declines by 50% over 20 km and down to zero over 60 km (Figure 4.3). 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 protected areas in the reserve group (Table 4.3a). 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 $\Delta_i < 4$:

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

Equation 4.2.

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 (σ = 25); no.PA is the number of protected areas in the reserve group; medpop is the median population density within the reserve group; VRM is the terrain ruggedness index and (LAFR), (NFR), (NR) and (NP) are factor levels according to reserve type (Table 4.2).

Using population pressure resulted in better model fit than using population density within a buffer (Table 4.3b).

Table 4.3. 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 AICc value (Δi) of 1.4, when compared to the best model from the set that uses population pressure.

a)

Intercept	Population pressure[1]	Type[3] NFR	NR	NP	Median population	Number of protected areas	VRM	Log(L)	K	AIC_c	Δ_i	w_i	r^2_{adj}	n
-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)[2]	(1.00)		(1.00)		(0.84)	(0.38)	(0.40)							

b)

Intercept	Population density[4]	Total area	Human use	Median population	Number of protected areas	VRM	Log(L)	K	AIC_c	Δ_i	w_i	r^2_{adj}	n
-0.4582	3.117				-0.383	-139.0	-48.34	4	110.2	0.00	0.51	0.43	23
-2.2980	3.060					-99.91	-51.70	3	113.6	3.42	0.09	0.53	23
-2.8920	2.681						-53.29	2	113.8	3.64	0.08	0.49	23
-1.7540	2.617-				-0.272		-51.81	3	113.8	3.64	0.08	0.53	23
-0.6796	3.154		$5.9* 10^{-3}$		-0.384	-142.4	-48.30	5	113.9	3.65	0.08	0.61	23
-0.5342	3.130	$1.2* 10^{-6}$			-0.386	-139.3	-48.33	5	113.9	3.70	0.08	0.61	23
-0.4237	3.129			$-1.3* 10^{-3}$	-0.382	-142.1	-48.33	5	113.9	3.71	0.08	0.61	23
(RVI)[2]	(1.00)	(0.08)	(0.08)	(0.08)	(0.83)	(0.84)							

1. Population pressure calculated using a sigma value of 25 (σ = 25).
2. Relative Variable Importance (RVI)
3. Coefficients for National Forest Reserve (NFR), Nature Reserve (NR) and National Park (NP) compared to Local Authority Forest reserve (LAFR).
4. Mean population density within a 20 km buffer.

4.3.3 Spatially explicit model of necessary spend

In examining how to estimate necessary spend for protected areas for which there are no data, I first considered using a multiplier of actual spend to estimate necessary spend for any EAM reserve. The median proportion of necessary spend that is actually received, is 0.31 (mean = 0.43; interquartile range = 0.16 to 0.42; n = 29), which could be used with the model of actual spend to predict total necessary spend across the study area. However, this shortfall varies spatially, so that when actual spend was used 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, a spatially explicit model of variation in necessary spend was generated as a function of geographic and socio-economic variables. First the best population pressure and population density layers (each calculated using a different σ value or buffer size, respectively) were chosen. Once again, population pressure with a sigma value of 25 (Figure 4.3) maximised goodness of fit, while the best fixed-buffer population density layer was five kilometres.

Using population pressure (σ = 25; Table 4.4a) enabled better models to be built than when population density within a fixed buffer was used (Table 4.4b). Alongside population pressure, the best models of necessary spend contained number of protected areas in reserve group, total area and VRM (Table 4.4a) and the average model for the subset with Δ_i < 4 can be calculated as:

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

Equation 4.3.

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

In order to use this model to make spatially explicit predictions of the spend needed per ha for protected areas anywhere in the study region, the predictions were calculated at similar scales to the analysis. Therefore, necessary spend (USD ha^{-1} y^{-1}) was mapped at the median reserve size for the study area (9 km^2), having also verified that the reserves used in this analysis had a representative size distribution (Figure 4.2a). The effect of number of protected areas within a reserve group was controlled for by setting this parameter to be equal to one in the modelled surface (Figure 4.2b). The effect of population across the study area then becomes very clear, with the most populous areas being the most costly to conserve effectively (Figure 4.5).

Out of 23 reserve groups, 21 showed a funding shortfall (observed actual spend was less than modelled necessary spend; Figure 4.6). One NR received approximately the same amount as their modelled necessary spend, while one NR received approximately 50% more than the modelled necessary spend.

Table 4.4. 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 (Δ_i = 2.3), when compared to the model set that uses population pressure.

a)

Intercept	Population pressure[1]	Number of protected areas	Total area	VRM	Log(L)	K	AIC$_c$	Δ_i	w_i	r^2_{adj}	n
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)[2]	(0.65)	(0.44)	(0.27)	(0.14)							

b)

Intercept	Population density[3]	Number of protected areas	Total area	VRM	Log(L)	K	AIC$_c$	Δ_i	w_i	r^2_{adj}	n
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)[2]	(0.18)	(0.56)	(0.56)	(0.13)							

1. Population pressure calculated using a sigma value of 25 (σ = 25).
2. Relative Variable Importance (RVI)
3. Mean population density within a 5 km buffer.

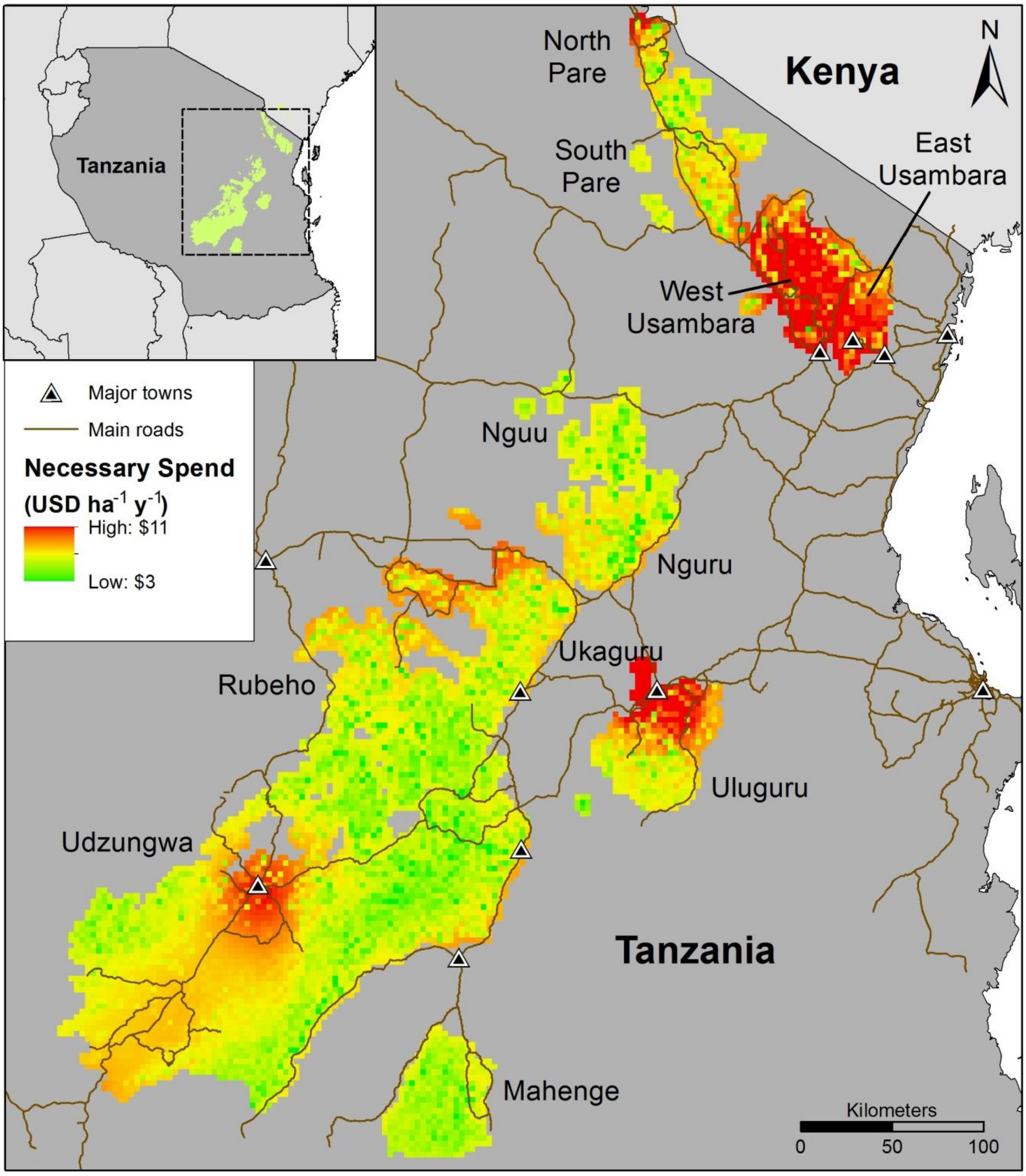


Figure 4.5. Map showing spatial variation in modelled necessary spend per hectare per year for protected areas across the Eastern Arc Mountains. Spend per hectare varies from 3 to 11 USD ha^{-1} y^{-1}. Major towns (population greater than 20,000) are also marked.

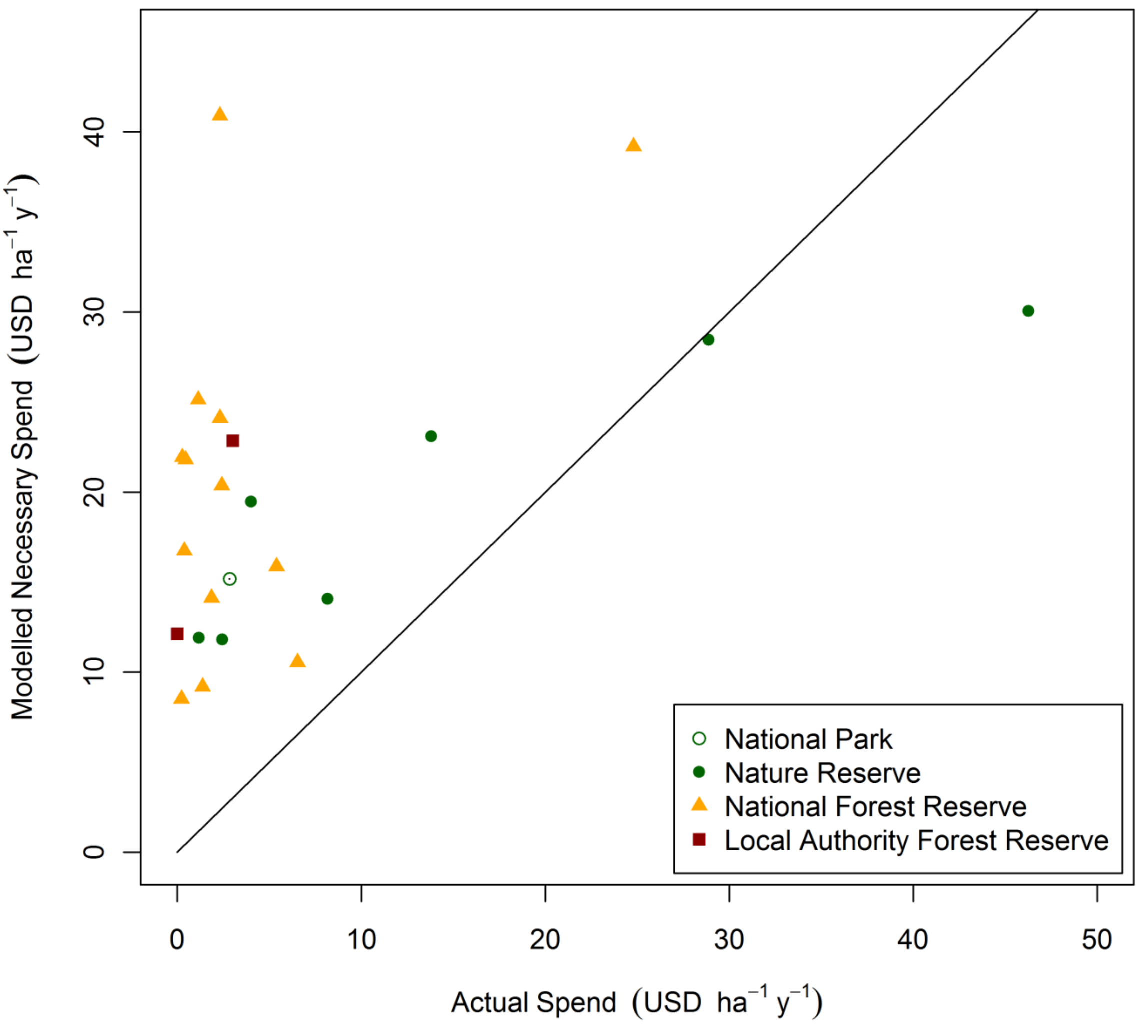


Figure 4.6. Modelled necessary spend (average weighted model; Table 4.4a) compared with actual spend (both in USD ha^{-1} y^{-1}) for 23 reserve groups. The black line indicates where actual and necessary spend are equal. Points above the line are underfunded, while those below the line receive more money than is needed according to the model of necessary spend. The symbols indicate different reserve types.

4.3.4 Reserve establishment costs

I was unable to collect many data on establishment costs; however, some data on transition costs were available where reserves have been upgraded from NFRs (under the catchment manager) to NRs (under a conservator). Such upgrading, including costs of upgrading facilities, negotiating agreements and mapping boundaries, has been common in recent years, resulting in the establishment of six NRs since 2008. The transition is usually projected to take five years, for which detailed budgets are drawn up. The median estimate of spend over the course of this transition (not including recurrent expenditure) was 45 USD ha^{-1}; mean = 49 USD ha$^{-1;}$ range = 33 to 73 USD ha^{-1}; n = 5 NRs).

4.3.5 Effectiveness

Finally, level of disturbance was plotted for LAFRs (n = 3), NFRs (n = 11) and NR (n = 1), measured as number of poles and trees cut per ha, against observed shortfall (Figure 4.7; Madoffe and Munishi 2010). Disturbance appears to decrease with increased funding. A hypothetical fourth point is also shown for necessary spend and assumes that 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.

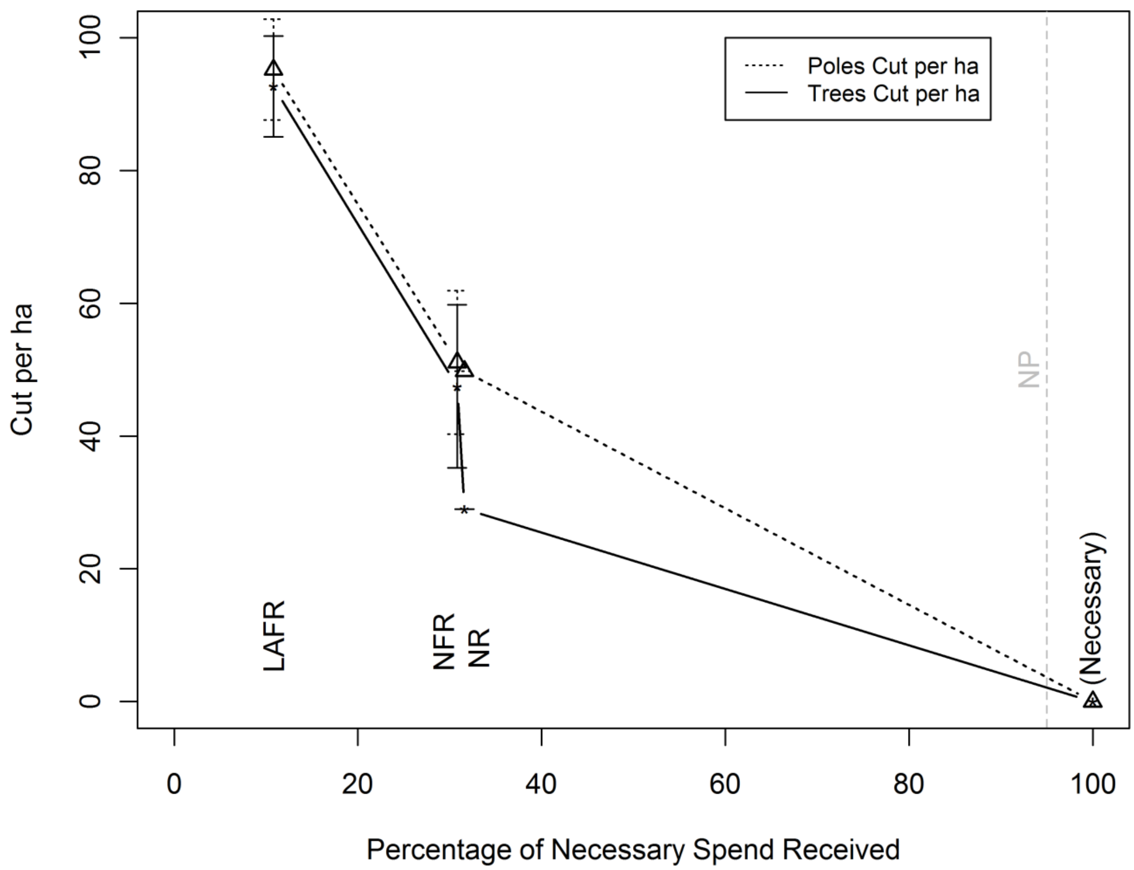


Figure 4.7. Levels of disturbance from forest surveys of observed number of poles and trees cut per ha (±1 SE; Madoffe and Munishi 2010) for Local Authority Forest Reserves (LAFRs; n = 3), National Forest Reserves (NFRs; n = 11) and Nature Reserves (NR; n = 1) plotted against observed actual median funding shortfalls (percentage of necessary spend that is received) from 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.

4.4. Discussion

Median actual spend across all sites in the EAM was 2.3 USD ha^{-1} y^{-1} (IQR= 1 to 6 USD ha^{-1} y^{-1}). Placed in context, this actual spend can be compared to the figure of 7.7 USD ha^{-1} y^{-1} that was spent in Tanzanian NPs (TANAPA 2009; data from 2007/8, adjusted for inflation to 2009), and an historical spend reported for Tanzanian NPs in 1996 of 2.5 USD ha^{-1} y^{-1} (adjusted for inflation; James et al. 1999). Actual funding across all reserve types is, therefore, around one third of NP spending in 2007/8 (although note that Udzungwa Mountain National Park, the only NP within the study region, received a similar amount of funding as NRs). The median necessary spend reported by managers was 8.3 USD ha^{-1} y^{-1} (IQR: 5 to 17 USD ha^{-1} y^{-1}), which is slightly higher than current spend in Tanzanian NPs (7.7 USD ha^{-1} y^{-1}; data from 2007/8, adjusted for inflation) and two and a half times greater than the required spend identified by James et al. (1999; 3.3 USD ha^{-1} y^{-1}, data from 1996, adjusted for inflation). However, although these estimates of necessary spend may appear high, both their median and interquartile range are well within the range of 1.6 to 62 USD ha^{-1} reported from protected areas 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; $r^2 = 0.36$, $n = 139$, $P < 0.001$) and spending per unit area was found to increase linearly (on a log-log scale) with population density by James et al. (1999). 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 (Figure 4.3).

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. No other studies have investigated in detail at the distance over which population exerts an effect and its correlation with protected area 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, it seems that pressure is more likely to drive the increased spending, particularly as the distance over which populations exert pressure (Figure 4.3: σ = 25) is consistent for both actual and necessary spend and is similar to that found in other studies of resource use in the EAM (see chapter 5).

Terrain ruggedness appeared to be negatively correlated with actual and necessary protected area management costs. I 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. Negative relationships between management costs per unit area and reserve size have been found in other studies (Balmford et al. 2003; Frazee et al. 2003; Balmford et al. 2004; Bruner et al. 2004; McCrea-Strub et al. 2010) and total area of reserve group was 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 (a useful explanatory variable for actual and necessary spend) could also lead to greater economies of scale and decreased costs per unit area. This effect may exist for actual spend but is difficult to detect, as individual reserve attributes are smeared out by the unavoidable aggregation of protected areas into reserve groups for analysis. The number of protected areas in a reserve group was also found to be 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).

Protected area type was a significant predictor in models 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. Burgess and Rodgers (2004) suggest that LAFRs in the EAM are generally managed for resource extraction, are not of particular biodiversity importance, and generally have no international IUCN protected area designation (International Union for the Conservation of Nature; IUCN 2001). Meanwhile, many of the NFRs in the EAM have been coded as category IV protected areas and NRs have been graded as category II protected areas – the same as Tanzanian NPs (Burgess and Rodgers 2004; FBD 2007). This order correlates with the amount of funding that these reserves are receiving, with higher category reserves currently receiving more funding. Although this analysis of protected area spending and forest condition (Figure 4.7) is both speculative and rough, it does suggest that management effectiveness of Tanzania's protected areas could be expected to improve under an adequately funded system.

Obviously, differences in performance are not all down to funding. Governance will also play a major role and may explain the difference between NFRs and NRs, which are modelled as having a similar level of funding currently, yet NRs have a lower number of trees cut (Figure 4.7).

This work contributes significantly to our understanding of the funding shortfalls in the current protected area 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 international money), while 73% is from central government and 5% from local government. Furthermore, the 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.

The model to predict necessary spend was less robust than that for actual spend, reflecting the small sample size and the errors associated with the unavoidably subjective assessment of how much money effective conservation would require. Despite these difficulties, models of necessary spend explained 39% of variation (weighted average; Table 4.4a). Although global and international models have been constructed, these are unlikely to perform so well at sub-national scales or for this type of reserve system (Balmford et al. 2003; Frazee et al. 2003; Balmford et al. 2004; Bruner et al. 2004; Moore et al. 2004; see introduction also).

Many studies have shown particularly strong relationships between spend per unit area and total area. It is likely that the full effect of this relationship is masked by the fact that the analyses presented here were done on reserve groups. In addition, the grouping of reserves led to the analysis being conducted at small sample sizes, which reduces the ability to see smaller but still important effects. Nevertheless, this analysis provides a realistic framework for estimating the actual and necessary costs of management in a complex system with complex funding pathways.

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 protected areas (NPs, NRs, NFRs an LAFRs) falling within the EAM, as recognised and mapped by the WDPA (IUCN 2010a), cover 17% (8,613 km^2) of the mountains' extent and my 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, this is a useful figure which 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 2007/8 (TANAPA 2009; SIPRI 2010). So, with the important caveats that management cost is only one part of the total cost of conservation (see chapter five for quantification of indirect costs: damage by wild animals and opportunity costs) and that there are more protected areas 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.

4.5. Summary

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 I collected data from protected area managers in the Eastern Arc Mountains (EAM) of Tanzania to establish how much is currently spent on reserve management and how much is required to meet conservation objectives. I 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.

The 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.

I use the 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 in conservation planning in the region and can also be used to estimate management costs under future scenarios of population growth and migration.

5. Indirect costs of protected areas

"If all land had the same properties, if it were unlimited in quantity, and uniform in quality, no charge could be made for its use, unless where it possessed peculiar advantages of situation. It is only, then, because land is not unlimited in quantity and uniform in quality, and because in the progress of population, land of an inferior quality, or less advantageously situated, is called into cultivation, that rent is ever paid for the use of it. When in the progress of society, land of the second degree of fertility is taken into cultivation, rent immediately commences on that of the first quality, and the amount of that rent will depend on the difference in the quality of these two portions of land."

David Ricardo, On the Principles of Political Economy and Taxation (1821)

5.1. Introduction

The total cost of conservation includes not only the direct cost of managing protected areas, but also the indirect costs, such as opportunity costs and damage costs (Balmford and Whitten 2003). Recognising where indirect costs are borne is of particular importance to the equitability of global conservation efforts due to the fact that although these costs largely accrue locally, many of the benefits of conservation accrue at national and global scales (Balmford and Whitten 2003; Linnell et al. 2010). Historically, this externality was rarely dealt with, even if acknowledged. More recently, however, the idea that the wider community should compensate the minority bearing the bulk of the cost has gained traction as a key tenet to payments for ecosystem services (Ferraro and Kiss 2002; Jack et al. 2008; Fisher et al. 2010). Moreover, in the political arena of applied conservation, the socio-economic context of conservation efforts is increasingly recognised as being of equivalent importance to biodiversity information in making land use decisions that are both efficient and effective (Brechin et al. 2002; Morrison and Boyce 2009). Understanding and addressing these issues of equity is, I would argue, a moral imperative of conservationists.

Systematic Conservation Planning (SCP) provides a spatially explicit framework within which these costs can be recognised and minimised, whilst also taking the benefits of conservation into account (Naidoo et al. 2006; Wilson et al. 2009; chapter one). SCP aims to meet conservation targets for minimum cost (Margules and Pressey 2000). When identifying efficient protected area networks, however, SCP exercises often consider the costs of conservation to be proportional to area and do not take their spatial heterogeneity into account (Naidoo et al. 2006; chapter one). Not only can inclusion of socio-economic data help us find efficient and equitable solutions, but it can prove particularly useful for ranking spatial options for conservation when biological data are poor or when conservation benefits vary less than the costs (Ardron et al. 2008; Ban et al. 2009). In this chapter, I describe the methods I used to map the opportunity costs of conservation and the costs of wildlife damage in the Eastern Arc Mountains (EAM) of Tanzania. These data are suitable for use as input layers to SCP analyses, where cost-efficient solutions are sought. I will describe my work to integrate both indirect costs and direct costs (previous chapter) into SCP in chapter six.

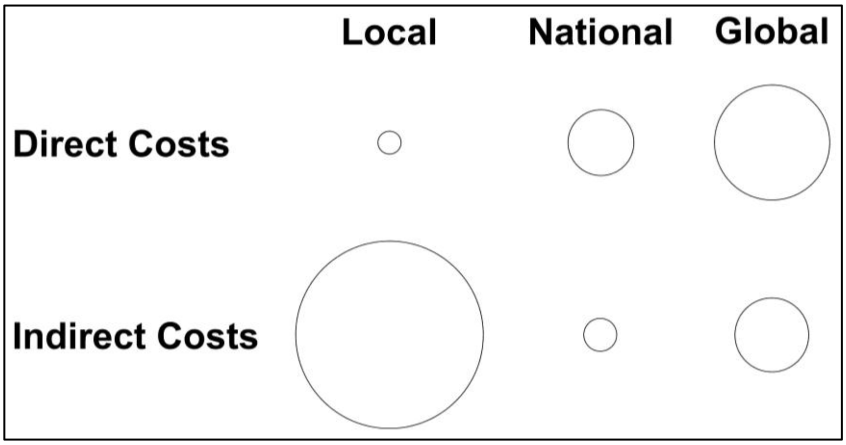


Figure 5.1. The indirect costs of conservation (including opportunity and damage costs) are predicted to be greater than the direct costs (such as management costs of protected areas) and disproportionately borne by those living closest to protected areas. Diagram adapted from Balmford and Whitten (2003).

5.1.2 Opportunity cost

The opportunity cost of conservation is equal to the benefits foregone when land is conserved for biodiversity or ecosystem service conservation and is a passive cost that accrues to land users not comprehensively compensated for loss of access to land. It can be calculated as the potential profit from the most likely alternative use of the land, net of any costs incurred in obtaining that benefit. Previous studies of opportunity cost found that farming was the most likely alternative land-use in Kenya and Paraguay (Norton-Griffiths and Southey 1995; Naidoo and Adamowicz 2006) and this is true of Tanzania too, where 80% of the workforce is involved in agriculture, accounting for 50% of the nation's GDP (Kelly et al. 2008). The main drivers of deforestation and degradation in the EAM are smallholder agriculture and charcoal production (Burgess et al. 2002a; Geist and Lambin 2002; Newmark 2002; Ahrends et al. 2010; Fisher 2010). Furthermore, cropping in highlands, such as the EAM, is expected to continue or increase in the face of climate change over the near to mid-term future (Jones and Thornton 2009).

Consideration of opportunity costs is valuable for understanding the land use decisions made by farmers and for working towards achieving more equitable and effective conservation. A study in Southeast Asia by Fisher et al. (2011a) shows that, when opportunity costs are taken into consideration, there is a considerable shortfall in the amount of money that has been proposed for direct payments for conservation. Proposed levels fall far short of the level of compensation needed if direct payments are to be an incentive to reducing deforestation. In addition, several studies have highlighted the relative magnitude of opportunity costs and how their inclusion in a systematic conservation plan will significantly alter its spatial priorities (Ando et al. 1998; Naidoo and Adamowicz 2006; Naidoo and Iwamura 2007; Adams et al. 2010; Fisher et al. 2011a). Therefore, calculating opportunity cost is important; for quantifying disparities in who pays for and who benefits from conservation and for addressing efficiency, equitability and effectiveness in SCP.

5.1.2.1 Methods for quantifying opportunity cost

Opportunity cost has been modelled at different scales, including global (Balmford et al. 2000; Naidoo and Iwamura 2007; Carwardine et al. 2008; Loyola et al. 2009), national (Norton-Griffiths and Southey 1995; Busch et al. 2012) and sub-national (Ando et al. 1998; Polasky et al. 2001; Ferraro 2002; Kniivilä and Saastamoinen 2002; Chomitz et al. 2005; Naidoo and Adamowicz 2005; Polasky et al. 2005; Chiozza 2006; Naidoo and Adamowicz 2006; Naidoo and Ricketts 2006; Ban et al. 2009; Ban and Klein 2009; Börner et al. 2009; Adams et al. 2010; Bauer et al. 2010; Bryan et al. 2011; Fisher et al. 2011a; Shaw et al. 2012; Wise et al. 2012).

One widely used source of data is land prices, both modelled and observed (Ando et al. 1998; Balmford et al. 2000; Polasky et al. 2001; Chomitz et al. 2005; Bauer et al. 2010; Busch et al. 2012; Shaw et al. 2012; Wise et al. 2012). If land is bought and sold voluntarily in an open market then, according to the Ricardian assumption that land is valued according to its potential profitability now and in the future, the price is equal to the opportunity cost (Ricardo 1821). If land price data are lacking, the productivity of land can be used as a proxy for opportunity cost. Both potential productivity (land capability; Naidoo and Iwamura 2007; Carwardine et al. 2008) and average observed levels of productivity (Norton-Griffiths and Southey 1995; Ferraro 2002; Kniivilä and Saastamoinen 2002; Naidoo and Adamowicz 2005; Polasky et al. 2005; Chiozza 2006; Naidoo and Adamowicz 2006; Naidoo and Ricketts 2006; Ban et al. 2009; Börner et al. 2009; Adams et al. 2010; Bryan et al. 2011; Fisher et al. 2011a) have been used to estimate opportunity cost. Annual returns to the land are then discounted into the future to arrive at a Net Present Value (NPV), which represents the land's market value or its opportunity cost if it is conserved. Alternatively, although the majority of studies are terrestrial and use agricultural data, the opportunity costs of other stakeholder groups, such as fishermen (Ban et al. 2009) and palm oil producers (Fisher et al. 2011a) have also been used to derive estimates of land value. Lastly, when socio-economic data are particularly scarce, surrounding population densities (Balmford et al. 2001; Ban et al. 2009) have been used as proxies for the opportunity cost of conservation and land value has even been estimated from protected area management costs, following findings that the two are correlated (James et al. 2001; Balmford et al. 2003; Underwood et al. 2008; Loyola et al. 2009).

Opportunity costs are the profit gains from the most likely alternative use of the land. For that reason, when land capability (a measure of the land's inherent potential) is used, rather than observed levels of productivity (a measure of likely productivity if the land is converted), the opportunity cost of land is overestimated (e.g. Naidoo and Iwamura 2007; Carwardine et al. 2008). Estimates of opportunity cost should also include information on currently non-productive land, which should include adjustment for the cost of converting the land. In addition, although some studies assume opportunity costs within protected areas to be zero (e.g. Polasky et al. 2001; Naidoo and Adamowicz 2006), wherever land within a protected area has not been paid for (or people perceive it has not) there is an argument for extending the analysis to estimate opportunity costs within it. It is, therefore, important to develop an approach which can model opportunity costs beyond currently farmed land and within protected areas so that policy makers can consider the opportunity costs of putting areas that are currently not farmed into a conservation network and so that they are informed about the equitability and efficiency of the existing reserve network.

In Tanzania, 80% of the population are employed in agriculture and 88% of agricultural land is managed by small-scale peasant farmers (Newmark 2002; Kelly et al. 2008). The EAM have been farmed for at least 2000 years and were seen as a major source of revenue by early colonialists, as land utility in and around the EAM is increased by the relatively high levels of precipitation found there (Farler 1879; Hamilton and Bensted-Smith 1989; Newmark 2002). Land in Tanzania has been bought and sold since pre-colonial times, and this continues despite official policy, which states that when land ownership is transferred only compensation for private property and crops is due (Hamilton and Bensted-Smith 1989). Nevertheless, these transactions are often informal and land prices are not recorded in a database; thus opportunity cost must be modelled using information on land productivity. The productivity of a land parcel under current smallholder farming practices will only reflect the value of a land parcel that has already been converted. For unconverted land, the likelihood that it will be converted in any given period should also be accounted for in estimates of future profits. Most studies use probability of conversion to farmland multiplied by the value of the land under some production system (e.g. Naidoo and Adamowicz 2006; Naidoo and Ricketts 2006). However, rather than modifying an opportunity cost based on conversion probability, here I propose and describe and new method (R. Green pers. comm.), in which the probability of conversion reveals the opportunity cost of a land parcel based on its value to agriculture and the costs of converting it.

5.1.3 Damage cost

The other indirect cost considered here is the cost of damage by wild animals. Although perceived costs of crop damage may be higher than the actual cost imposed (Balmford and Whitten 2003), experience of crop damage is correlated with negative attitudes towards wildlife, suggesting that crop damage should be a prime consideration to those trying to establish conservation areas (De Boer and Baquete 1998). In addition, with human populations densities high and expected to rise in Tanzania (Cincotta et al. 2000; United Nations 2011) and with increasingly fragmented landscapes likely to result in meso-predator release of pest species (particularly baboons), damage costs can be expected to increase (Prugh et al. 2009; Brashares et al. 2010; Mackenzie and Ahabyona 2012). Villagers living adjacent to protected areas in Tanzania expressed a desire to protect wildlife, but viewed crop-raiding animals as pests that should be killed, which is indicative of the costs that communities endure (Cunneyworth and Stubblefield 1996; Gillingham and Lee 1999, 2003). In communities bordering protected areas, farmers considered wildlife crop damage to be more limiting to their potential yields than insect pests or rainfall (Porter and Sheppard 1998; Gillingham and Lee 2003; Linkie et al. 2007; Marchal and Hill 2009).

5.1.3.1 Previous studies of damage cost

Previous work to investigate the impacts of crop damage by animals in conservation areas has focussed on medium to large vertebrate species – particularly elephants and primates. The most commonly reported species are primates (particularly baboons and vervet monkeys), bushpig, squirrels, elephants, birds, and rodents (Newmark et al. 1994; Naughton-Treves 1997, 1998; Naughton-Treves et al. 1998; Porter and Sheppard 1998; Hill 2000; Saj et al. 2001; Sillero-Zubiri and Switzer 2001; Gillingham and Lee 2003; Kagoro-Rugunda 2004; Naughton-Treves and Treves 2005; Osborn and Hill 2005; Tweheyo et al. 2005; Linkie et al. 2007; Warren et al. 2007; Marchal and Hill 2009; Priston 2009; Priston and Underdown 2009; Mackenzie and Ahabyona 2012). Across different study regions (including Tanzania) and for different damaging species, the most consistently reported predictor of damage is distance to protected areas or wildlife refugia (Mascarenhas 1971; Jhala 1993; Naughton et al. 1999; Gunn et al. 2005; Thirgood et al. 2005; Graham 2006; Kideghesho and Mtoni 2008; Nijman and Nekaris 2010; Mackenzie and Ahabyona 2012). Rather than collecting quantitative data on damage costs in the EAM directly, I use published studies from East Africa to estimate damage costs in the EAM.

5.2. Methods and results

I conducted structured interviews with farmers to collect information on the value of crops grown and the level of crop damage by wildlife. These data were then coupled with spatially explicit data on environmental variables to estimate indirect costs across the landscape. In the next few sections, I describe the data, before detailing the methods used to estimate opportunity costs and then damage costs.

5.2.1 Data

5.2.1.1 Farmer interviews

During August to October 2009, I worked with two Tanzanian students to conduct a survey of yield levels and crop damage in a representative sample of farmers in the EAM. Questionnaires were piloted through a week of farmer interviews in the Morogoro Rural district of Tanzania (Figure 1.4). The relevance of questions asked, length of survey and interviewing technique were assessed and modifications made as necessary. Farmers were asked about the year of conversion, costs of conversion, crops grown, annual yield, crop prices, input costs and crop damage by wildlife; questions were asked in Kiswahili and answers recorded in English (see Appendix D for questionnaire). Farm locations were recorded using a GPS.

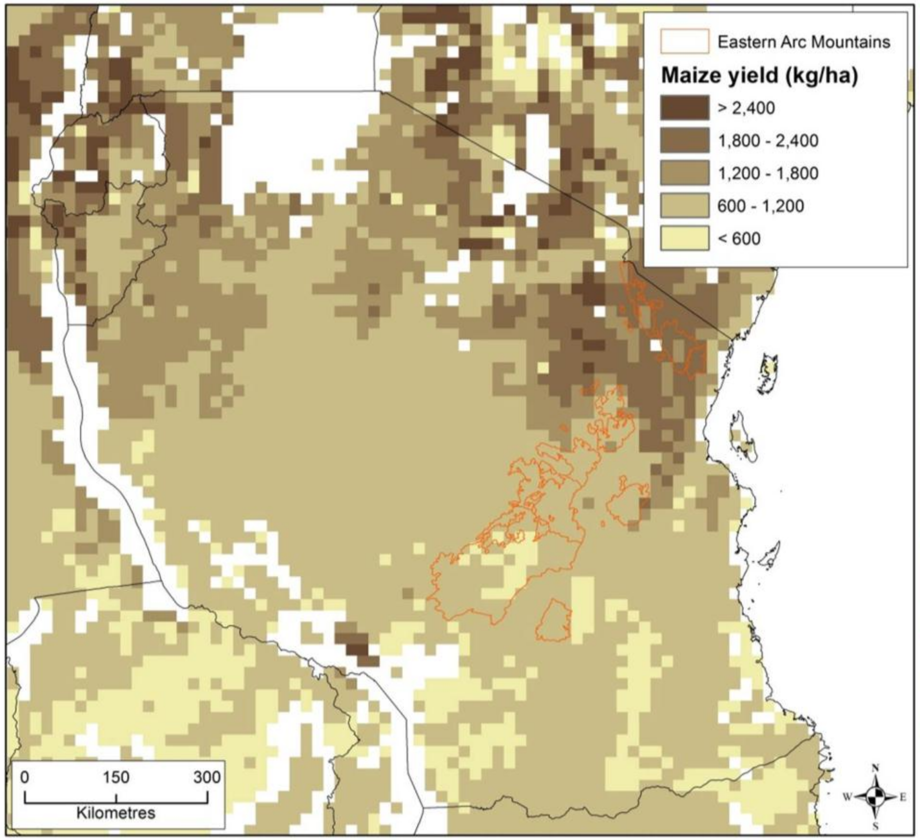
The sampling strategy for subsequent Arc-wide interviews was as follows: The entire EAM area was divided into 25 km^2 grid cells and 25 were selected at random from those which contained cultivation (in the year 2000). I assessed how well the sample represented the extremes of those variables expected to significantly affect opportunity or damage cost (altitude, rainfall, distance to roads, distance to protected areas, distance to towns and population density). For those variables whose range was not well represented in the initial random selection, cells that would help capture the underrepresented strata were identified and a random sample was chosen from these. Once the sample was suitably stratified, the village closest to the centre of each chosen cell was identified with the help of ordnance survey maps and district forest officials. In every village, two farmers from each of three wealth categories (poor, medium and rich, as assessed by the village chairman) were selected at random (the selection process was weighted by the area of land that was farmed by each individual in order to estimate values for an average piece of land, rather than for an average farmer). When a selected farmer cultivated more than one plot of land, the surveyed field was selected at random, weighted by the area of the field. Of the 25 villages, two were inaccessible and in three of them I was only able to get five interviews. In total, 135 farmers were interviewed.

5.2.1.2 Spatially explicit data

Annual Net Rent and Net Present Value of agriculture

I obtained spatially explicit data on maize and bean yield from Thornton et al. (2009; Figure 5.2). These are the highest resolution data available that show predicted yield of maize in the EAM (and across East Africa) under typical current smallholder farmer practices. Bean yield is also modelled where climatic conditions allow for a second crop to be harvested (i.e. the bimodal rainfall pattern found in the northeast of the EAM; Zorita and Tilya 2002). Annual net rent was calculated by multiplying the modelled maize and bean yields (kg ha^{-1} y^{-1}) by their market price and subtracting the costs of fertiliser, seed and labour. The data used to estimate input costs and crop values and methods to derive data layers of expected net profits for maize and bean farming are described in Appendix E. On average, land will not be farmed if net rent is less than zero. Therefore, where input costs exceed yield value, net rent is set to zero. Annual net rent was then calculated by summation of maize and bean yield to give total net returns to farming. To estimate the NPV of agriculture, annual net rent was discounted into the future. Between December 2005 and November 2010 the discount rate used by the Bank of Tanzania ranged from 3.7% to 21.4% (Bank of Tanzania 2006, 2007b, a, 2008b, a, 2009, 2010a, b). Using a long-term average is sensible in a developing country, where decision makers will take the highly variable nature of discount rates found there into account. In these analyses, the median rate of 15%, applied over 25 years, was used. Results for a low discount rate of 5% and a high discount rate of 20% are also reported. Given that private discount rates are expected to be high amongst smallholder farmers in the Eastern Arc, for whom immediate survival is likely to be more important than long-term investment (Reardon and Vosti 1995), a lower bound discount rate estimate of 5% is more reasonable than the minimum rate of 3.7% reported for this period. Using 20% as an upper estimate is supported both by the Bank of Tanzania data and by other studies of smallholder farmers in a developing country (Naidoo and Ricketts 2006; Bank of Tanzania 2007b).

Livestock were not included in these analyses, in part due to the increased complexity of doing so (increased questionnaire length, difficulties of ascribing the quantity or quality of livestock to one particular parcel of land and the fact that livestock are often grazed on communal or marginal land). Furthermore, only 3.5% of households in the Uluguru Mountains (n = 262; Hess et al. 2008) and 0.6% of households in the EAM districts of Ulanga and Kilombero (n = 177; Haule et al. 2002) had livestock (excluding fowl and pigs, which need no pasture to feed). The vast majority of the livestock subsector are pastoralists operating in the plains of central and northwest Tanzania where soils and climate combine to create conditions unsuitable for growing crops (FAO 2005; Government of Tanzania 2011).



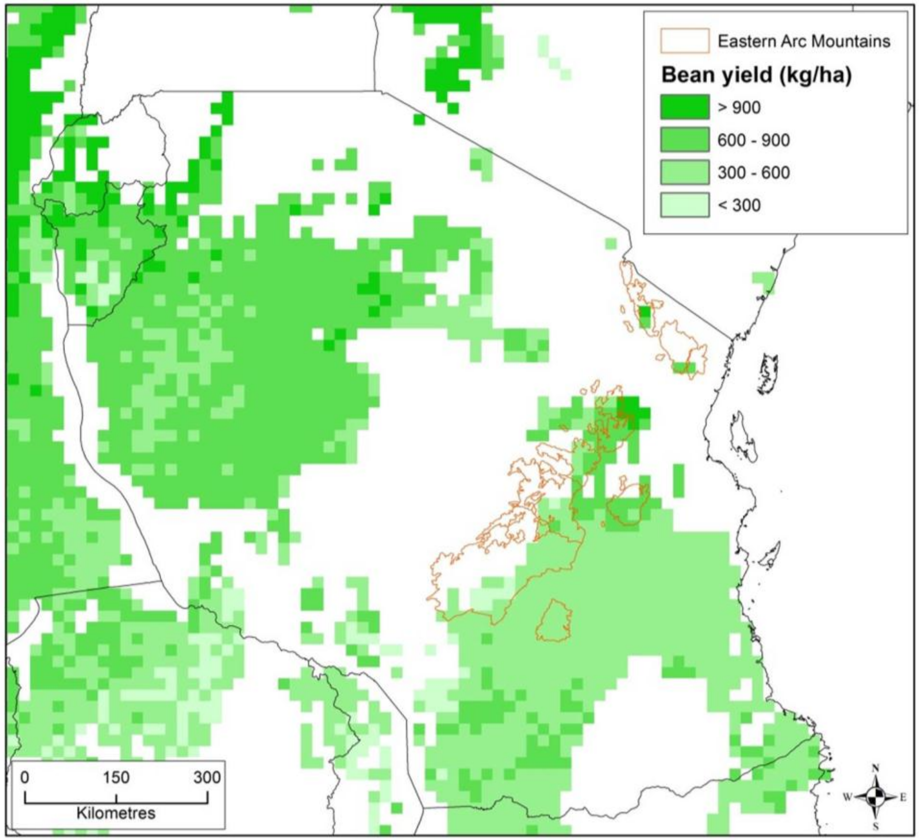


Figure 5.2. Map of maize (left hand panel) and bean (right hand panel) yields (kg/ha/y) at a resolution of ten arcminutes (approximately 18.5 km by 18.5 km) across Tanzania under current climatic conditions and for typical smallholder farmer practices (Figure adapted from Thornton et al. 2009).

Other spatially explicit data

Spatially explicit datasets used for analysis are summarised in Table 5.1. There are few data with which to assess land use change in the EAM, but the best available are from Mblinyi (2006), in which forest and woodland lost between 1975 and 2000 is mapped. Information on land use (for the year 2000), roads and markets have been compiled as part of the Valuing the Arc project (Burgess et al. 2009; Platts et al. 2011; Swetnam et al. 2011). These, together with the data on land use change since 1975, are used to derive measures of accessibility such as distance to roads, distance to markets, distance to non-natural habitat, distance to non-forest and distance to human-modified land use in 1975 and 2000. In addition, travel time to the nearest city was extracted for every land parcel in the study region using a published global dataset (Nelson 2008). Population density data are from LandScan (2006) and modified to match the 2002 census (NBS 2002) according to methods in Platts et al. (2011). Population pressure was calculated from these data using the methods described in chapter four. Data on the current protected area system are from the World Database on Protected Areas, modified to include recently designated nature reserves (IUCN 2010a; MNRT 2010). Topographic variables are derived from Jarvis et al. (2008; see also Platts et al. (2011)). Where possible, socio-economic data were obtained for both 1975 and 2000.

5.2.1.3 Data processing and analysis

Analysis of opportunity cost was done at 0.25 km^2 resolution. As well as information on forest and woodland loss, information on altitude, distance to markets, remoteness, distance to roads, terrain ruggedness, population density and population pressure was also extracted at this resolution. The derived opportunity cost layer was later resampled to a resolution of 9 km^2 - the same resolution at which damage costs are calculated. This size corresponds to the median state-owned protected area size of 8.8 km^2 and so is an appropriate size at which to conduct SCP analyses. On the other hand, protected areas smaller than this are unlikely to sustain viable populations of crop-damaging animal species and using small cell sizes to map damage costs will overestimate them. All values were recorded and analysed in Tanzanian shillings (TZS), but are reported here in United States Dollars (USD) for the year 2009, using an exchange rate of 1,450 TZS = 1 USD (Exchange Rates UK n.d.).

For clarity, the methods and results for estimating opportunity cost are presented together, followed by the methods and results for estimating damage cost.

Table 5.1. Spatially explicit data used in analyses. Reference year for datasets is usually 2000; however, because land cover change is described between 1975 and 2000, variables that are expected to have changed between these years were also estimated for 1975 where possible [square brackets].

	Variable	Description
Environmental Variables	**Altitude**	Digital Elevation Model (DEM) was from Jarvis et al. (2008) and resampled to 90 m resolution (Platts et al. 2011).
	Slope	Slope was calculated using the altitude data (at 90 m resolution) as described in Platts et al. (2011).
	Flow accumulation	A measure of water availability within an area: flow accumulation was derived from the data on altitude and slope. For every 0.25 km^2 cell, the number of cells in the landscape that flow into it was calculated.
	Distance to water	To estimate ease of irrigation, areas where flow accumulation (100 m resolution) exceeded 200 cells were classified as streams. Combined with the land cover map, I derived a lakes and streams surface. Euclidean distance to water was then calculated at 500 m resolution.
	Terrain ruggedness	Ruggedness was calculated (90 m resolution) based on the difference between the slope, altitude and contour of a focal cell and the 9 pixels in a cell's immediate neighbourhood (tool described by Sappington et al. 2007).
Socio-economic Variables	**Remoteness**	A global accessibility surface of estimated travel time (in minutes) to the nearest city (population>50,000) was used as a measure of remoteness (Nelson 2008).
	Distance to human-modified land use	Based on the current land cover map, distance to human-modified land use was calculated (cultivation, plantations, scattered crops and urban areas).
	Distance to markets	Village and town population data were compiled as part of the Valuing the Arc project (Burgess et al. 2009). Settlements with over 5,000 people were classed as markets and Euclidean distance to them was mapped at 500 m resolution.
	[Distance to non-forest]	Euclidean distance to non-forest was calculated for 1975 (based on land use change map) and 2000 (based on current land cover map).
	[Distance to non-natural habitat]	Same as previous, but for both woodland and forest together. Other natural habitats not included, as they are not mapped for 1975.
	Distance to roads	Road data were compiled as part of the Valuing the Arc project (Burgess et al. 2009). Euclidean distance to roads was mapped at 500 m resolution.
	[Land cover]	Land cover, mapped at one ha resolution, is based on MNRT (1997). The layer has been modified and is described in Platts et al. (2011). Forest and woodland cover in 1975 are based on Mbilinyi et al. (2006).
	Land use change (1975 to 2000)	Forest and woodland lost between 1975 and 2000 are mapped for 400 m^2 pixels across the EAM (Mbilinyi et al. 2006). Forest and woodland area were calculated for the years 1975 and 2000 are calculated for 0.25 km^2 cells and a threshold of 50% loss is used to map forest/woodland lost.
	[Population density]	Human population density was based on LandScan (2006) and modified to exclude populations from National Parks and Game Reserves and to match ward-level census data for the year 2002 (NBS 2002). See Platts et al. (2011) for description. Population density was also estimated for 1975 using the population growth rate data described below.
	Population growth	Ward-level data on population growth rates (1988 to 2002; NBS 2002).
	[Population pressure]	Methods for calculation of population pressure are described in chapter four. Using the growth rates data above, population pressure is calculated for 1975 and 2002 for sigma values of 5, 15, 25 and 50.
	Protected areas	Shapefiles were from the World Database on Protected Areas (WDPA; IUCN 2010a), modified to include recently designated reserves (MNRT 2010).
Value	**Annual net rent**	Methods to estimate net rent, based on maize and bean yield (Thornton et al. 2009) and input costs, are described in section 5.2.1.2 and Appendix E.
	Net Present Value (NPV) of agriculture	Annual net rent (see above) was discounted at 15% over 25 years to estimate the NPV of agriculture. Lower (5%) and upper (20%) discount rates also reported.

5.2.2 Opportunity cost: methods

The opportunity costs associated with not converting natural habitat to agricultural use are estimated using data on observed conversion rates of natural habitat outside of protected areas. These models are then applied to all areas (whether they are under statutory protection or not) to derive an estimate of the value of opportunities foregone by their reservation and how that varies spatially. Opportunity costs are modelled within protected areas under the assumption that, prior to gazettement as a protected area, the local communities were not fairly compensated (Neumann 2002; Lovett 2003); the validity of this assumption is discussed later. Natural habitat conversion to cultivation is likely when the marginal benefits of doing so exceed the marginal costs (Parks 1995; Vera-Diaz et al. 2008). I assume that agents operating rationally would convert to farmland all natural or semi-natural vegetation in the EAM that has no statutory protection provided the NPV of agriculture minus the one-off cost of conversion is greater than zero (Naidoo and Adamowicz 2005). Of the forest and woodland lost since 1975, much has been to cultivation (45% of forest lost and 33% of the woodland lost is now classed as cultivation, scattered crops or plantation forest). Of the remaining forest and woodland loss, most of the transition appears to be in the form of degradation; 17% of forest lost is now classed as woodland or grassland, while 48% of woodland lost is now open woodland, bushland or grassland. A further 38% of the forest lost since 1975 is still classed as forest in the 2000 land cover map, but has been degraded to varying degrees through natural resource harvesting and fire. Much of this is likely due to charcoal production, which accounts for a large proportion of forest and woodland degradation in Tanzania (Ahrends et al. 2010).

5.2.2.1 Land cover classification

The current land cover map was reclassified into six categories: cultivated, forest, woodland, grassland, woodland with scattered crops and other. All land use types were assigned to one of these categories (Table 5.2). Ideally, separate models would be used to estimate opportunity costs of each land cover type, but land cover conversion data are only available for forest and woodland. As these two land cover types are expected to have different costs and benefits of conversion, I modelled the loss of each one separately (see section 5.2.2.2). Given the lack of data on conversion of other land cover types, I took the following steps to estimate opportunity cost across the EAM: Bushland was treated as if it was woodland, because it is likely to have similar costs and benefits of conversion (Makundi and Okiting'ati 1995; Turpie 2000; Holding et al. 2001). The opportunity costs for the hybrid classes (bushland with scattered crops and woodland with scattered crops) were calculated as the mid-point between the NPV of currently cultivated land and the modelled opportunity cost value for woodland. Last, grassland was treated with a very simple model, based on observations that very high altitude grassland (altitude > 2000m) or grassland on high-altitude plateaus (plateaus over 1500m, as defined by Platts et al. (2011)) is generally naturally occurring and of low agricultural potential (because of low orographic rainfall, frost and nutrient leaching; Pratt et al. 1966; Newmark 2002; Finch and Marchant 2011). Hence I assumed such grassland has an opportunity cost value of zero. Other grassland (assumed to be forest-derived) is expected to have low costs of conversion, as it only exists due to human disturbance. I therefore treated it the same as cultivated land by estimating the opportunity cost of its conservation as equal to its NPV. This simple classification is supported by the relationship, for grassland patches, between the estimated NPV of agriculture (if they were converted) and altitude, which showed the majority of grassland at high altitude to be in large, unprotected patches and of low agricultural value (Figure 5.3).

Table 5.2. Land cover classes and categories for modelling opportunity cost.

Treatment	Land use types incorporated
Cultivation There are no costs to conversion, so opportunity cost is equal to the Net Present Value of agriculture. Although cultivated land is rarely included in conservation plans, it may sometimes be of value for biodiversity conservation (e.g. creating migration corridors). Therefore, it is important to estimate the value of such land.	Cultivation, monocrop unspecified, rice, sugarcane plantation, tea plantation, teak plantation, sisal plantation, plantation forest, grassland with scattered crops
Forest Opportunity cost is calculated using the forest model derived from the methods described in sections 5.2.2.2 to 5.2.2.4.	Lowland forest, sub-montane forest, montane forest, upper-montane forest, forest mosaic
Woodland Opportunity cost is calculated using the woodland model derived from the methods described in sections 5.2.2.2 to 5.2.2.4. Bushland is treated as woodland due to the fact that they are expected to have similar costs to conversion and roughly comparable charcoal benefits.	Closed woodland, open woodland, bushland
Grassland Grassland in the Eastern Arc Mountains can be naturally occurring or derived from forest through human disturbance. Natural grassland occurs at high altitudes and on highland plateaus. It is usually of low agricultural potential compared with the rest of the Arc, due to lower orographic rainfall, frost and nutrient-poor soils (Pratt et al. 1966; Newmark 2002; Finch and Marchant 2011). Therefore, grassland above 2000 m or on plateaus above1500 m (definitions in Platts et al. (2011)), is assigned an opportunity cost of zero. All other grassland is assumed to exist through human disturbance and the cost of access and conversion very low. Therefore, opportunity cost is calculated in the same way as for cultivation and is equal to the NPV of agriculture.	Grassland
Woodland with scattered crops The mid-point between the woodland model and NPV is used.	Woodland with scattered crops, bushland with scattered crops
Other Opportunity cost for these landcover types is not calculated, as they are not important to terrestrial conservation.	Urban, water

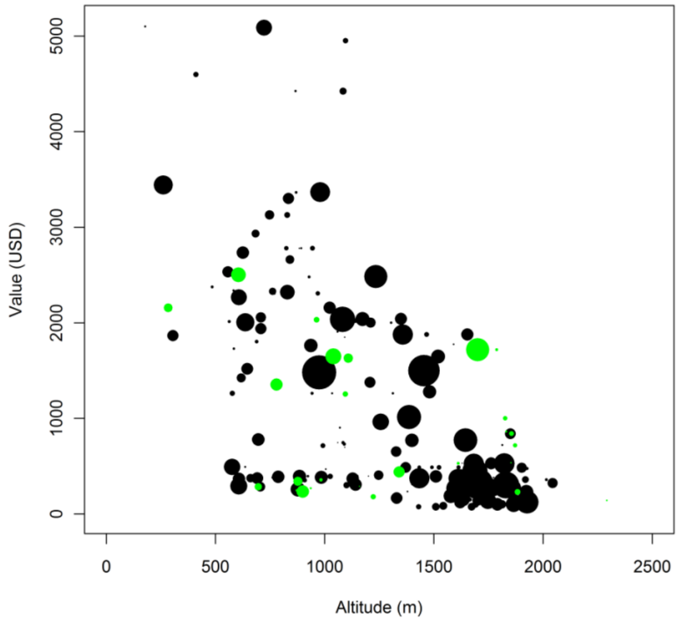


Figure 5.3. Mean Net Present Value (NPV) is plotted against mean altitude for all grassland patches in the Eastern Arc Mountains. Green circles are patches within protected areas and black are outside of protected areas. Circle size is proportional to patch area. Grassland value decreases with increasing altitude. Much of the grassland above 1500m is in large patches on high-altitude plateaus (Newmark 2002; R. Marchant pers. comm.; Finch and Marchant 2011; Platts et al. 2011).

5.2.2.2 Range of opportunity cost

The following method to estimate opportunity cost for unconverted natural habitat was developed by Rhys Green (pers. comm.), who also ran the initial calculation of opportunity cost, as described in section 5.2.2.3.

First, I determined the range within which opportunity costs should fall, which I calculated separately for forest and woodland. As a minimum bound, I assumed that the actual opportunity cost of conserving forest or woodland (including the NPV of agriculture, one-off benefits such as making charcoal and one-off conversion costs) should never be less than zero for any cell in the landscape that was observed to be converted during the period 1975 to 2000. To determine the maximum value of land, I used work by Fisher et al. (2011b) to estimate the gross benefits from charcoal harvesting of that land cover type (assuming complete clearing for agriculture in the first year), which I then summed with my values for the NPV of agriculture. The opportunity cost of conservation should, then, not be greater than the NPV of agriculture plus the gross benefits from charcoal production (i.e. with no conversion costs).

5.2.2.3 Spatial variation of likelihood of conversion

The next step was to model spatial variation in opportunity costs (NPV of agriculture minus the cost of conversion). For land already under cultivation, this opportunity cost was equal to the NPV of agriculture (Table 5.2). For land that has not been converted to agriculture, a measure of the likelihood of conversion should also be included. For instance, a parcel of land that is remote from road networks, markets and cultivated areas is less likely to be converted than a parcel of land close to a large population and adjacent to other cultivated land, even if the parcels' intrinsic land attributes and climatic envelopes mean that their expected agricultural yields are the same. This is because the costs of accessing the land and transporting the products to market are higher, so the net benefits are lower. It is assumed that the decision to convert a parcel of land was made by the farmers, who weigh up the NPV of agriculture against the costs of conversion. Therefore, probability of conversion (*P*) in a specified time period is calculated as:

$$P = q(OC),$$

Equation 5.1.

where *q( )* is a function and *OC* is the opportunity cost and I assume that *OC* = NPV - *C*, where NPV is the Net Present Value of agriculture and *C* is the one-off net cost of conversion.

Hence,

$$P = q(\mathrm{NPV} - C).$$

Equation 5.2.

If farmers were perfectly rational, profit-maximising decision makers with perfect information, then it could be assumed that *P* = 1 (parcel is converted) when NPV > C and *P* = 0 (parcel is not converted) when NPV < C (solid line in Figure 5.4). However, given that farmers do not have perfect information; that the decision to convert land and its implementation take time; that food prices can be volatile (adding uncertainty to the potential profit from agriculture); that measurements of NPV and C are imprecise; and that likelihood of conversion will decrease as uncertainty about potential benefits increases and size of the potential benefits decreases (Elhorst 1993; Ellis 1993; Parks 1995), a logistic relationship between opportunity cost and the observed proportion of parcels of land converted is assumed, where the probability of conversion is 0.5 when NPV minus C is equal to zero (dotted line in Figure 5.4).

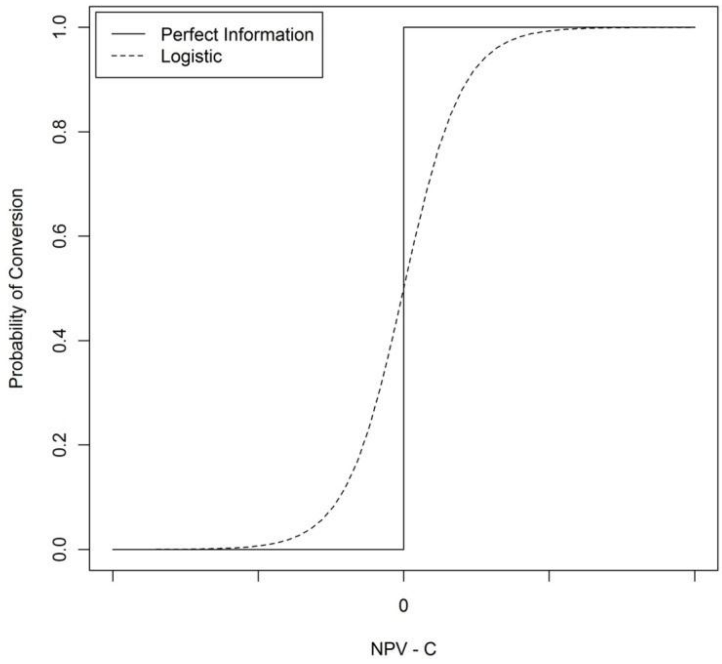


Figure 5.4. If farmers were perfectly rational, profit-maximising decision makers with perfect information on land profitability, then probability of conversion would be equal to one whenever the Net Present Value (NPV) of agriculture minus the cost of conversion (C) is greater than zero (solid line). However, because their information is imperfect, our assessments of agricultural NPV and C are imprecise and land conversion likelihood increases as potential profits increase, a logistic function best describes the expected relationship between net profit and conversion probability. Conversion probability increases as expected net profits increase.

This representation is given by:

$$log_e\left(\frac{P}{1-P}\right) = vOC,$$

Equation 5.3.

where *v* is a constant, which is the same as:

$$P = \frac{\exp(vOC)}{1 + \exp(vOC)}$$

Equation 5.4.

Expected NPV, after conversion, was estimated according to the methods described in section 5.2.1.2. Next, the one-off net cost of conversion (*C*), which is unknown, must be estimated. This net cost will include the one-off cost of obtaining access to the parcel and the equipment and labour to clear and prepare the soil. The cost of conversion will be reduced by subtracting any one-off benefits of conversion, such as the value of timber and other forest or woodland products obtained. The method estimates only the net one-off cost of conversion; it does not separate out the gross costs and benefits that determine it. It is assumed that the net one-off cost of conversion can be modelled as a function of explanatory variables which influence the costs of obtaining access, of labour and of the value of forest and woodland products obtained as a result of clearance. For example, where roads are close by, obtaining access is likely to be less costly and transport costs to markets for forest or woodland products are expected to be lower (Hymas 2001; Perz et al. 2008; Vera-Diaz et al. 2008). The logarithm of *C* was modelled as being a linear function of explanatory variables. Hence, if there were two explanatory variables:

$$C = exp(k_0 + k_1 x_1 + k_2 x_2),$$

Equation 5.5.

where $k_0..k_n$ are constants and $x_1..x_n$ are explanatory variables. This could be extended to a larger number of explanatory variables. Hence, opportunity cost *OC* is given by

$$OC = \mathrm{NPV} - \exp(k_0 + k_1 x_1 + k_2 x_2),$$

Equation 5.6.

and the probability of conversion per time period is given by

$$P = \frac{\exp\big(v(\mathrm{NPV} - \exp(k_0 + k_1 x_1 + k_2 x_2))\big)}{1 + \exp\big(v(\mathrm{NPV} - \exp(k_0 + k_1 x_1 + k_2 x_2))\big)}$$

Equation 5.7.

If there are data on the observed probabilities of conversion for a sample of parcels, we can use this model to estimate constants *v* and *k*. Having estimates of *k* in turn allows calculation of *C* for each parcel and, given that NPV is also estimated for each parcel, the two values can be used to estimate opportunity cost for any particular parcel ($_i$):

$$OC_i = \mathrm{NPV} - exp(k_0 + k_1 x_1 + k_2 x_2)$$

Equation 5.8.

A practical difficulty in implementing this scheme is that it is difficult to know how to specify the time period within which the logistic function described above can be assumed to determine the probability of conversion. Very short time periods, seconds, minutes or days, are unlikely to be realistic, given the time taken to make land clearance decisions and implement them. A time period of one year was chosen for these analyses, so the values of P in the models presented above are considered to be annual probabilities of conversion.

The available data describe changes between two land cover scenarios 25 years apart. Hence, the probability M of conversion of a parcel over the whole of this time period is given by:

$$M = 1 - (1 - P)^{25},$$

Equation 5.9.

which expands to:

$$M = 1 - \left(1 - \left(\frac{\exp\big(v(\text{NPV} - \exp(k_0 + k_1x_1 + k_2x_2))\big)}{1 + exp\big(v(\text{NPV} - \exp(k_0 + k_1x_1 + k_2x_2))\big)}\right)\right)^{25}$$

Equation 5.10.

Suppose that a series of values for *v* and *k* are guessed. For each of these guesses, this equation can be used to calculate the expected value of *M* for each parcel *E(M)* using the estimated NPV and explanatory *x* variables. The log-likelihood (LL) of the data on conversion, given the model, is then calculated as:

$$LL = \sum\big(w * log\big(E(M)\big) + ((1 - w) * log((1 - E(M))))\big),$$

Equation 5.11.

where *w* specifies for each parcel whether it was ($w = 1$) or was not ($w = 0$) converted during the 25 years and summation is performed over all the parcels of unprotected forest or woodland in the survey. Explanatory variables were only included in the final model if their deletion and refitting of the model caused LL to decline by 1.92 or more (a statistically significant decrease in model performance at the threshold of P = 0.05). An algorithm was used to determine the values of *v* and *k* that maximised LL. These parameter estimates were then used to determine OC_i for each grid cell, whether it was protected or not.

5.2.2.4 Rescaling opportunity cost

The OC_i values derived using this model cannot be taken to be real measurements of opportunity cost because of the arbitrary assumption made above about the time period over which decisions are made. Instead, they reflect relative value and so I assume that differences in modelled OC_i among pairs of parcels are directly proportional to equivalent differences in true opportunity cost. Hence, the modelled OC_i values must be rescaled to fit with the initial assumptions about the permissible range of opportunity cost (see section 5.2.2.2). This is done, separately for forested cells and wooded cells, in the following way:

1. First, I calculated the distribution of opportunity cost values for converted habitat that was not under formal protection. To do this, I obtained the modelled opportunity cost values for those cells that are not in protected areas and that were converted from forest or woodland during the study period (from Equation 5.8) and I calculated the 5th (OC_5) and 95th (OC_{95}) percentiles.

2. I then calculated the upper bound for the opportunity cost of these converted cells. To do this I summed the NPV of agriculture and the one-off benefit from charcoal (charcoal values were from Fisher et al. (2011b), standardised to 2009 dollars) to give NPV' for the same set of cells.

3. For this same set of cells, I then calculated the 5th and 95th percentiles of the distribution of NPV' values from step 2 - call these NPV'_5 and NPV'_{95}. This is from the distribution of actual opportunity cost values.

4. To convert the modelled opportunity cost values (OC_i from equation 5.8) to fit my assumptions of their actual distribution, I then re-scaled the values. For any non-converted cell of the appropriate land cover type in 2000 (including cells both inside and outside of protected areas), the modelled opportunity cost values, OC_i, are rescaled to give the estimated opportunity cost of conservation OC'_i:

$$OC'_i = (OC_i - OC_5)(NPV'_{95} - NPV'_5)/(OC_{95} - OC_5)$$

Equation 5.12.

5. Finally, if OC'_i is less than zero then it is given a value of zero. This last step is included to avoid negative opportunity costs, which would imply that local communities should be paying to maintain intact forest and woodland.

5.2.3 Opportunity cost: results

5.2.3.1 Opportunity cost in forest

The opportunity cost of any particular forested cell (OC_i) is a function of the Net Present Value (NPV; discount rate = 5%), distance to non-natural habitat (*dist.nonnat*) and population pressure with a sigma value of 25 (*PP25*). It is expressed as:

$$OC_i = NPV - \exp(22.09 + 1.8 * 10^{-4} * dist.nonnat - 9.7 * 10^{-7} * PP25)$$

Equation 5.13.

This result shows the importance of intactness in determining the likelihood of forest conversion. Forest that is close to non-forested or non-wooded land has a greater opportunity cost and a greater probability of conversion. In addition, and independently, increased pressure from surrounding populations (presumably due to increased food demand and relative land scarcity) also increases opportunity costs. To calculate the actual opportunity cost, OC_i must then be re-scaled to derive OC'_i values (see Equation 5.12 and section 5.2.2.4).

5.2.3.2 Opportunity cost in woodland

The opportunity cost (OC_i) of woodland is a function of the NPV, distance to non-natural habitat (*dist.nonnat*), distance to market (*dist.mkt*) and remoteness (*remote*). It is expressed as:

$$OC_i = NPV - \exp(18.95 + 1.7 * 10^{-4} dist.nonnat + 4.2 * 10^{-6} dist.mkt + 1.3 * 10^{-4} remote)$$

Equation 5.14.

Once again, land on the edge of forest and woodland patches has a greater probability of conversion and a greater opportunity cost. In addition, distance to markets and remoteness are important. These two variables capture slightly different factors. Remoteness is measured as travel time to large towns (district capitals) and is largely influenced by the road network. Timber and the products of commercial agriculture are commodities that are likely to be explained by this variable. Distance to markets, on the other hand, is measured as Euclidean distance to population centres of more than 5,000 people. This variable captures more of the variation in access to the markets at which the majority of small-holder farmers sell commodities such as charcoal or surplus harvest (farmer survey data). Calculation of opportunity cost is then done by re-scaling OC_i to derive OC'_i values (see Equation 5.12 and section 5.2.2.4).

5.2.3.3 Goodness of fit

It is not possible to validate these models, as land-use change data are only available for one time period and, as these data are used to calibrate the models, they cannot be used to validate them (Pontius Jr. and Schneider 2001; Pontius Jr. et al. 2004). However, goodness of fit was assessed using Receiver Operating Characteristic (ROC) curves and Area Under the Curve (AUC) values (Figure 5.5). The observed AUC values of 0.76 and 0.67 for forest and woodland, respectively, indicate reasonable goodness of fit. The analyses were restricted to cells within the EAM that were entirely forest or entirely woodland in 1975 to avoid inflation of AUC values by the models correctly predicting non-forest or non-woodland for cells that were already converted in 1975 (Pontius Jr. and Schneider 2001).

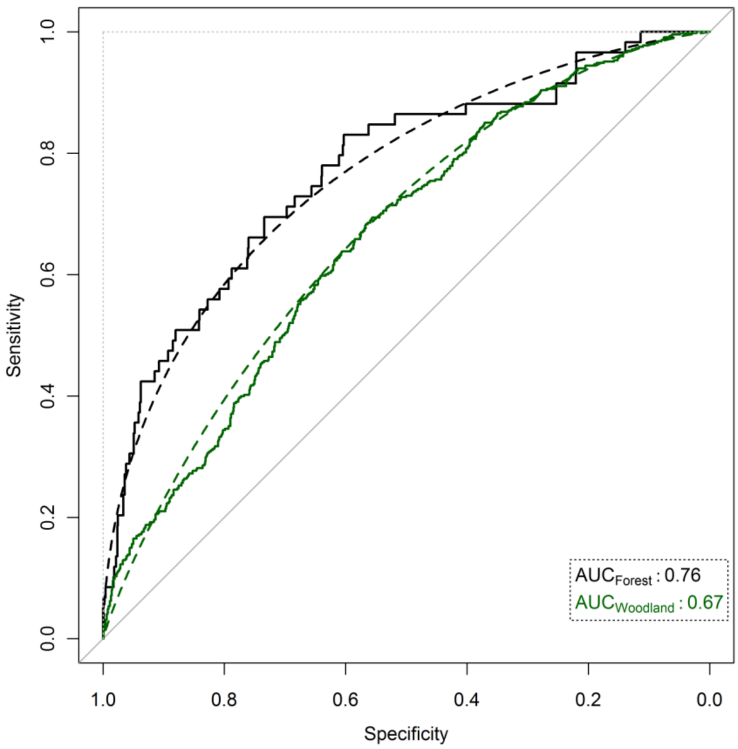


Figure 5.5. Smoothed (dashed lines) and unsmoothed (solid lines) Receiver Operating Characteristic curves for forest (black) and woodland (green). Area Under the Curve (AUC) values of 0.76 (forest) and 0.67 (woodland) demonstrate reasonable goodness of fit for modelled opportunity cost against observed conversion. Perfect model fit (AUC = 1) is plotted as a grey dotted line and a model that is no better than random (AUC = 0.5) is shown with a solid grey line.

5.2.3.4 Opportunity cost across the Eastern Arc

The total opportunity cost of the current protected area network is ~281 million USD y^{-1}, and median costs within protected areas was 264 USD ha^{-1} y^{-1}. Mapped across the entire study area, opportunity costs show enormous spatial heterogeneity, varying from 0 to 738 USD ha^{-1} y^{-1} (Figure 5.6). Overall, the median opportunity cost of conserving all cells (whether currently protected or not) is 273 USD ha^{-1} y^{-1}. Opportunity costs were lower in protected areas (Figure 5.7a), which, in the case of forest and woodland (where the difference is most pronounced), is because protected areas have significant tracts of natural habitat that are less accessible and face lower human population pressures. In grassland, the median opportunity cost within protected areas is reduced by the large tracts of grassland that occur on plateaus, which is assigned an opportunity cost of zero. Countering this is woodland with scattered crops and, to a lesser degree, cultivated areas. In these agricultural land use types, higher opportunity cost values are found within protected areas. This could be because the risks of farming within protected areas (i.e. the risk of having crops confiscated, being fined or having to pay bribes) are only justified by greater profits, which are reflected as higher opportunity values. A large proportion of the modelled opportunity cost is the profits that are realised through charcoal harvesting; modelled opportunity costs increase from cultivation and grassland (no charcoal value) through to forest (highest charcoal value).

Mountain blocs also appear to show different opportunity cost distributions (Figure 5.7b): lowest median opportunity costs are found in the Malundwe, Udzungwa, Mahenge and Rubeho Mountains, where remoteness, low population density and low fertility (Figure 5.2) contribute to lower land values. Highest median opportunity costs are found in the more populous and fertile Nguru, Pare and Uluguru Mountains, where median opportunity costs reach over \$400 ha^{-1} y^{-1}. Parts of the Ukaguru and Usambara Mountains also show particularly high opportunity costs (Figure 5.6).

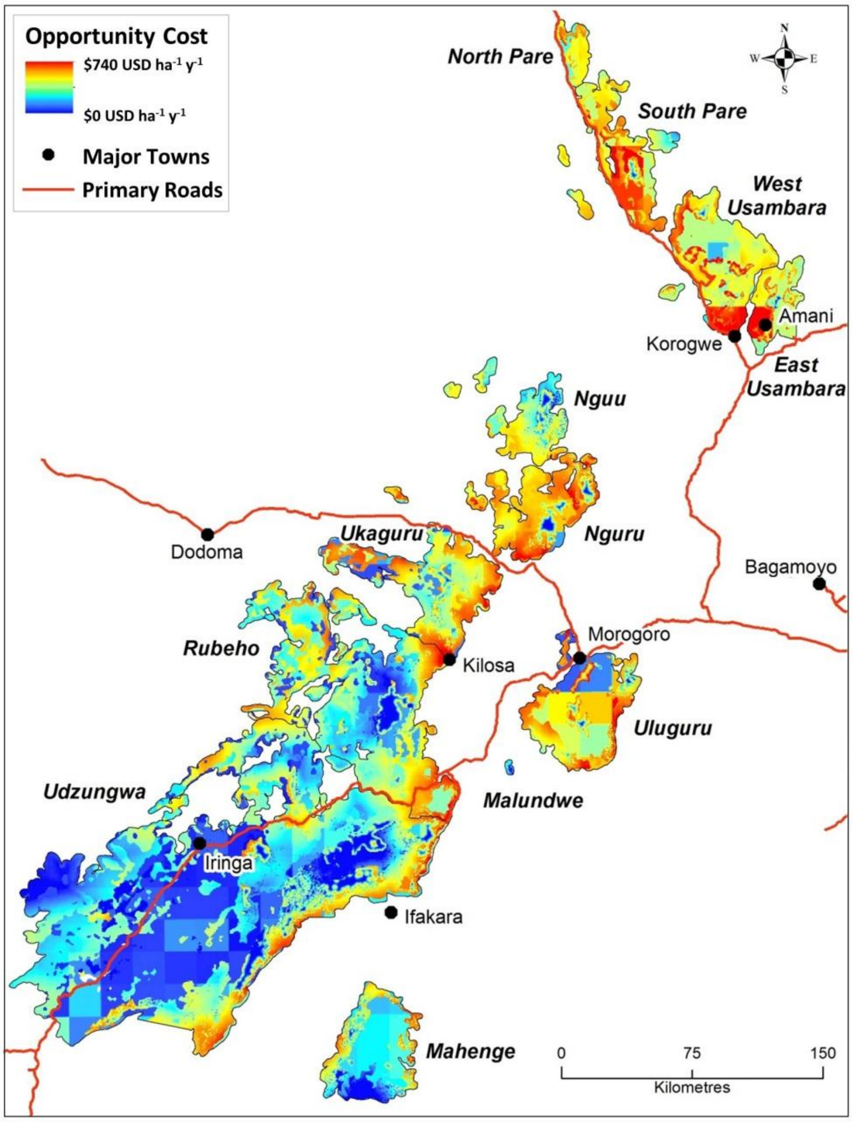


Figure 5.6. Opportunity costs in the Eastern Arc Mountains vary enormously. Highest opportunity costs (in red) occur in the populous and fertile Usambara Mountains. Large areas of land with very low opportunity cost are found in the southwest, particularly in the Udzungwa Mountains. The ten arcminute squares that show up on the map are an artefact of the resolution at which the maize and bean yield were mapped (Figure 5.2).

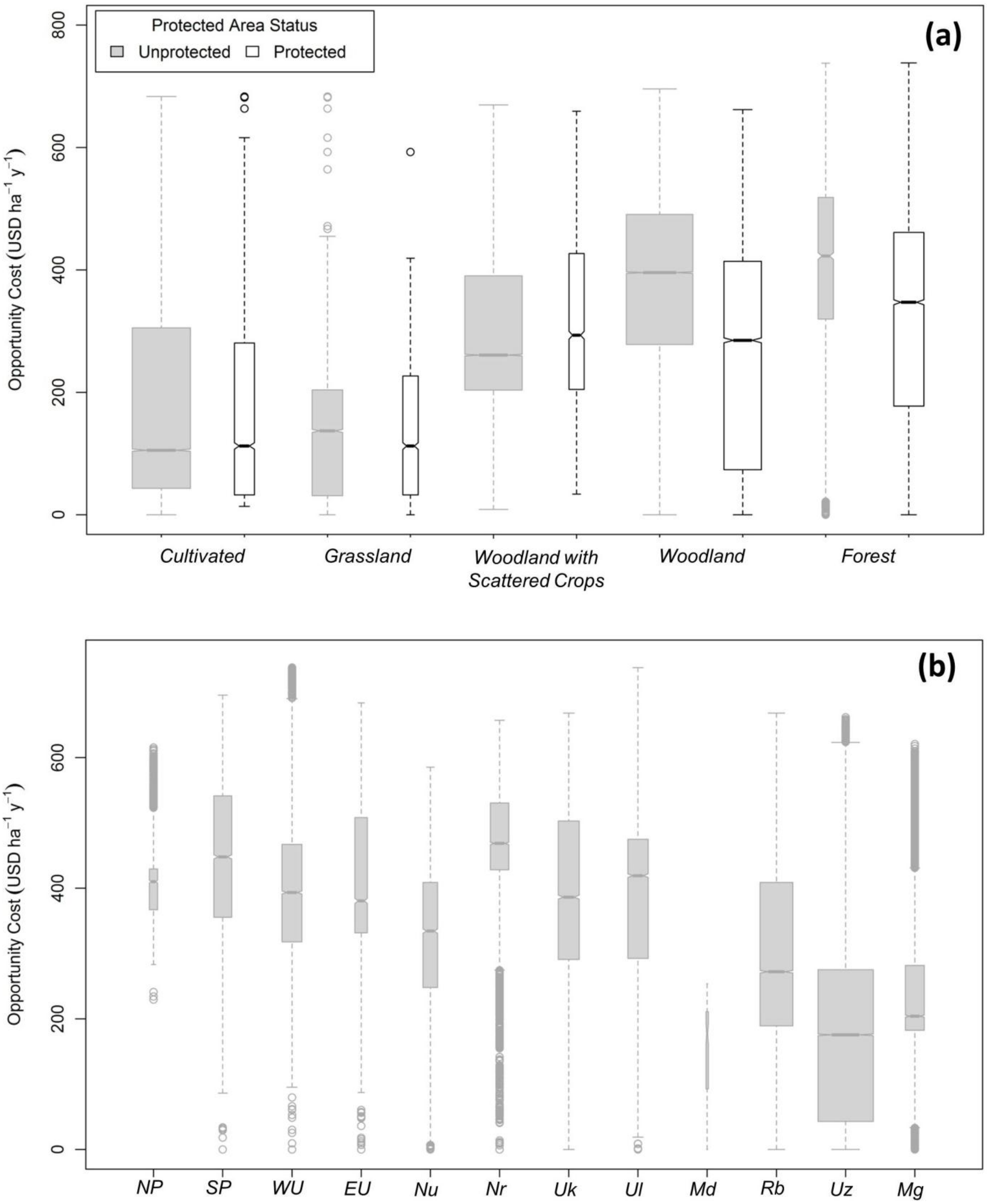


Figure 5.7. Boxplots showing the distribution of opportunity cost values for each land cover category (a) and for each mountain bloc (b). Solid bars = median; grey/white boxes show interquartile range; box area is proportional to the square root of the group size; whiskers indicate the lowest and highest points within 1.5 times the interquartile range of the upper and lower quartiles ; outliers are shown as circles. Non-overlapping notches around the medians is good evidence for significantly different distributions (Chambers et al. 1983). Generally, opportunity cost appears lower in protected than non-protected areas but increases from cultivation (lowest) through to forest (highest). Mountain blocs also appear to show significantly different opportunity cost distributions: lowest median opportunity costs are found in the Malundwe, Udzungwa, Mahenge and Rubeho Mountains, while the highest are found in the Nguru, South Pare and Uluguru Mountains. Key to codes: NP, North Pare; SP, South Pare; WU, West Usambara; EU, East Usambara; Nu, Nguu; Nr, Nguru; Uk, Ukaguru; Ul, Uluguru; Md, Malundwe; Rb, Rubeho; Uz, Udzungwa; Mg, Mahenge.

5.2.4 Damage cost: methods

As part of the farmer survey (section 5.2.1.1), data on crop damage were collected. Farmers were asked about the worst damage event they experienced, the five most recent damage events, and what, in their opinion, the most damaging species is. For each question both the crops and species involved were recorded. Alone, the results from this survey were insufficient to produce a regression model of damage cost for the EAM; farmers discounted the low levels of damage that they experienced and did not report them. As a result, I judged a questionnaire survey to be insufficient for developing a quantitative spatial model of damage costs. However, the most commonly damaged crops were maize and banana (48% and 17% of recent events) and 59% of the worst events reported involved maize, 8% banana, 7% beans, 6% cassava and 5% cocoa (n = 135 surveys). In addition, my data showed monkeys, baboons and bush pigs to be responsible for most crop damage in the EAM, corroborating other studies in East Africa and Asia (Newmark et al. 1994; Naughton-Treves 1997; Priston 2009; Table 5.3).

Table 5.3. Crop damaging species in the EAM. Respondents were asked which one species was responsible for the worst damage event in the surveyed field (column two), all the species responsible for the five most recent events (column three) and which one species is, in their opinion, the worst for crop damage (column four). A tally was kept so that every time a species was mentioned it was recorded to derive a ranking (column one) of the worst pest species [with the raw frequency given in square brackets]. All 135 farmers were asked these questions, but some farmers did not provide a response.

Rank	Worst damage event	Last 5 damage events	Worst damaging species
1	Monkey [39]	Monkey [6]	Monkey [43]
2	Baboon [17]	Baboon [5]	Baboon [23]
3	Bushpig [14]	Bushpig [3]	Bushpig [20]
4	Rat [12]	Squirrel [3]	Squirrel [10]
5	Squirrel [7]	Mongoose [2]	Cane rat / Guinea fowl [5]
6	Cane rat [4]	Bird / Cane rat / Porcupine [1]	Mongoose / Porcupine / Rat [1]
7	Guinea fowl [2]		
8	Porcupine [1]		

These results were used to validate the decision to use a series of studies by Naughton-Treves and others to estimate damage costs in the EAM (Naughton-Treves 1997, 1998; Naughton-Treves et al. 1998; Naughton-Treves and Treves 2005). These published studies were also conducted in East Africa (so farming techniques are broadly similar) and many of the species mentioned are similar to those causing damage around the EAM (Table 5.3). Most importantly though, the work describes both the size of the conflict zone around protected areas and also quantifies the expected crop damage within this zone. This is necessary if a spatially explicit map of expected damage costs for protected areas is to be derived. Other studies quantify damage amounts or describe how these vary with distance to PAs, but none do both. These studies, which I used in my analyses, suggest that over 90% of wildlife damage from a protected area occurs within 200 m of its boundary. Within this conflict zone, a yield loss of 7% can be expected, although yield losses can be as low as 4% and as high as 10% (Naughton-Treves 1997, 1998; Naughton-Treves et al. 1998; Naughton-Treves and Treves 2005). In this study, I therefore estimate damage costs at 7% of yield, but report results for these lower and upper bounds too.

For consistency with my estimates of opportunity cost I focus on damage to maize and beans. Moreover, according to my farmer survey, maize is by far the most frequently grown (63% of fields sampled) and is the most frequently and severely raided crop in the EAM: 61% of the worst damage events were to maize, followed by banana at 8% and beans at 7%. Quantifying damage to beans is useful for estimating damage costs in high yielding areas that have two growing seasons and are, consequently, able to cultivate both maize and beans in a single year's cycle. As for opportunity costs, I used modelled maize and bean yields (Thornton et al. 2009) together with data on crop prices and input costs to derive a surface describing annual net rent of agriculture (see section 5.2.1.2). In order to map only current yield, this was clipped to currently cultivated areas (taken as bushland with scattered cropland, grassland with scattered cropland, woodland with scattered cropland, rice, unspecified monocrops, sugarcane plantations and cultivation). This was then used to quantify losses of 4%, 7% and 10% of yield within 200 m of the current PA system. In addition, the damage cost that would result from protecting any cell in the entire study area was estimated by quantifying yield loss within 200 m buffers of every 9 km^2 pixel in the EAM.

Human injury was not included in this study due to its unpredictable and infrequent nature and the paucity of studies that describe its spatial occurrence with respect to PAs. Moreover, the EAM have relatively low numbers of dangerous animals and human injury or loss of life was never mentioned in my interviews of 135 farmers.

5.2.5 Damage cost: results

A 7% yield loss within 200 metres of protected areas gave a median modelled damage cost for currently listed reserves in the World Database on Protected Areas (WDPA; IUCN 2010a) of 4.7 USD ha^{-1} y^{-1} (Figure 5.8; range = 0 - 82 USD ha^{-1} y^{-1}; *n* = 180 reserves; median cost for 4% and 10% yield loss = 2.7 USD and 6.7 USD ha^{-1} y^{-1}). Summed across the current reserve network of 10,540 km^2, damage costs are estimated at 2.1 million USD y^{-1} (lower and upper estimates at 4% and 10% of yield: 1.2 to 3 million USD y^{-1}).

Using 9 km^2 pixels to map how damage costs would vary across the EAM if any given pixel was protected, the median modelled damage cost for a 7% yield loss is 1.7 USD ha^{-1} y^{-1}

(Figure 5.9; range = 0 - 15.9 USD ha^{-1} y^{-1}; n = 6,670; median cost for 4% and 10% yield loss is 1 and 2.4 USD ha^{-1} y^{-1} respectively). Figure 5.9 shows relative variation in damage costs across the landscape, but it is not intended to show the true cost of damage, as it assumes a healthy population of damaging species within each cell and assumes that all four sides of the cell are boundaries, across which damaging species will foray onto cultivated land (this is discussed further in chapter six).

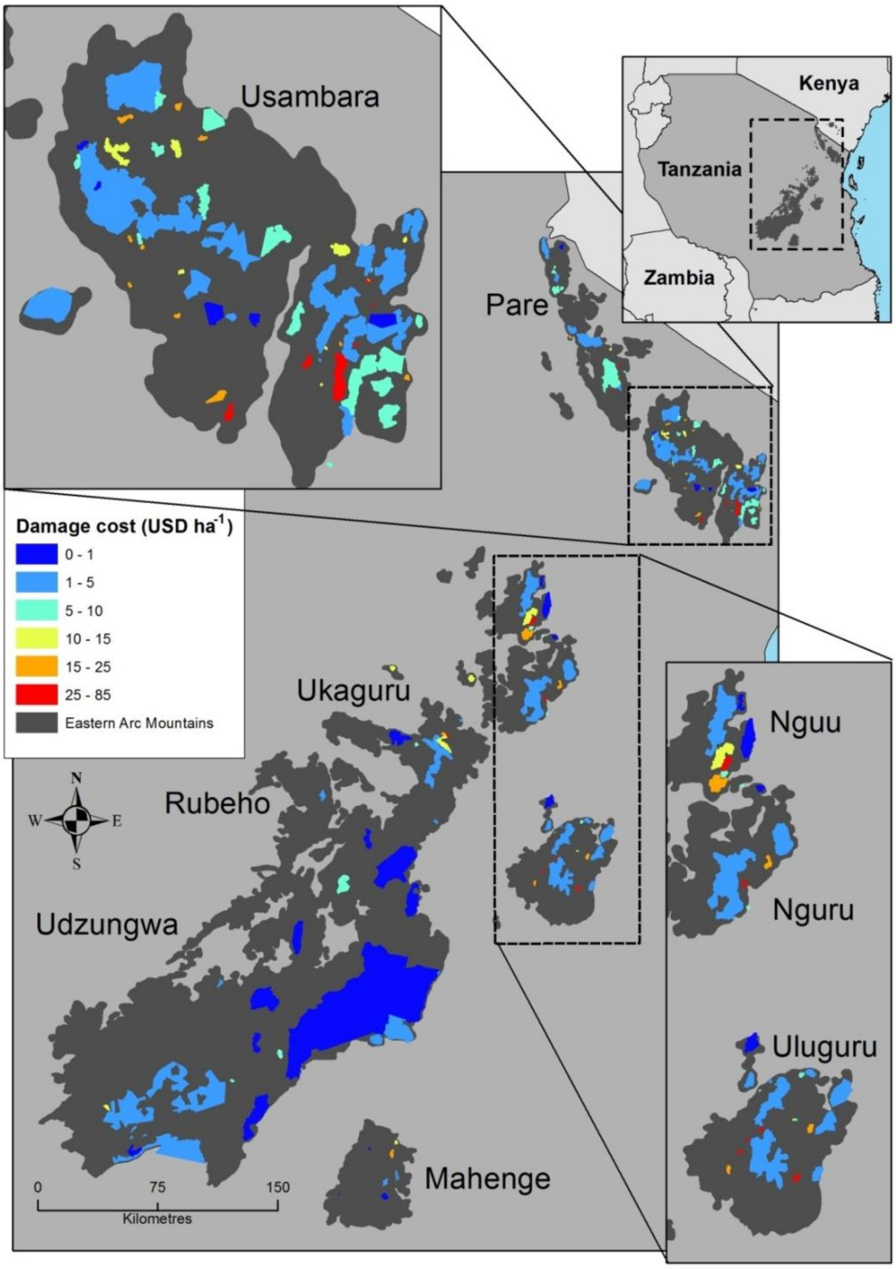


Figure 5.8. Damage costs per hectare for the current reserve network. Highest damage costs are expected in reserves in the Nguu and Usambara Mountains, where land productivity around the protected areas is currently greatest.

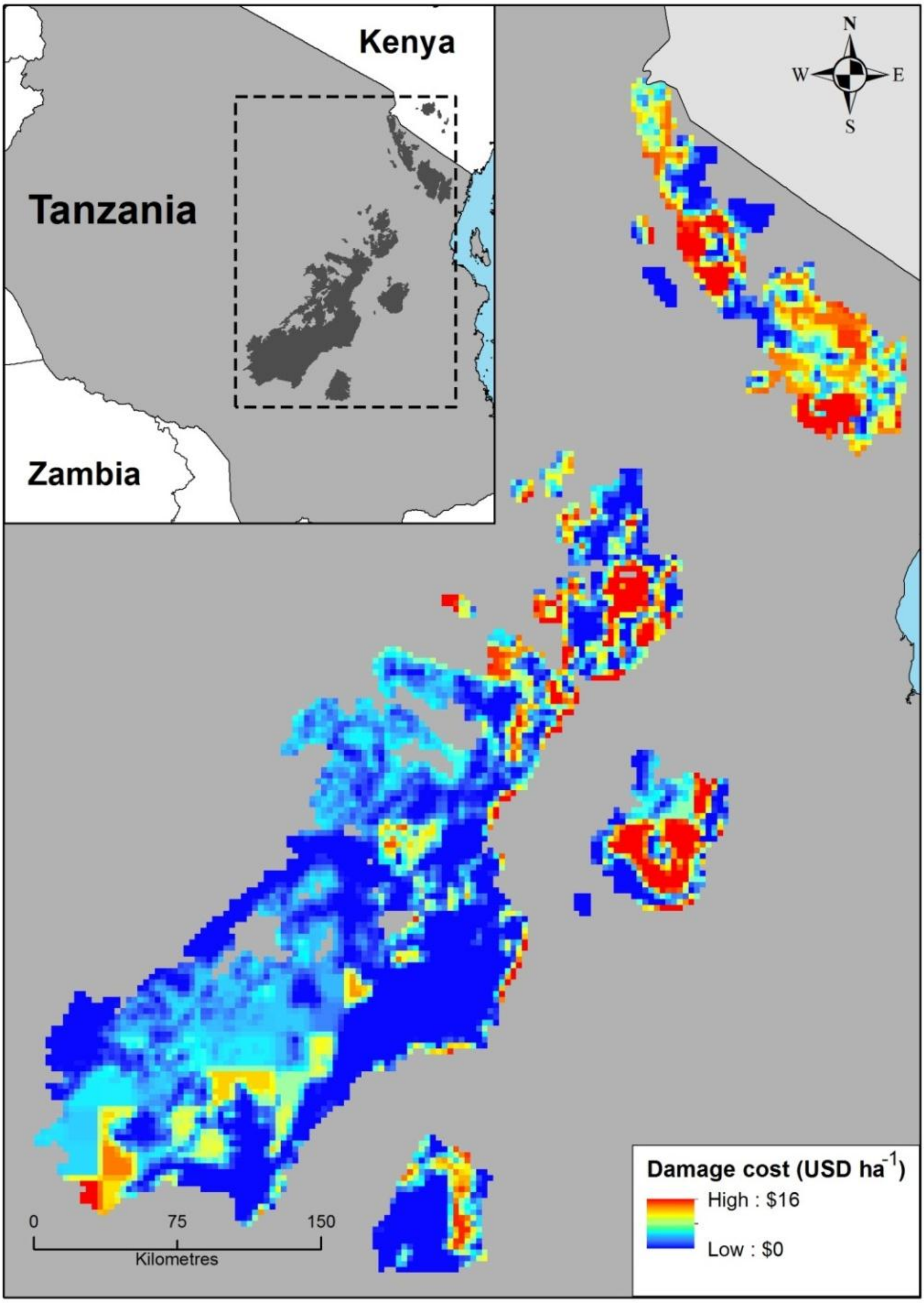


Figure 5.9. Damage costs per hectare if any 9 km^2 pixel was conserved. Highest costs are found in the productive Pare, Usambara, Nguru and Uluguru Mountains where much of the land is under cultivation and yields are high.

5.3. Discussion

5.3.1 Opportunity cost

Inclusion of a measure of the opportunity cost of conservation in conservation planning is crucial to the equity, efficacy and efficiency of conservation efforts. In the EAM, the opportunity costs of unconverted forest and woodland are found to vary from zero to over 738 USD ha^{-1} y^{-1}. Such variation, over 2 orders of magnitude, has the potential to significantly influence conservation priorities.

Elsewhere, population density has been found to have only a limited association with deforestation (Rudel 2007; Perz et al. 2008; Rudel et al. 2009; DeFries et al. 2010); however, the analyses presented here substantiate Fisher's (2010) findings that smallholder agricultural expansion, driven by land value and local population (distance to local markets and population pressure) and facilitated by accessibility (travel time to cities and distance to non-forested or non-wooded habitat), are dominant drivers of deforestation in the EAM. A major advantage to the methods described here is that they use readily-available information to map revealed preferences (expressed as land conversion). Important to this work has been the use of models that emphasize economic factors, which are crucial to generating insights into land use decisions (Perz et al. 2008; Vera-Diaz et al. 2008).

5.3.1.1 Limitations

In this method to estimate opportunity costs, three important assumptions are made and should be considered carefully. The first is the relationship between conversion probability and opportunity cost (NPV of agriculture minus a cost of conversion). This is assumed to be a logistic function, with the probability of conversion passing through 0.5 when opportunity cost is zero (Figure 5.4). In reality, farmers do not have perfect information upon which to base their decisions and farmers are also responding to the risk and uncertainty that is inherent in agricultural investment and which plays an important part in their decision making (Parks 1995). Furthermore, the level of risk taken by individuals depends on a variety of factors, not least of which are personal experience and attitude. Therefore, the curve may take a different shape and is likely to trend towards positive; however, with the limited data available, determining the curve's shape and exact location along the x-axis is not possible. However, re-scaling the values to fit with known ranges of opportunity cost in the region does help to minimise the effect of violations to this assumption. The second assumption is that farmers make the decision to convert a piece of land on an annual basis. This assumption can shift the absolute opportunity cost values (derived in Equation 5.8), but not their distribution relative to one another. If it is assumed that the decision to convert is made every week, then the values calculated in Equation 5.8 are significantly lower than if the decision is assumed to be made just once in 25 years. Assuming that decisions to cultivate are made on an annual basis is reasonable; cropping cycles take place on annual cycles and land conversion is likely to follow a similar pattern. The influence that this assumption has on the absolute values, however, necessitates that they are rescaled to fit within the range of known opportunity costs. Lastly, this range, within which spatial variation in opportunity cost is rescaled, could be inaccurate. Although a minimum opportunity cost of zero is reasonable, the maximum, which assumes that all of the charcoal benefits of conversion are realised with near-zero costs, may be unrealistic.

Another limitation to these analyses has been data availability. The only data on natural habitat conversion within the EAM in recent years are limited to forest and woodland at two points in time, 25 years apart. Such few data on land cover change prevent model validation and mean that predictions must be treated with caution. In addition, having socio-economic variables, such as protected areas and roads, mapped for the start of the period over which deforestation models are validated and for the present, against which deforestation predictions can be made, would be very valuable. It is also possible that both drivers and rates of deforestation have changed significantly during this time (although Fisher (2010) suggests that this is not the case). Nevertheless, in the absence of such data on land cover change and on the change in socio-economic drivers over time, the methods described are a pragmatic way in which opportunity costs can be estimated.

Aside from these, there are other factors that this model does not account for. The method used to estimate opportunity cost is based on a hedonic price function (where likelihood of conversion is proportional to the price that might be expected in a free market) that predicts land value in non-converted and protected areas (Bolt et al. 2005; Jack et al. 2009). Poor model performance can arise as a result of hidden costs or benefits that are difficult to observe and not captured in the socio-economic variables upon which the model is calibrated (Jack et al. 2009). Indeed, there are likely to be risk preferences, time preferences, option values, cultural values and subjective beliefs that operate but are unobservable (Parks 1995; Jack et al. 2009).

The opportunity cost calculated here is calculated as the profit from the most likely alternative use of the land. This considers only the perspective of a single stakeholder group: namely, crop farmers. If conservation interventions are based entirely on this opportunity cost, it may result in a disproportionately large adverse effect on other stakeholder groups, particularly when the land that is valuable to them is land that is of low value for agriculture (Adams et al. 2010). For instance, because these analyses exclude livestock, they are likely to have underestimated the utility of marginal land that might be of higher value to pastoralists. Although smallholder farmers are an important and widespread group in Tanzania, consideration should also be given to other stakeholders, particularly those that are marginalised, before conservation interventions are implemented. Furthermore, although opportunity cost is predicted to be approximately equal to the value of the land, it does not account for farmers willingness to sell (Guerrero et al. 2010). The cost of purchasing the land could be higher - for instance, the farmer may receive utility from his lifestyle as a farmer that he would not receive if he were in another form of employment or if he were farming in another place. Therefore, opportunity cost cannot be assumed to be a direct proxy for equitable compensation.

Finally, these analyses model opportunity costs in protected areas and in converted land. This is based on two assumptions: First, those who originally used the land in the past (prior to gazettement as a protected area) did not sell the land in a fair market and were not fairly compensated. Under colonial rule, land was claimed by the crown and, following independence in 1961, land largely became state-owned under the new Republic (Haule et al. 2002; Neumann 2002; Lovett 2003). Although customary rights of access were recognised and mandated under colonial rule, the gradual tightening of policy and increasing restrictions on land use have resulted in the loss of land use rights without fair compensation ever being provided (Haule et al. 2002; Neumann 2002; Lovett 2003). In the majority of cases, therefore, this assumption is valid. The second assumption is that if currently converted land were to be incorporated into the protected area network, then the land could no longer be farmed. This assumption is reasonable: for the species of conservation concern in the EAM, few, if any, depend on cultivated land; on the other hand, restoring cultivated land to be a potential corridor might be vital to allow for ecological processes to continue (see chapter three). Both these assumptions should be considered prior to any conservation implementation.

Despite these caveats, the median opportunity cost of 273 USD ha^{-1} y^{-1} found in the EAM is less than one tenth of the 2,380 USD ha^{-1} y^{-1} opportunity cost (annualised) reported from oil-palm and logging profits in Borneo – an area expected to have particularly high opportunity costs (Fisher et al. 2011a). Although median opportunity costs in the EAM are greater than the global mean of 68 USD ha^{-1} y^{-1} reported by Naidoo and Iwamura (2007; adjusted for inflation to 2009 USD), these global estimates do not consider charcoal value – an important commodity in the EAM. In addition, median opportunity costs within the EAM fall well below the 8,020 USD ha^{-1} y^{-1} global upper estimate (Naidoo and Iwamura 2007; adjusted for inflation to 2009 USD). Furthermore, both the median (273 USD ha^{-1} y^{-1}) and maximum (738 USD ha^{-1} y^{-1}) opportunity cost values fall within the range of land prices recorded from around the Udzungwa Mountain bloc (23 to 856 USD ha^{-1} y^{-1}; A. Marshall pers. comm. – land value annualised using 15% discount rate over 25 years).

5.3.2 Damage cost

Damage costs under the current reserve network are estimated at 2.1 million USD y^{-1}. Furthermore, there is high spatial heterogeneity, making them a potentially important consideration for increasing efficiency of conservation planning. These calculations are based on studies of actual cost; however, perceived cost might be much higher. High perceived costs can lead to unfavourable public opinion of conservation agencies and can have negative repercussions for conservation efforts; unfavourable public opinion can lead to increases in the costs of law enforcement, public relations, education, compensation and subsidies, while negative attitudes and an unwillingness to cooperate with protected area authorities make implementation and justification of conservation efforts much more difficult.

Although I did not quantify them, it is important to bear in mind the costs associated with damage by wild animals other than crop losses (Bell and McShane-Caluzi 1986; Ogra 2008; Mackenzie and Ahabyona 2012). Examples are costs associated with loss of life and trauma, costs of loss of other property (e.g. livestock and fencing), the opportunity cost of the time the farmer must devote to guarding his crops (although see Mackenzie and Ahabyona (2012)), the potentially enormous social cost of missed education (both because of the time that children must spend guarding crops rather than at school and because their household cannot afford school fees due to loss of income through crop raiding; Mackenzie and Ahabyona 2012), and even increased likelihood of mosquito-borne disease through being outside at night guarding crops, rather than inside, away from disease vectors (Bell and McShane-Caluzi 1986; Ogra 2008; Mackenzie and Ahabyona 2012). Finally, when the amount of crop damage varies greatly between years and a farmer has no insurance against complete crop destruction, an average annual cost will mean little to him (Yudelman et al. 1998).

5.3.3 Cost comparison

These results can be compared with the direct costs of conservation (chapter four) to confirm Balmford and Whitten's (2003) predictions that the indirect costs of conservation far outweigh the direct costs. This set of analyses suggests that the annual cost of the current reserve network in the EAM can be valued at 6.5 million USD in management costs, 2.1 million USD in damage costs and 281 million in opportunity costs.

5.4. Summary

Indirect costs of conservation are predicted to be larger than direct costs and to accrue disproportionately to communities living adjacent to protected areas. The cost of foregone opportunities for agricultural and charcoal production when land is set aside for biodiversity conservation is expected to be high in the Eastern Arc Mountains. Damage costs, although not expected to be so great, are also important due to their link to negative perceptions of wildlife and conservation. I estimated both costs across the current reserve system and for each 9 km^2 pixel in the landscape, if it were used for conservation. Variation in opportunity cost was related to the expected value of land under agriculture and to the cost of conversion – less populous and less accessible areas are more costly to convert. Both opportunity cost and damage cost are concentrated in the Usambara and Pare Mountains in the north. Large areas of the more remote Mahenge, Rubeho and Udzungwa Mountain blocs show much lower indirect costs. Overall, these analyses confirmed earlier predictions: the indirect costs dwarf the direct costs by two orders of magnitude. In addition, both opportunity and damage costs are highly heterogeneous across space, demonstrating that their inclusion as part of a Systematic Conservation Planning exercise has great potential for increasing the efficiency of conservation in the region.

6. The Influence of costs and processes on conservation priorities

"In preparing for battle, I have always found that plans are useless, but planning is indispensable"

General Dwight D. Eisenhower (1890 – 1969)

6.1. Introduction

Systematic Conservation Planning (SCP) is an approach to spatial prioritisation that seeks to minimise the cost of representing a group of conservation features, usually chosen as surrogates for total biodiversity, within a hypothetical reserve network. It has arisen because of the need to invest conservation funds efficiently and is particularly useful when funds are limited and when efficiency is likely to vary spatially because either costs or conservation values are spatially heterogeneous. Although the long-term persistence of species is the implicit aim of SCP, Ecological and Evolutionary Processes (EEPs) essential for generating and maintaining diversity are rarely included in priority setting analyses. Without explicit consideration of EEPs, protected area networks may, in the long-term, experience inexorable declines of the species that they were established to conserve (see chapter three; Balmford et al. 1998; Burgess et al. 2006). In addition, although efficiency is the cornerstone upon which the approach was founded, and despite the fact that conservation costs per unit area can vary dramatically across a landscape (Balmford et al. 2000; Balmford et al. 2003; Moore et al. 2004; Naidoo et al. 2006; Polasky 2008), area is often used as a proxy for cost. Aside from improving the efficiency with which limited conservation funds are spent, enhanced information on the costs of conservation may reduce the cost burden placed upon local people thereby helping to minimise negative attitudes towards wildlife conservation and helping to increase protected area effectiveness (see chapters four and five). The effect of including EEPs and costs in SCP analyses should, therefore, be tested.

A summary of the data compiled and how they will be incorporated into a comprehensive SCP analysis is shown in Figure 6.1. I assembled these data for the Eastern Arc Mountains (EAM), an area of high biodiversity and ecosystem service value facing significant threat (see chapters one, two and five; Brooks et al. 2002; Burgess et al. 2002a; Newmark 2002; Burgess et al. 2007c; Burgess et al. 2009; Ahrends et al. 2010; Fisher 2010; Swetnam et al. 2011). I mapped species of conservation concern and the EEPs that will help ensure their long-term persistence. I also derived spatial layers of the costs of conservation (protected area management costs, opportunity costs and wildlife damage costs), enabling efficiency to be examined. Finally, because small-holder agriculture and charcoal production are major drivers of deforestation and degradation in the EAM (Burgess et al. 2002a; Geist and Lambin 2002; Newmark 2002; Ahrends et al. 2010; Fisher 2010) and the potential profits from these two activities should correlate with likelihood of natural habitat loss (chapter five; Ahrends et al. 2010; Fisher 2010), I used the opportunity costs of conservation (the foregone profits from agriculture and charcoal production) as a metric of threat with which to investigate the scheduling of priorities (Margules and Pressey 2000; Pressey and Taffs 2001; Noss et al. 2002; Lawler et al. 2003; Possingham et al. 2009b).

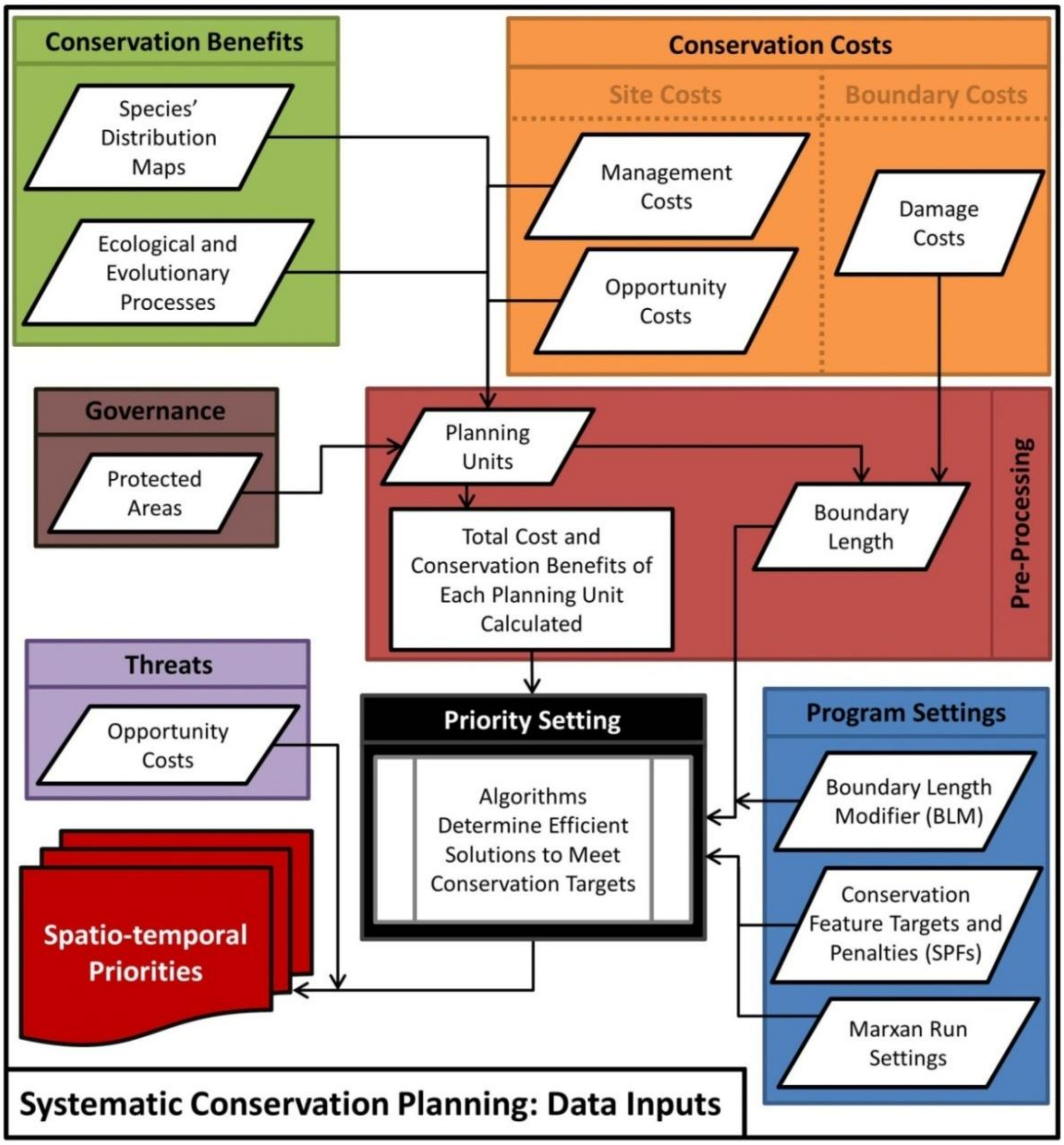


Figure 6.1. Data inputs to a Systematic Conservation Planning (SCP) analysis. Within planning units (claret) that are often based on the existing reserve network (brown), the conservation benefits (green) and conservation costs (orange) are calculated. Program settings (blue) include setting the conservation feature targets and the Species Penalty Factors (SPFs; penalties applied to the algorithm if targets are not met). Algorithm settings and the relative contribution of damage costs (via the Boundary Length Modifier) can also be adjusted. Once efficient priorities have been identified, information on threats (purple) can be used to identify temporal priorities; important areas that are under a high degree of threat may require more immediate action than areas of high priority that face lower threat. This adds a further level of complexity and real-world applicability to the analysis.

In this chapter I pay particular attention to the spatial and financial constraints within which SCP operates. As spatial constraints increase, I expect a decrease in the efficiency savings achieved through incorporating cost data. In the results and discussion, I begin with the least restrictive of the analyses before moving on to those with more spatial constraints. I compare the cost of a reserve system that represents species of conservation concern for a near-minimum area with a system that aims to represent those same species for a near-minimum cost (hereafter referred to as minimum-area and minimum-cost approaches, respectively). I then consider more constrained analyses by adding targets for EEP conservation into the priority setting analyses. Adding further spatial constraints, I then include the current protected area network to conduct a gap analysis. This considers conservation priorities under area-minimising and cost-minimising approaches using the current protected area network as a starting point from which to identify the most complementary areas for species and EEP conservation. Having considered spatial constraints, I also show how financial constraints affect the size of the efficiency savings of incorporating cost data and how the size of this effect is dependent upon the spatial constraints. Last of all, there is a danger that by focussing attention away from expensive areas the more imperilled sites will be afforded less conservation effort. Therefore, in the final analysis I use data on threat (using opportunity cost as a proxy) to investigate the scheduling of conservation action.

6.2. Methods

The data used in these analyses are described in the preceding chapters. The distributions of 504 species of conservation concern (chapter two) and proxies for nine EEPs (chapter three) are used as measures of conservation benefit. The costs of conservation used here are the direct costs of protection (whether or not a cell is currently protected; chapter four), the estimated opportunity costs of its conservation (an estimate of profits from charcoal production and agricultural rents, net of costs of land conversion) and the associated wildlife damage costs (both chapter five). Opportunity cost was also used as a proxy for threat. All costs are reported as USD per hectare per year, unless stated otherwise. Opportunity costs and management costs were simply summed to give the cost of acquiring and managing a site. In all analyses, opportunity costs were calculated for cultivated areas and protected areas (chapter five). Although cultivated areas are generally of limited conservation value in the EAM, there are occasions when their inclusion in a reserve network may be beneficial, such as when large mammal migration corridors cut through cultivated areas. Opportunity costs were assigned to current protected areas as it is useful to identify the relative costs of protected areas and their irreplaceability on a consistent scale (when compared to other protected areas or to unprotected planning units within the EAM).

Incorporating damage costs into an SCP workflow is novel and I describe it in detail here because it necessitates significant manipulation of input files. Populations conserved within a protected area will primarily raid crops grown along its border (chapter five); therefore, damage costs should only apply along external reserve boundaries and only when crops are present (Figure 6.2). Using simple values of damage cost per 9 km^2 cell (such as those in Figure 5.9) that ignore whether their neighbours are protected would overestimate the damage costs of a reserve network.

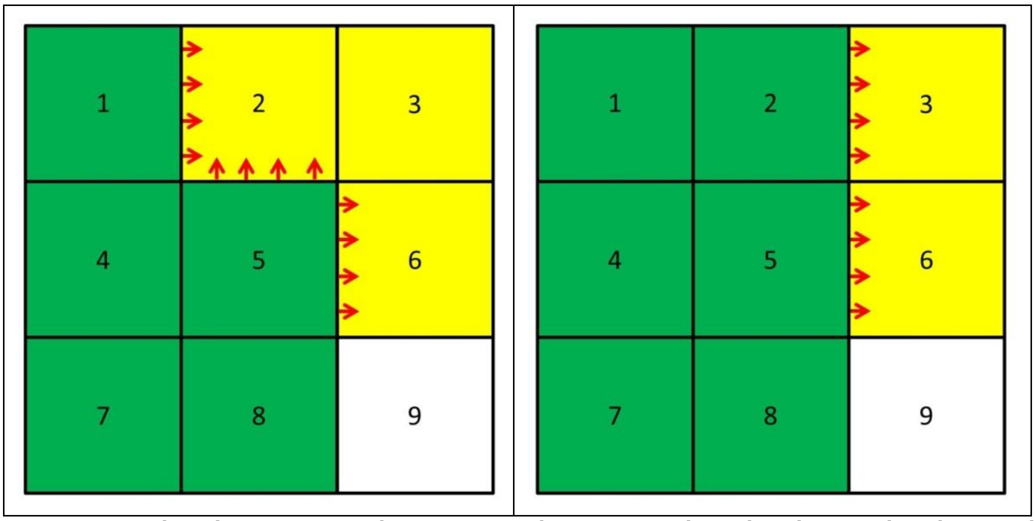


Figure 6.2. Incorporating damage costs into Systematic Conservation Planning. In the above schematic (left-hand panel), only those parts of the reserve (in green) boundary that are adjacent to farmland (in yellow) should have a damage cost attributed to them (red arrows indicate wildlife incursions to croplands). Therefore, the borders 1-2, 5-2 and 5-6 should have a combined cost equal to that of the yield that is expected to be damaged in cells 2 and 6. Boundaries that are internal to the reserve system (e.g. boundary 1-4) do not incur a boundary cost. If, under a new scenario, cell 2 was incorporated into the reserve (right-hand panel), then a new boundary cost would be calculated for boundary 2-3. Under this new scenario, however, the cost of the boundaries 1-2 and 2-5 would become internal and would be reduced to zero. The new damage cost that would be calculated would be offset against the cost of acquiring and managing cell 2. Damage costs for the border 8-9 should not be calculated for either scenario, as no crops are grown in cell 9.

In order to incorporate damage costs in my analyses I employed the boundary cost feature in Marxan (chapter one; Possingham et al. 2000; Ardron et al. 2008). This feature was developed to enable users to develop more aggregated solutions. By assigning costs to boundaries, SCP algorithms will attempt to minimise them, which results in the clumping of planning units to generate solutions with lower edge to area ratios. In order to utilise this feature for the inclusion of damage costs in my analyses, I took the following steps:

1. The length of the shared boundary (SB) between every planning unit was calculated using the boundary length calculator tool developed by ABPmer for ArcGIS 10 (ABPmer 2011).
2. The damage costs calculated for the two planning units on either side of each boundary were extracted (in 2009 USD y^{-1}).
3. The damage costs for the boundary between cells x and y were calculated as:

$$DC_{xy} = SB\left(\frac{dc_x+dc_y}{2}\right),$$

Equation 6.1.

where DC_{xy} is the damage cost between cells x and y, SB is the length of the shared boundary (in metres) and dc_{xy} *and* dc_{xy} are the average damage costs (per metre) of cells x and y respectively.

Damage costs were calculated in this way for every boundary so that the boundary cost, which is entered as an input to Marxan, was equal to the damage cost. Minimising damage costs in SCP analyses was then implemented using the standard method for minimising boundary costs (Possingham et al. 2000; Figure 6.1; Ardron et al. 2008). The Boundary Length Modifier (BLM; see chapter one and Figure 6.1) is a multiplier that can be used to weight the values ascribed to boundary costs relative to the site costs of a planning unit (Stewart et al. 2003). In my analyses both boundary costs (i.e. damage costs) and site costs (i.e. protected area management costs and opportunity costs) were measured in the same currency (2009 USD ha^{-1} y^{-1}). Therefore, the BLM was set to be one (BLM = 1) for all analyses in which damage cost was included, so that damage cost did not receive greater or lesser weight than other costs. The only analysis for which this was not the case is for the investigation into the effect of the BLM size on the overall cost of the final solution, in which the values for the BLM are stated in the results. This analysis was conducted to examine the sensitivity of the results to damage costs, which are significantly lower than either opportunity or management costs (chapters four and five). I discuss whether there are factors that justify the use of a higher BLM value to increase the relative importance of damage costs in assigning efficient priorities. Aside from this analysis, all damage costs are unweighted (i.e. BLM = 1).

SCP analyses were run in Marxan (Ball and Possingham 2000), which identifies near-optimal protected area networks that represent conservation targets for a minimum cost. Simulated annealing with iterative improvement was used to run analyses and one million iterations with ten thousand temperature decreases (periodically allowing the solution to decrease in efficiency so that it does not become trapped at sub-optimal reserve designs) were performed for each of 100 or 1,000 runs, as stated in the text (details on how Marxan works are given in chapter one). For species with an Area of Occupancy (AO) of less than 100 km^2 representation targets were set at 100%, while those with an AO of 100 km^2 to 1,000 km^2 received a target of 100 km^2 and those with an AO of greater than 1,000 km^2 received a target of 10% of their AO. For spatially flexible EEPs, targets were set at half of the area that was identified as being most suitable for their conservation. Targets for species and EEPs are described more fully in chapters two and three. Planning units were based on 9 km^2 grid cells (3 km x 3 km; the median reserve size in the EAM) and the current system of protected areas (IUCN 2010a). Three planning unit scenarios were considered: First, 9 km^2 cells only. Second, the current protected area network was included as planning units and those that did not contribute to conservation targets could be removed. Third was the same as the second, except that protected areas were fixed in the solution and the areas most complementary to the current protected area network were identified (see Figure 1.1).

6.3. Results and discussion

6.3.1 Efficient conservation of species pattern

Median species richness per 9 km^2 planning unit was compared across mountain blocs (planning units at the edge of the study region that were truncated and thus less than 9 km^2 were excluded from these analyses). Median species richness per unit area in the Usambara Mountains, particularly the East Usambara Mountains, is consistently higher than the EAM median (Figure 6.3). Other mountain blocs show lower median species richness, although several mountain blocs, particularly the Nguru, Uluguru and Udzungwa Mountains clearly show some areas of very high richness per unit area (Figure 6.3). On the other hand, when cost is considered, the Usambara Mountains, although high in species richness, also show above-average costs of conservation (measured as the sum of opportunity, management and damage costs and expressed in USD ha^{-1} y^{-1}; Figure 6.4). In these analyses, the Udzungwa and Rubeho blocs appear to provide greater opportunities for efficient conservation.

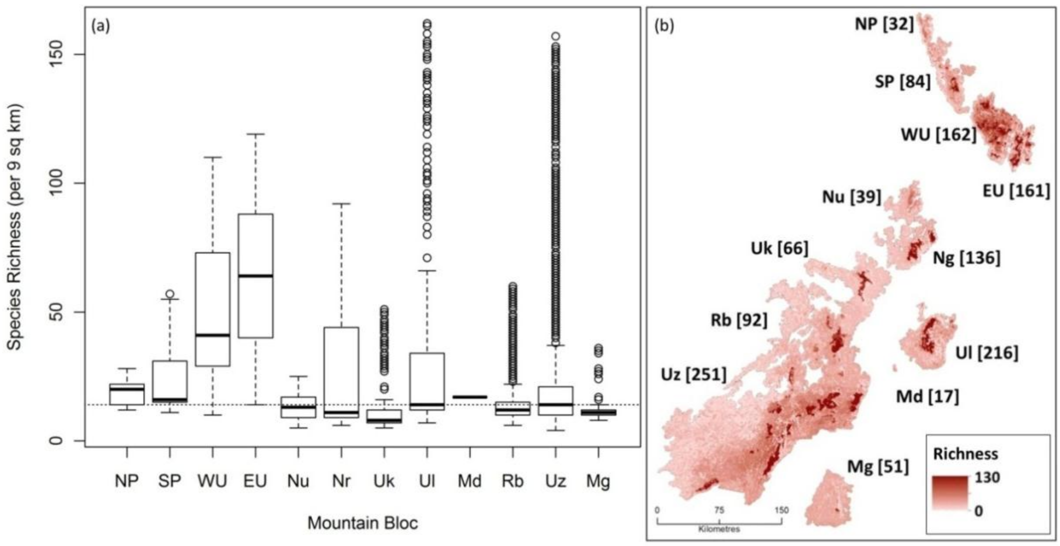


Figure 6.3. Species richness per unit area. (a) Bar plots show the distribution of species richness per 9 km^2 cell for each mountain bloc. For each bloc, solid black bars show median; boxes show interquartile range; whiskers show points within 1.5 times the interquartile range above and below the upper and lower quartiles; circles show outliers. Median species richness is also plotted (dotted line, 14 species). Median species richness is highest in the Usambara Mountain blocs but the Uluguru and Udzungwa Mountains also show areas of high richness. (b) Fine-scale patterns of species richness at their native resolution of 1 km^2 (light to dark red) are shown with mountain bloc locations. Total bloc richness is also given [in square brackets]. Mountain bloc codes are as follows: NP, North Pare; SP, South Pare; WU, West Usambara; EU, East Usambara; Nu, Nguu; Nr, Nguru; Uk, Ukaguru; Ul, Uluguru; Md, Malundwe; Rb, Rubeho; Uz, Udzungwa; Mg, Mahenge.

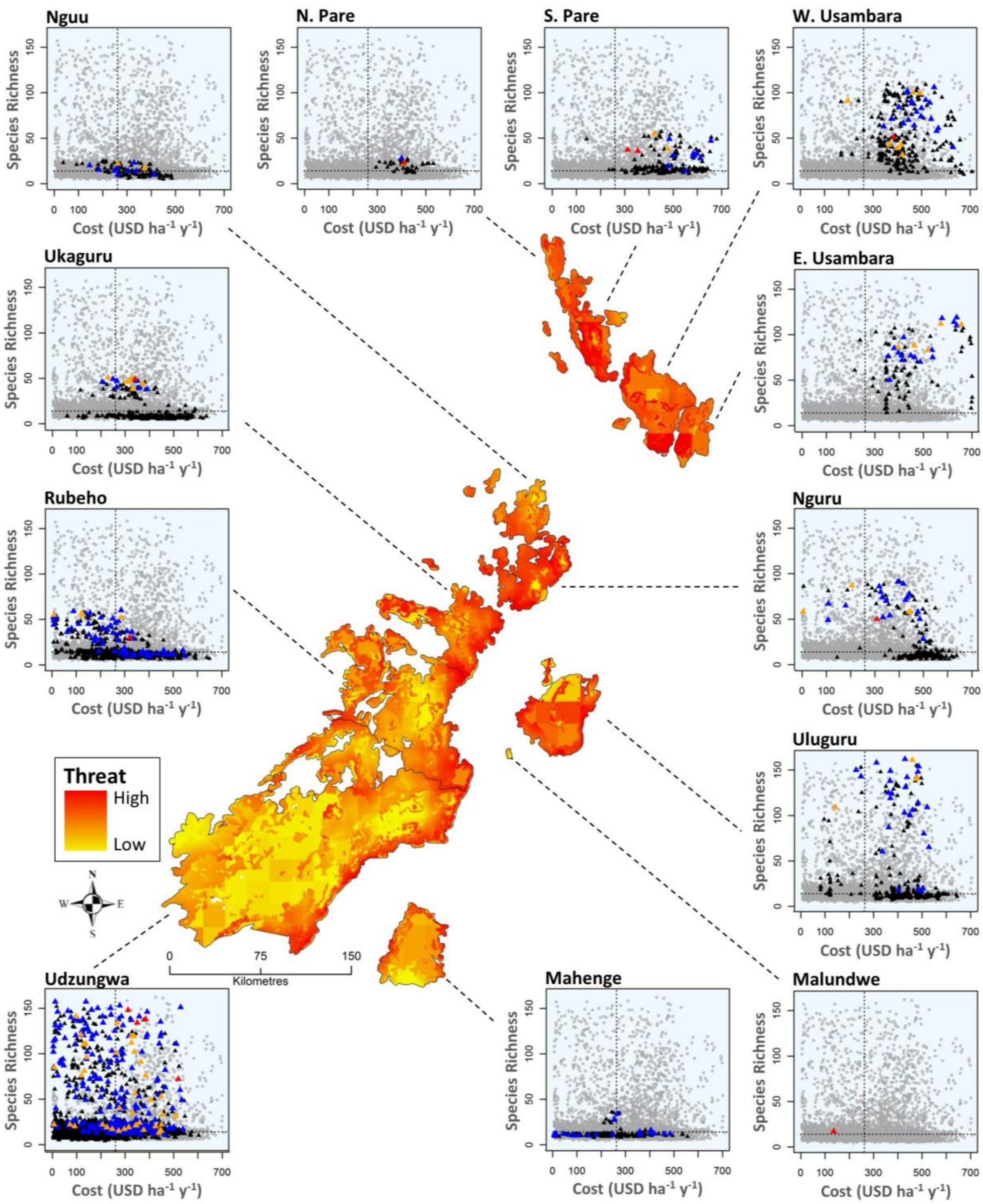


Figure 6.4. Spatial summary of data inputs. Threat (yellow to dark red; directly proportional to opportunity cost) is mapped for the Eastern Arc Mountains (EAM). Scatterplots show species richness per 9 km^2 planning unit (y-axis; chapter two) plotted against the cost of conservation (sum of management costs, opportunity costs and damage costs; see chapters four and five). Dotted lines show median richness (14 species) and median cost (260 USD ha^{-1} y^{-1}). For each mountain bloc, black and coloured triangles show planning units within that bloc relative to the entire EAM (grey circles). The contribution of each planning unit to conservation of Ecological and Evolutionary Processes (EEPs; chapter three) is indicated by triangle colour (black, no contribution to EEP conservation targets; blue, orange and red contribute to conservation of 1, 2 and 3 EEP targets, respectively). The Usambara Mountains, with high richness per unit area, show relatively high costs of conservation. On the other hand, the Udzungwa and Rubeho Mountains have many areas of above high richness and low cost. For the targets used in these analyses, the North Pare bloc appears to represent the worst choice, showing high costs and generally low richness.

Under analyses that consider only the number of species per unit area, mountain blocs in the northeast are prioritised (Figure 6.3). The data presented in Figure 6.4, however, suggest that cost efficiency should be a top consideration for conservation planning in the EAM. Mountain blocs in the southwest are more remote and appear to offer greater efficiency for conservation (Figure 6.4). Nevertheless, the data presented in this figure do not account for endemism nor gamma diversity; species richness of a particular area gives little indication of its irreplaceability. If a planning unit contains a species found nowhere else in the EAM, then, even if it has low species richness or high cost, it will need to be included in the planned reserve system if the system is to meet all its targets for species' representation.

To explore patterns of biological uniqueness as well as richness and cost I ran Marxan under a series of cost thresholds. These cost thresholds can be used in Marxan to identify the maximum representation of species (or of some other type of conservation feature) for a given budget. Plotting the mean percentage of targets met and area conserved against the total cost of the solution shows that the efficiency gains of using cost rather than area to derive spatial priorities for conservation are most pronounced when the budget is limited (Figure 6.5). Once all targets are met, the spatial priorities for conservation look very similar because many targets have few spatial options (due to endemism and small species' range sizes). Such areas are, therefore, forced into the solution and will often help meet targets for less restricted range species. In solutions that do not meet all targets, species that are less range-restricted can be conserved in cheaper planning units, resulting in the efficiency increases seen. When mapped, the results suggest that under (more realistic) scenarios of scarce conservation resources, species conservation efforts are more efficiently focussed in the southwest of the EAM (Figure 6.6) and larger areas are included (Figure 6.5). In contrast, when area is used to prioritise conservation, the algorithms select much smaller areas in the Uluguru and Usambara Mountains, where median species richness per unit area is higher (Figure 6.3), but so too are costs (Figure 6.4).

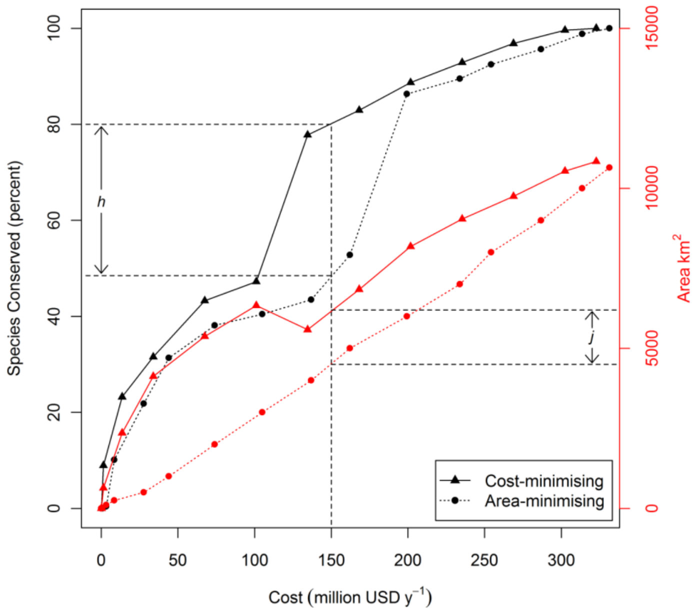


Figure 6.5. Efficiency savings of using cost to set priorities, rather than area. The percentage of targets met (black lines, left hand y-axis) and the area conserved (red lines, right hand y-axis) are plotted against cost (sum of opportunity, management and damage costs) as the available budget is increased. Cost-minimising (triangles, solid lines) and area-minimising (circles, dotted lines) approaches differ in their efficiency with which targets are met. Due to high rarity and few spatial options for many species, the two approaches have similar performance at high levels of target compliance. For 150 million USD per year (dashed black vertical line), *h* and *j* show the increase in species and area conserved within a hypothetical protected area network when a cost-minimising approach is used (80% of species' targets met and 6,200 km^2 conserved), rather than an area-minimising approach (49% of species' targets met and 4,500 km^2 conserved).

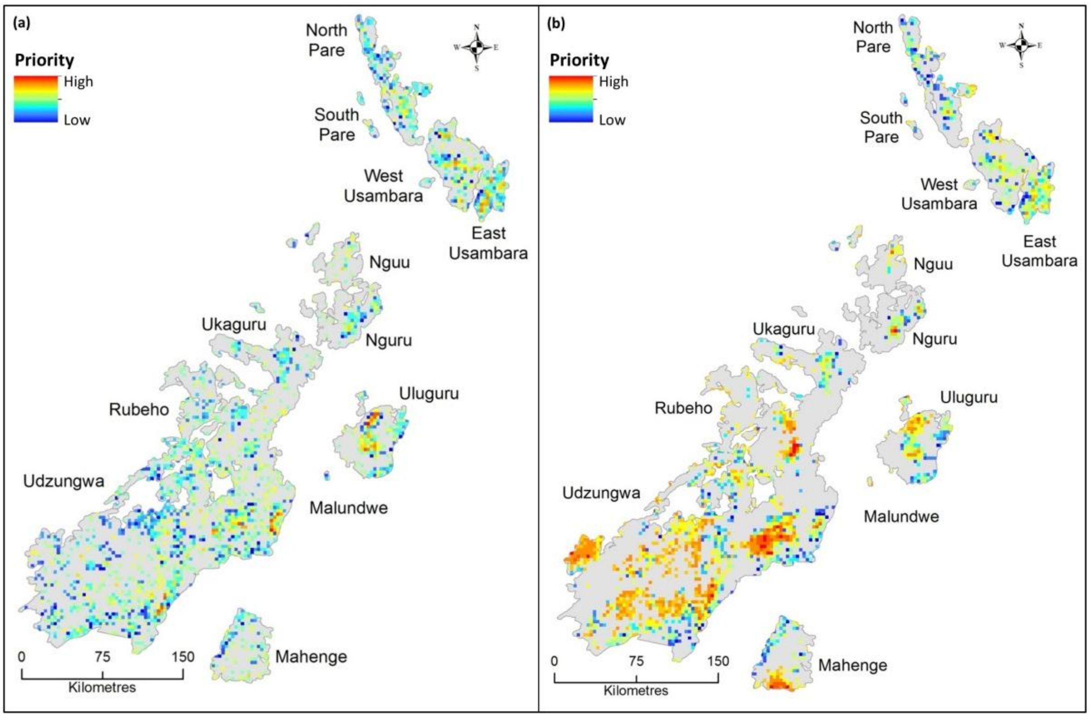


Figure 6.6. A series of budget thresholds were used (as in Figure 6.5) to derive maximum species coverage for a certain area (a) or cost (b). In red are the areas included at the lowest budget thresholds, progressing through to blue areas which are only included when budgets are unlimited. When mapped it is clear that when budgets are limited a cost-minimising approach (as opposed to an area-minimising approach) identifies high priority sites in the southwest and that larger areas are incorporated (see also Figure 6.5).

6.3.2 Damage cost and the boundary length modifier

Damage costs are relatively small compared to other costs (under the current protected area network, estimates of the combined opportunity cost of conservation and protected area management costs are approximately 140 times the estimated cost of wildlife damage; see chapter five). This leads to three important questions:

1. Do damage costs affect the solutions identified by Marxan?
2. If not, then should damage costs be included in an SCP analysis or is it a waste of resources to collect and analyse such data?
3. If they are worth considering, how should they be incorporated?

To answer these questions, a series of SCP analyses were run, in which all inputs and settings were kept constant except for the BLM. Increasing the BLM gives greater weight to damage costs (chapter one; section 6.2). For this series of analyses in which only the BLM setting was altered, damage costs (i.e. boundary costs unmodified by the BLM) were then plotted against the overall cost of the solution (Figure 6.7). Using these results, the first of the questions above is easily answered; when damage costs are considered in the same currency as opportunity and management costs (i.e. the BLM is equal to one and all costs are measured in USD ha^{-1} y^{-1}), there is only a 2% decrease in the total expected cost of damage when compared to an analysis in which damage costs are not considered at all (i.e. BLM is equal to zero). This is because damage costs are over-ridden by the other costs of conservation used in these analyses. There is some justification for using a BLM>1 because there are other costs which I have not measured (so are otherwise effectively ignored in my analyses) and which are associated and likely to be correlated with the cost of damage to crops. These include the costs of negative perceptions towards wildlife conservation, the costs of increased disease transmission to humans and livestock, the cost of time spent guarding crops (which may be time that is spent away from education for children in farming households – a significant, but unquantifiable social cost) and the cost of risk and uncertainty that can be a barrier to economic development (Mackenzie and Ahabyona 2012). It might, therefore, be decided that damage costs should be weighted so that they have greater influence over the spatial priorities identified for conservation. These analyses suggest that a value of around 100 would generate a significant reduction in damage costs without sacrificing the overall efficiency of the solution (Ardron et al. 2008; Possingham et al. 2009a).

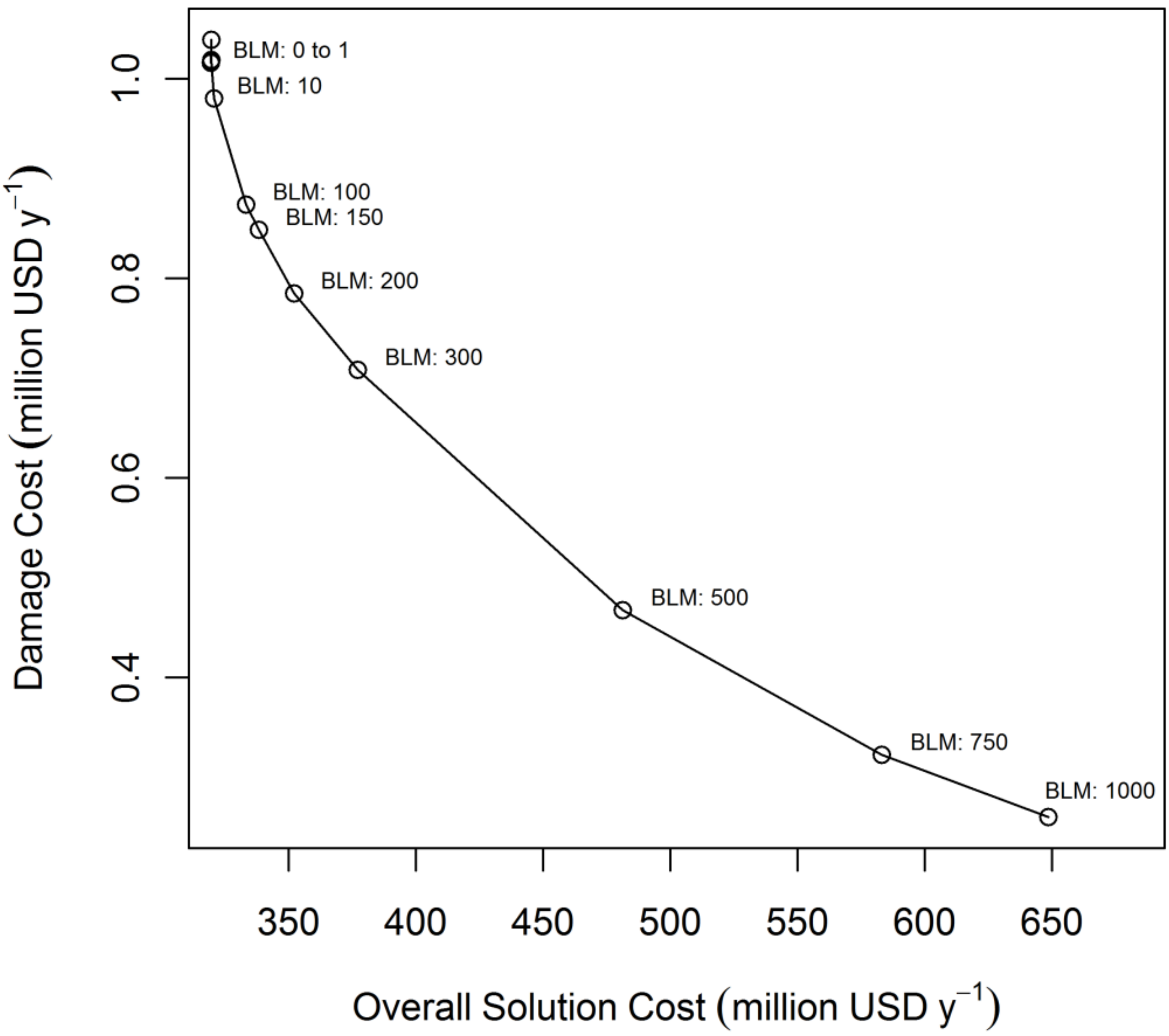


Figure 6.7. Trade-off between minimising damage costs and overall solution efficiency. Varying the Boundary Length Modifier (BLM) shows that inclusion of damage costs results in a solution that is only 2% cheaper than that if damage costs are not considered at all (BLM =1 compared to BLM = 0). However, although using a BLM of 100 (bringing damage costs to a similar order of magnitude as site costs) reduces damage costs by 140,000 USD y^{-1}, it increases the overall cost by 13.6 million USD y^{-1}.

Although damage costs are small in comparison to other costs in the Eastern Arc, they disproportionately affect those living close to protected areas, whilst others, potentially bearing no cost, benefit. Furthermore, communities local to the protected areas are the same people whose cooperation and willingness to conserve might have the biggest effect on management spending. If these costs of wildlife damage are much more substantial than those estimated simply for crop damage (whether due to the costs of negative perceptions of wildlife or some other unquantified cost) then they might best be mitigated by developing conservation plans that are specifically designed to minimise them (e.g. by increasing the BLM; Figure 6.7). Alternatively, and particularly when damage costs are high, it may be more efficient to invest in damage prevention measures such as fences or deterrents (Mackenzie and Ahabyona 2012).

6.3.3 Incorporating Ecological and Evolutionary Processes

Each representation target for a species that is not present across the entire landscape can be viewed as a constraint to the overall spatial flexibility of the potential solution. Incorporating targets for EEP conservation constrains the solution further and their inclusion results in an overall increase in the total cost of the protected area network under both an area-minimising (23% increase) and a cost-minimising (27% increase) approach (compare the bars on left with those on right in Figure 6.8). However, when cost data are included, the best solution (meeting all species and EEP targets) is 16.6 million USD y^{-1} cheaper than that derived when cost data are not available and an area-minimising approach is used as a proxy for cost reduction (compare orange section with grey section in right hand bar of Figure 6.8). Perhaps more importantly, however, a cost-minimising approach directs priorities for areas with low irreplaceability much more clearly than an area-minimising approach (blue areas in Figure 6.9). When area is used, planning units with similar contributions to biodiversity targets are viewed as equal, giving many spatial options that are equal in term of the efficiency with which they contribute to conservation targets. On the other hand, when cost is included planning units that have similar contributions to meeting targets for biodiversity conservation can be ranked by the efficiency with which they do so, resulting in less ambiguous solutions.

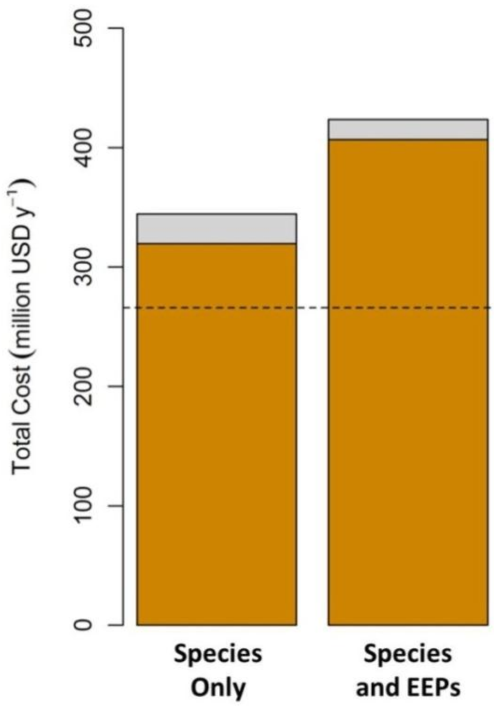


Figure 6.8. The effect of including Ecological and Evolutionary Processes (EEPs). The minimum cost (management, opportunity and damage costs) of meeting conservation targets is lower when EEPs are not considered (left hand bar) than when they are (right hand bar). For both sets of targets, inclusion of cost data (orange section) results in cheaper solutions than those found when area is minimised (grey section). The cost of the current network is indicated (dashed line).

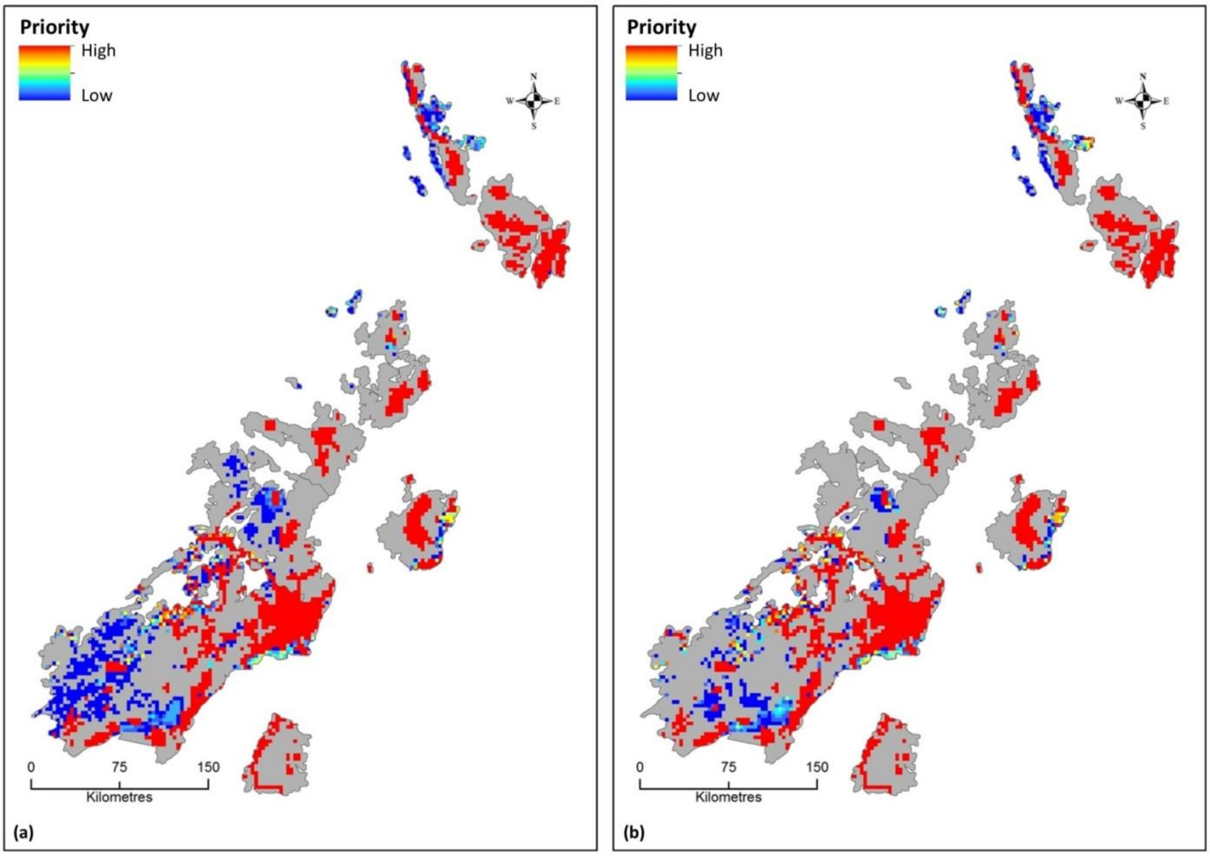


Figure 6.9. Inclusion of cost data helps to prioritise between cells that have similar contributions to biodiversity conservation. When area is minimised (a), cells with similar contributions to conservation targets are viewed as equal, giving many spatial options. On the other hand, when cost is minimised (b), clearer priorities are assigned, as heterogeneity in cost differentiates the efficiency ranking of cells with equal conservation value. These maps do not show the areas that would be included in a "best" solution; they simply show the likelihood of each planning unit being in any particular final solution. Hence, under an area-minimising approach, there are many more planning units with low irreplaceability that might be included in a reserve network, while in the cost-minimising approach priorities between these are guided by their relative cost.

6.3.4 Spatial constraints

Gap analyses, which identify the areas most complementary to the current protected area network, result in the most spatially constrained of solutions; however, they are also the most realistic and most useful, as they focus priorities towards areas that are most complementary to the current system. Although the differences in the mapped solutions are barely discernible, the area-minimising approach (476 million USD y^{-1}; left hand panel in Figure 6.10) that meets representation targets for all conservation features is still three million USD per year more expensive than the cost-minimising approach (473 million USD y^{-1}; right hand panel in Figure 6.10).

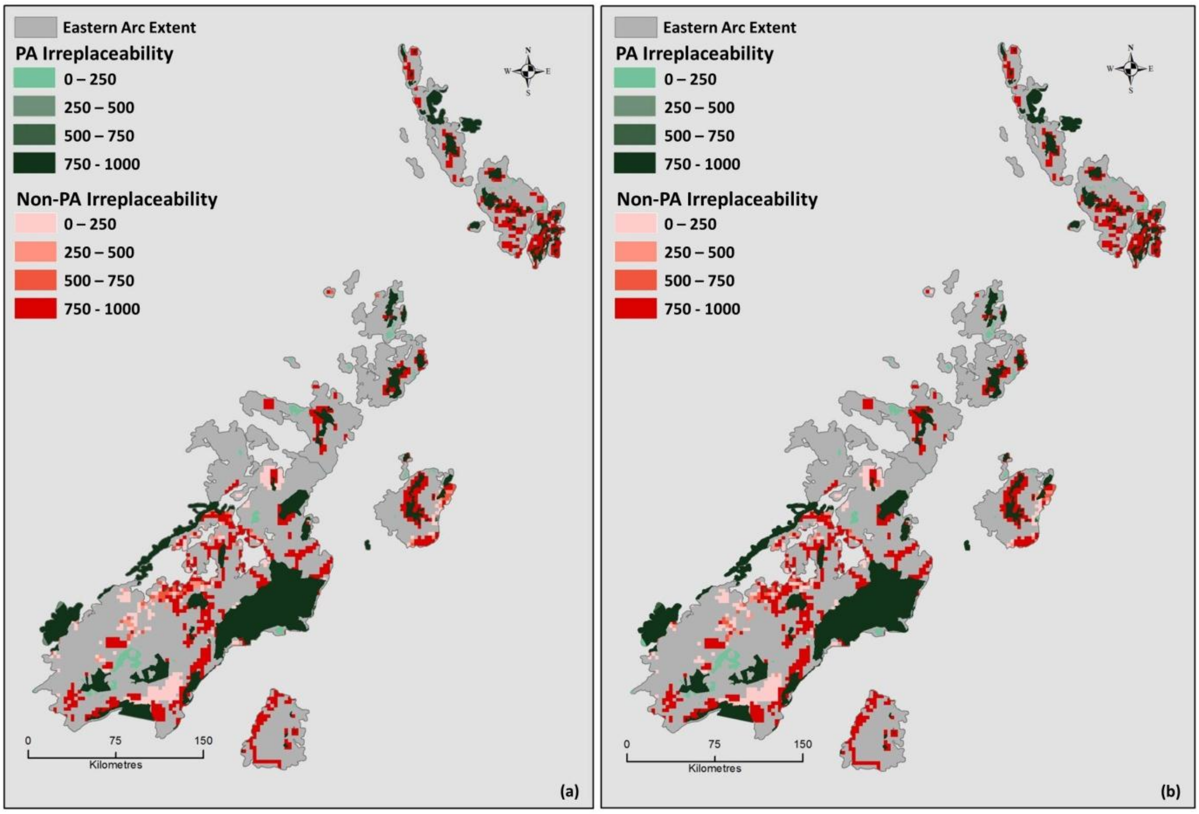


Figure 6.10. Identifying the areas most complementary to the current protected area network. When the current protected area network is used as a starting point (i.e. current protected areas are locked in to the solution) and all targets are met (for both biodiversity pattern and process), there is little difference between the solutions for an area-minimising approach (a) and a cost-minimising approach (b). This is due to the high levels of species' endemism and rarity, the spatial inflexibility of Evolutionary and Ecological Process conservation and the fact that many targets are captured within the current reserve network. These constraints result in few spatial options for meeting targets for many of the conservation features. Despite this, not using cost results in a solution that costs 2.75 million USD more per year).

Spatial constraints affect the ability of SCP software to find efficient solutions. If an area is forced into a hypothetical reserve network, then, for these areas, there are no efficiency savings when cost data are incorporated. In these analyses I consider two specific spatial constraints that limit the options for efficient conservation planning: The first is the inclusion of information on EEPs, as identified in section 6.3.3 and shown in Figure 6.8. EEPs must often be conserved in their entirety if that process is to be conserved (see chapter three). Migration corridors, for instance, must be conserved along their entire length to ensure the continued migration of populations between resources. However, their inclusion in a hypothetical reserve network increases its cost and area. The second and most important spatial constraint are the planning units used. When the current reserve network is not considered and just 9 km^2 planning units are used (left hand bar [orange] in Figure 6.11), the cost of meeting targets is lower than when compared to a reserve network that uses current protected areas as planning units (middle bar [blue] in Figure 6.11). Most expensive is when current reserves are used as planning units and must remain in the solution, whether or not they contribute to conservation targets (right hand bar [red] Figure 6.11). This does not necessarily mean that the current network is inefficient. The targets used in my analyses are a subset of many that conservation planners have taken into account in the past. For that reason, although allowing the software to identify and exclude protected areas that do not contribute to conservation targets is interesting, it would be dangerous to describe the excluded reserves as an inefficiency of the current network. More interesting than the cost increases that result from increased spatial constraints is that spatial constraints also limit the extent to which using cost data can increase the efficiency of a reserve network. For example, the increase in total cost when an area-minimising approach is used instead of a cost-minimising approach is greater when the spatial constraints imposed by the planning units are lower (compare the size of grey sections of each bar in Figure 6.11).

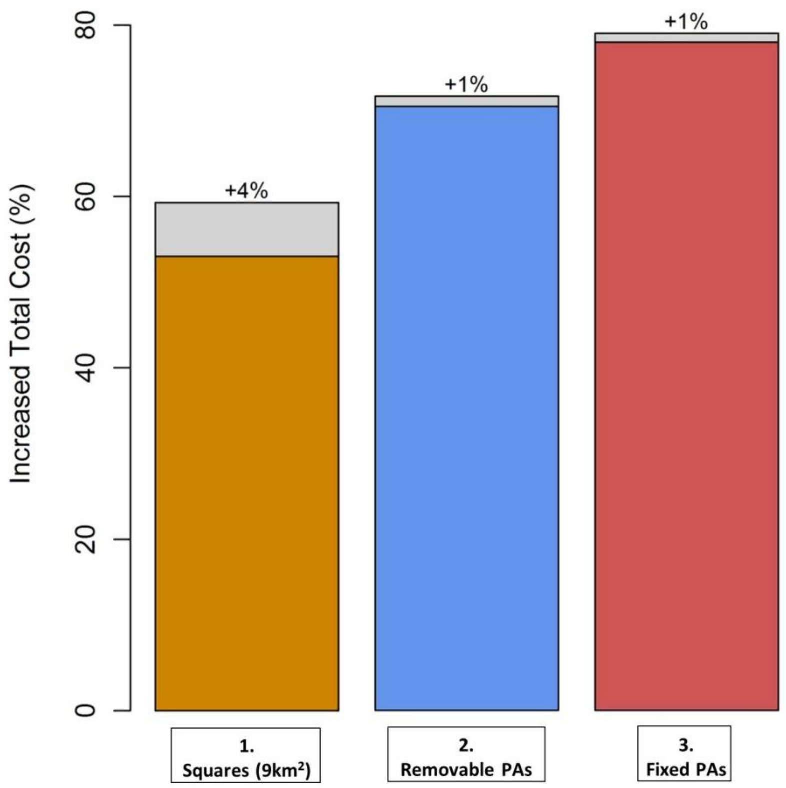


Figure 6.11. Planning units as a spatial constraint. The cost of meeting conservation targets of species and Ecological and Evolutionary Processes (as a percentage increase from the cost of the current reserve network) is plotted for cost-minimising solutions (coloured bars), which consistently identified cheaper reserve networks than area-minimising solutions (grey bars). The size of this effect (shown above each bar as a percentage increase from the cost-minimising approach) depends on the constraints within which the Systematic Conservation Planning algorithms operate. Three planning unit scenarios were used: First, most flexible, was a uniform grid of squares (9 km^2) across the entire Eastern Arc Mountains (orange bar). In this scenario, the current reserve network is not considered. Second, entire protected areas were used as planning units and squares (9 km^2) were used for areas beyond the reserve network (blue bar). In this scenario, protected areas not contributing to conservation targets could be removed from the solution. Third and most constrained was a gap analysis, in which the current reserve network is fixed and the most complementary squares (9 km^2) are identified (red bar).

6.3.5 Financial constraints

The effect of these spatial constraints on the ability of using cost information in SCP to find efficient solutions also depends on the degree to which budgets are limited. Using cost data provides greater efficiency savings when there is a financial constraint that limits the ability of SPC to meet its targets. In this scenario using cost data provides greater efficiency savings than when funds are unlimited and all targets can be met (compare coloured bars of limited financial resources with black crosshatched bars of unlimited resources in Figure 6.12). The difference is most pronounced when spatial constraints, as described previously, are lower. The least constrained scenario (i.e. 9 km^2 planning units and using only species targets) can be used as an example: When a cost-minimising, rather than area-minimising, approach is used to maximise the efficiency with which 80% of conservation targets are met, the area conserved is 33% greater and 15% cheaper. On the other hand, when 100% of targets are met, using cost results in just a 3% increase in area and a 7% decrease in cost (Figure 6.12).

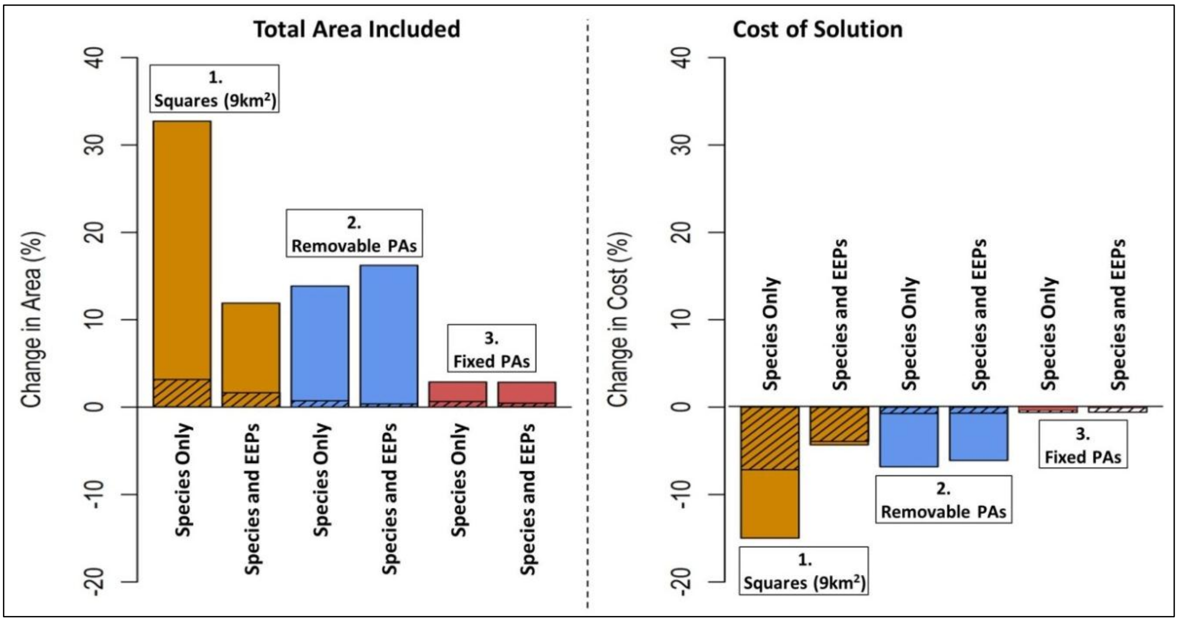


Figure 6.12. The efficiency savings of using cost data in Systematic Conservation Planning under different spatial and financial constraints. For a financially constrained scenario, in which only 80% of targets can be met, the efficiency savings achieved by a cost-minimising approach, rather than an area-minimising approach, are plotted as a percentage change in area (coloured bars in left hand panel) and cost (coloured bars in right hand panel). Greater spatial constraints imposed by the planning unit scenario (coloured bars –increasingly constrained left to right) limit the effect of incorporating cost data and result in smaller increases in area conserved and smaller decreases in the cost of meeting targets. The spatial constraints imposed by the addition of Ecological and Evolutionary Processes (EEPs) also reduces the amount to which incorporating cost data increases efficiency (within each planning unit scenario compare left and right hand bars of the same colour). With the removal of financial constraints, so that 100% of targets can be met (black cross hatched bars overlain), the utility of incorporating cost data to find efficient solutions decreases.

6.3.6 Constraints within the Eastern Arc

The difference between a solution that minimises area and a solution that minimises cost will also depend on the overlap of the conservation features that are being maximised. For instance, if all the targets are met within the three spatial constraints identified above, then there can be no difference between the solutions. If, on the other hand, very few of the targets are met within these constraints, then there is greater potential for efficiency savings, particularly if overlap between these unrepresented targets is low and there are many spatial options for their conservation.

I consider the most spatially constrained scenario to demonstrate how spatial constraints inhibit the utility of using cost data, rather than area, to set priorities (Figure 6.13). Once the current protected areas and spatially fixed EEPs are included in a hypothetical reserve network, 61% of conservation targets are already met (including 97% of the area required for conservation of spatially flexible EEPs – see chapter three). A further 34% of targets have restricted ranges for which there are no alternative options for their conservation and, consequently, no opportunities to use cost data to optimise their efficient representation in a reserve network. This means that, under the most spatially-constrained set of analyses, alternative options exist for the conservation of just 5% of the targets used. Furthermore, of those targets remaining almost 75% are met within the areas that need to be conserved for restricted range targets, so that the difference between an area-minimising and a cost-minimising approach is based entirely on just 1.3% of the targets (red box).

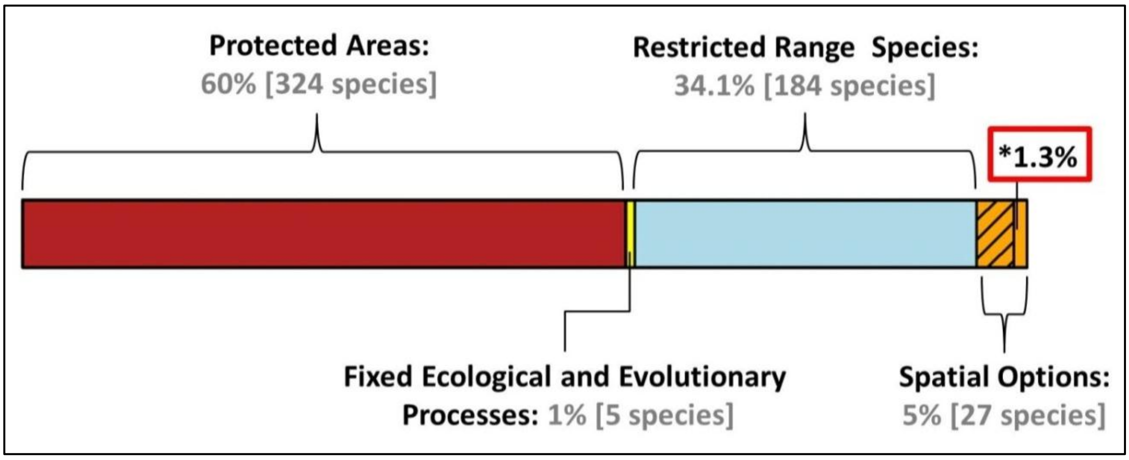


Figure 6.13. Constraints to finding efficient solutions in the Eastern Arc Mountains. Current protected areas (red) and fixed Ecological and Evolutionary Processes (yellow) are spatial constraints to Systematic Conservation Planning which together meet 61% of species' conservation targets. A further 34% of species have such limited ranges there are no alternative spatial options for their conservation (blue). This leaves only 5% of conservation targets which are not met within the spatial constraints and which have alternative options for their conservation (orange). Yet, of these species, 75% are represented when targets for restricted range features are met (cross hatched area). This means, therefore, that the difference between an area-minimising and a cost-minimising solution is based on just 1.3% of species (red box). The percentage of targets met within each category is shown with the number of species [in square brackets].

6.3.7 Considering priorities alongside vulnerability

Using cost data to derive conservation priorities has been advocated as a way to increase the efficiency with which resources are spent (Balmford et al. 2000; Polasky et al. 2001; Balmford et al. 2003; Moore et al. 2004; Naidoo et al. 2006; Polasky 2008). However, alongside questions of efficiency, cost data can also be used to estimate the degree to which natural habitats are threatened (Cowling et al. 1999; Margules and Pressey 2000; Pressey and Taffs 2001; Noss et al. 2002; Lawler et al. 2003; Possingham et al. 2009b). The opportunity costs described here are the foregone profits from agriculture and charcoal production – the two major drivers of deforestation in the Eastern Arc (Fisher 2010). Few studies have available to them such direct measures of the threat under which efficient spatial priorities exist and such an ideal metric with which the threat can be plotted against biological importance (Possingham et al. 2009b). These data allow me to investigate how the temporal scheduling of efficient conservation action might be achieved. Irreplaceability was calculated, using Marxan, for all species and process targets using 9 km^2 planning units. High and low irreplaceability (greater than and less than 50%) were mapped with high and low threat (greater than or less than the median opportunity cost) to highlight both where efficient priorities for conservation are found and the degree of threat that they face (Figure 6.14). The Usambara, Pare and Uluguru Mountains are highly irreplaceable, but are also more threatened due to the high population density around them. On the other hand, many areas of the Udzungwa Mountains are of similar conservation value (i.e. irreplaceability), but are less threatened (Figure 6.14). When the current protected area network is included in the analysis and the results plotted, most blocs show that they tend to have protected areas where opportunity costs are high (Figure 6.15). This might indicate that the reserves were formed in response to known threats. The Udzungwa and Rubeho Mountains, however, appear to show lower than average threat for most of their protected areas, while much of the more threatened habitat within these blocs has no formal protection (Figure 6.15).

An analysis based purely on efficiency might drive priorities to the southwest of the study region, whilst analyses that consider vulnerability highlight the areas of high irreplaceability in the north that are under threat. If all conservation targets are to be met, then it is these areas that face the highest degree of threat now and should receive the most immediate attention. Alternatively, if not all targets can be met, then protecting species at low cost and under less imminent threat may be the wisest use of conservation resources. Such areas might represent bargains for conservation that may not be available, or will cost much more, in the future (Balmford et al. 2003). Therefore, rather than providing a blueprint for conservation action, the set of analyses produced here should inform a decision making process, which should also incorporate other information: How much funding is available and can all conservation targets be met? Are all species valued equally? Where, and on what, can these funds be spent? What form should the conservation intervention take? What is the likelihood of success for a given conservation intervention? Does willingness to engage in conservation vary across the region? Are there any issues of governance or tenure that must be settled?

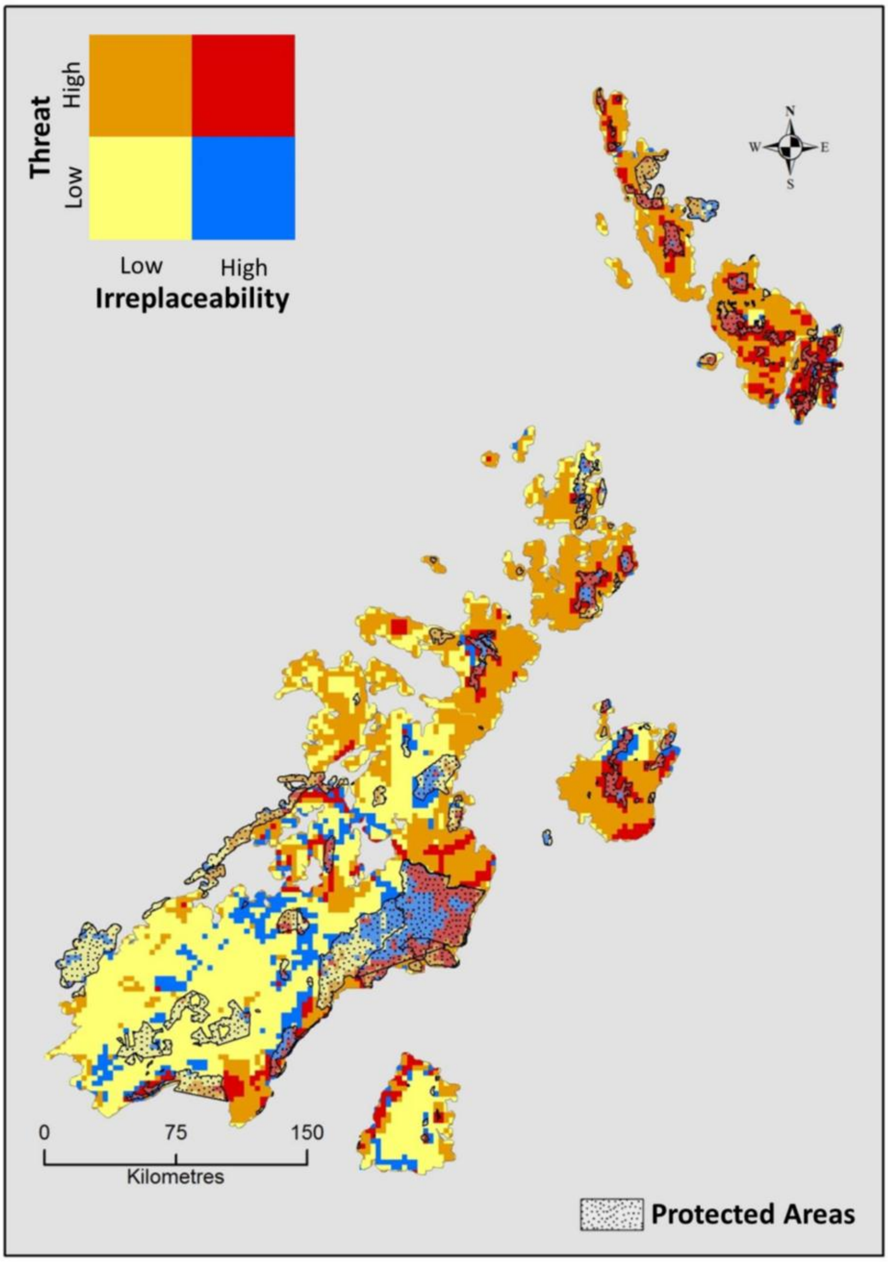


Figure 6.14. Irreplaceability and vulnerability. Threat (assessed as the likely net benefits from agriculture and charcoal production in the absence of protected areas) is mapped alongside irreplaceability (that accounts for all costs and all process and biodiversity targets). Low and high irreplaceability were determined by a 50% cut off (1000 runs), while the median opportunity cost (260 USD ha^{-1} y^{-1}) was used to divide high and low threat.

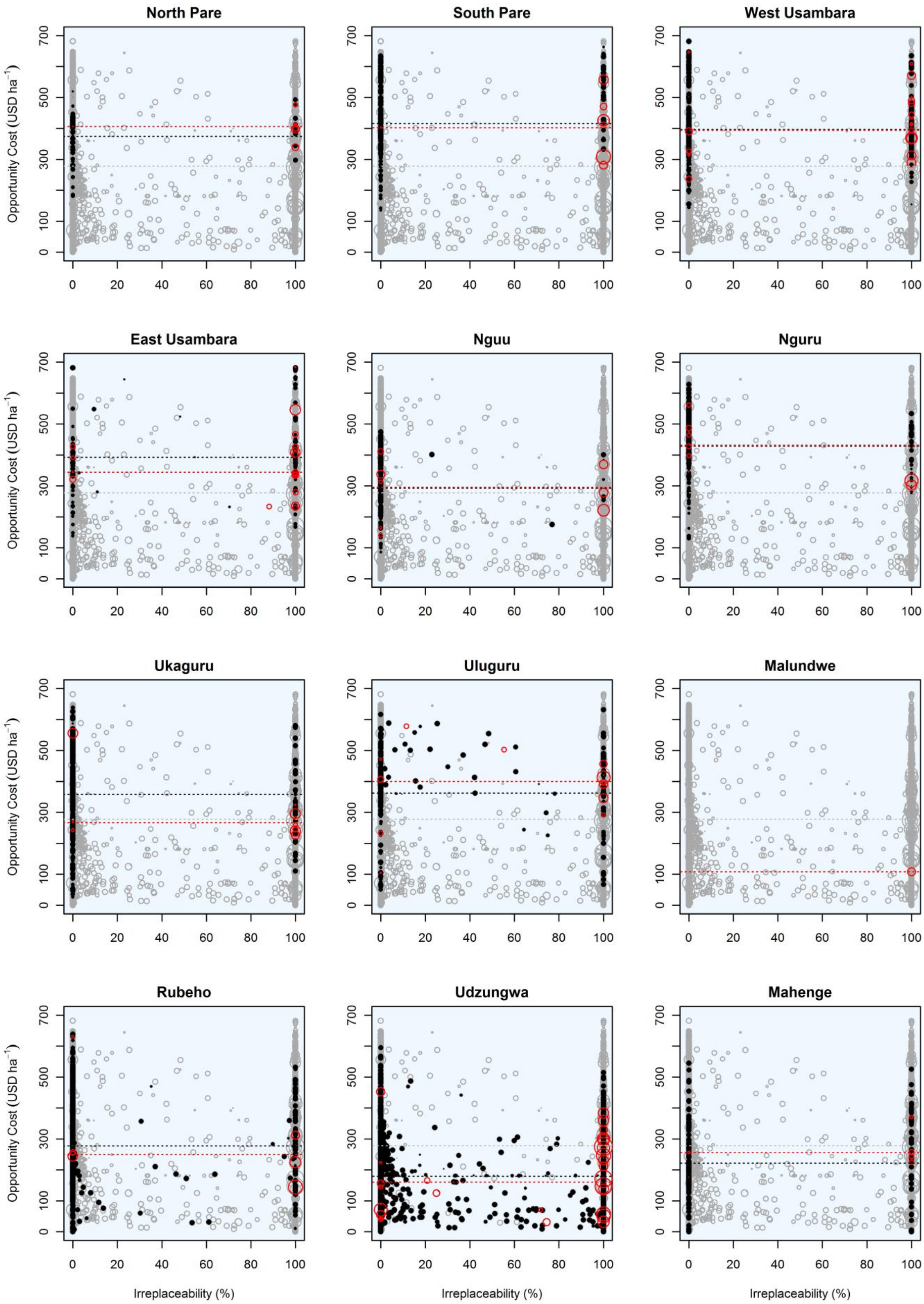


Figure 6.15. Opportunity cost (a measure of threat) is plotted against irreplaceability (out of 1000 runs, all biodiversity and EEP targets and costs were included) for all planning units. For each bloc, protected areas (red) and non-protected areas (black) are plotted to show the relative status of the current protected area network. The plot highlights gaps in the current protected area network and potential threats. Median threat is plotted with dotted lines for entire study area (grey) and for protected (red) and non-protected (black) areas in the bloc. Symbol size is proportional to the area of the planning unit.

There are some limitations to the data and analyses presented here. To begin with, the cost data are based on assumptions of an average farmer and an average protected area management regime (see chapters four and five). Another important limitation is that, although the effect of total reserve area on the modelled cost of effective protected area management is small (chapter four; Green et al. 2012), it is probable that operational efficiency in larger reserve networks and in larger individual reserves will be higher and the costs of mitigating external threats should decrease. The SCP software cannot account for this, so these analyses probably overestimate total management costs. Lastly, as populations increase and farming methods improve and if, as predicted, undernutrition in the region increases (Liu et al. 2008), then both opportunity costs and damage costs could be significantly greater than estimated here. It is largely for these reasons, therefore, that SCP analyses should be reviewed regularly and, as new data become available, they should be incorporated into new analyses. Despite these caveats, it is the relative spatial variation of costs and benefits that directs the spatial and temporal prioritisation of conservation, even if the absolute values are wrong or subject to change.

6.4. Summary

Even when solutions are constrained by having few spatial options in which conservation targets can be met, cost-minimising approaches can find larger and, usually, cheaper solutions. Nevertheless, the utility of using cost data to increase the efficiency of solutions is affected by both spatial and financial constraints. There are three scenarios in which inclusion of cost information is particularly useful. The first is for directing priorities when resources are scarce; when funding is limited and not all conservation targets can be met, inclusion of data on cost gives greatest efficiency savings. The second is when current protected areas do not exist or are poor at meeting targets identified for conservation action. Third, is when there are spatial options for meeting conservation targets: when there is low irreplaceability (i.e. there are several spatial options for meeting a particular conservation target), using data on cost is a more effective, efficient and fair way to direct spatial priorities.

In the Eastern Arc Mountains, inclusion of information on the cost of damage to crops has little effect on the solution because it is vastly outweighed by both management and opportunity costs. Therefore, their inclusion is probably not justified unless there are other costs that are associated and correlated with damage cost, and unless these associated costs can be estimated.

Analyses of pure cost-efficiency suggest that resources should focus on the more remote southwest, such as the Udzungwa or Rubeho Mountains where conservation value is often high, yet costs are lower. However, an analysis that considers threats finds that there are many irreplaceable components of a comprehensive reserve network that are situated in some of the more imperilled areas of the EAM. The implication is that, unless triage (Ochoa-Ochoa et al. 2011) is recommended (and it is beyond the scope of these analyses to do that), these areas should receive initial investments, whilst irreplaceable areas that are of lower value to agriculture and charcoal production (the two major drivers of deforestation in the EAM) could be purchased at a later date. These two perspectives, of cost-efficiency and of temporal priorities under impending threat, suggest quite different solutions and highlight the fact that no single analysis is able to provide a plan for conservation priorities.

These analyses are a comprehensive assessment of the biodiversity benefits, conservation cost and threat of habitat conversion at a scale that is useful to conservation practitioners. More generally, I have demonstrated the circumstances under which the use of cost data to derive priorities will give the greatest efficiency savings. This kind of information is useful for those considering the utility of investing resources to gather and incorporate such data into a conservation plan elsewhere in the world and under various spatial and financial constraints.

7. General discussion

"In the end, we will conserve only what we love"

Baba Dioum, speech to the General Assembly of the IUCN, New Delhi (1969)

7.1. Introduction

Over the last six chapters, I have set out the importance of including information on Ecological and Evolutionary Processes (EEPs) and conservation costs into Systematic Conservation Planning (SCP) and I have described my work to create these data and include them into an SCP framework for the Eastern Arc Mountains (EAM) of Tanzania. I now want to revisit the most important points and consider some emerging themes. For each of these points, I consider its relevance to conservation in the EAM and then to SCP more generally. I finish with some thoughts to the future.

7.2. Important findings

7.2.1 Ecological and Evolutionary Processes

I showed how some EEPs (in particular large mammal migration corridors) might occur in areas of low biodiversity and how these areas need to be included right at the start of an SCP workflow. Of particular concern in the EAM are forested altitudinal gradients. These gradients are proxies for five of the EEPs discussed here and probably others for which there are no data. The processes that such gradients represent operate on scales ranging from days to millions of years; they include both ecological and evolutionary processes and are also expected to provide resilience to human induced climate change. Not only do forested altitudinal gradients represent many EEPs, but they are also the least well represented of the EEP proxies included in my analyses. Those that remain should be a priority of conservation efforts, but conservationists should also consider whether there are viable restoration options to link low and high altitude forest.

I also identified a more general problem regarding planning for EEP conservation: many SCP analyses do not specifically consider EEPs. Instead EEPs are often ignored by conservation planners or they assume (or hope) that EEPs will be incorporated through developing conservation plans that meet targets for species representation (Pressey et al. 2007; Carvalho et al. 2011; Arponen 2012). This is probably because SCP is based upon efficiency and accountability, which are both easier to demonstrate when data are quantitative. However, information on EEPs is rarely quantitative and, being tricky to incorporate into the traditional SCP framework of developing near-optimal networks that represent biodiversity benefits (expressed quantitatively), it is often excluded from consideration.

7.2.2 Conservation costs

In chapter five, I presented a schematic of the division between direct and indirect costs and where these two types of cost are expected to be borne (Figure 5.1). I can now revisit this schematic and begin to put some numbers to it (Figure 7.1). There are three important points that arose from my work to quantify conservation costs. The first is that, as predicted, indirect costs are much greater than direct costs (Balmford and Whitten 2003; Figure 7.1). These indirect costs tend to accumulate locally while many of the benefits of conservation are distributed globally. This underscores the importance of conservation planners taking indirect costs into consideration, whether within an SCP framework or not. Typically only the costs of protected area management are considered, as such data are generally more readily available.

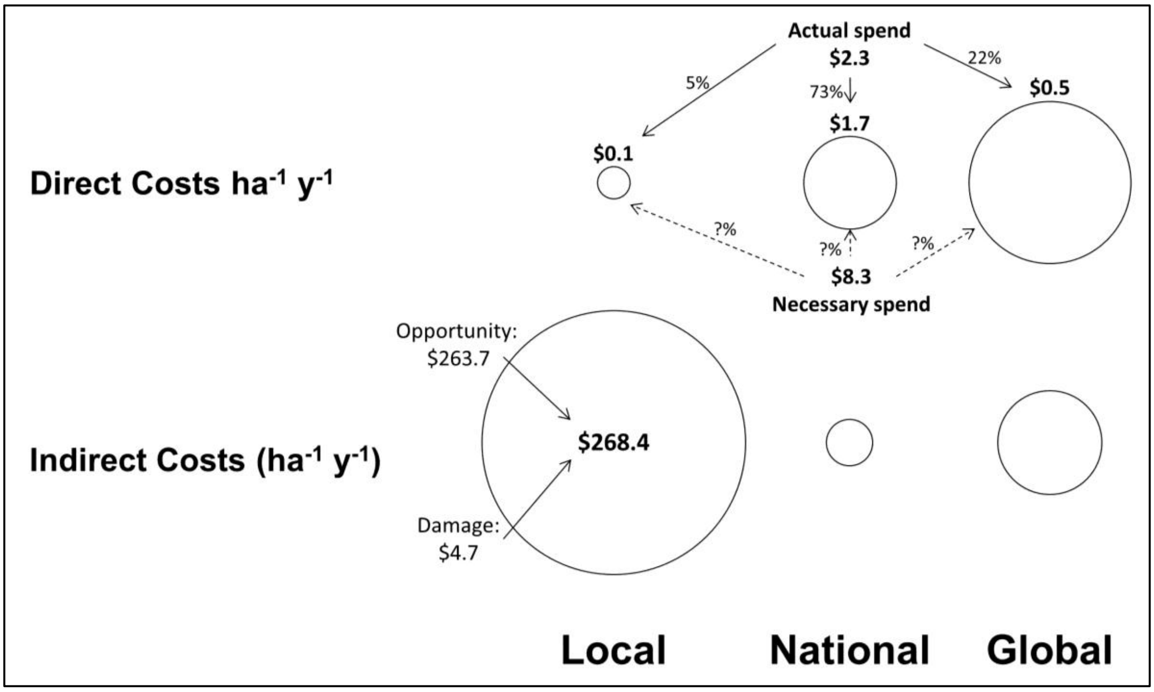


Figure 7.1. My estimates of median costs of conservation for protected areas in the Eastern Arc Mountains (EAM). The relative size of costs, as predicted by Balmford and Whitten (2003) is shown by the circles. The values calculated for the EAM (per ha per year) are also shown. The necessary protected area management spend (8.3 USD ha^{-1} y^{-1}) is over three times the amount actually received (2.3 USD ha^{-1} y^{-1}); however, both are dwarfed by the indirect costs (made up of opportunity costs and damage costs), which sum to almost 270 USD ha^{-1} y^{-1}. Opportunity and damage costs are expected to accrue locally, while the observed protected area management costs are approximately: 5% by the local authority; 73% by the Central Government (largely in the form of staff salaries); and 22% by NGOs and external donors, which are largely funded internationally. Figure adapted from Balmford and Whitten (2003).

Second, there is a chronic shortage of funds for protected area managers in the EAM (Burgess and Kilahama 2004). During my surveys, it became clear that many protected area managers are unable to carry out the operations that they deem necessary to effectively manage the reserves under their control. A noted problem was that salaries were paid by the Central Government, whilst operational budgets were set by local authorities. This may explain why managers in several areas complained that although staffing levels were too low, more frustrating was the fact that those staff who were available could not be properly utilised, as they lacked necessary equipment or transport (Burgess and Kilahama 2004). Crude estimates of the distribution of management costs suggest that the almost three quarters (73%) of protected area management costs are currently borne at the national level, compared to 22% internationally and 5% locally (see discussion, chapter four). This is at odds with the predictions of Balmford and Whitten (2003) who expected a larger proportion of the management costs to be borne internationally (circles in Figure 7.1). Under a scenario of adequate funding, the distribution of costs is unclear, but would probably be different.

Third, population pressure was a useful predictor of cost and outperformed other measures of anthropogenic impact (chapters four and five). It is based on the simple notion that populations will impact the areas around them, but that this impact will decrease as the distance from the population increases. It is particularly useful because it offers a more continuous measure of human use than population density. Population pressure was a useful predictor of both management cost (actual and necessary) and opportunity cost of forest. Moreover, the sigma value (determining the shape of the curve and, consequently, the weight accorded to remote populations) which best fitted the data was 25 (Figure 7.2). This is interesting as it gives an indication of the scale over which human populations influence threats and thus impact the cost of conservation interventions. A sigma value of 25 suggests that the pressure of human populations in the EAM falls steeply as distance increases so that within 20 km, the impact of a population is decreased by around one half. At 40 km it has reduced to one tenth and by 60 km, the population has no impact. By indicating the distances over which human populations exert an influence on conservation interventions and *vice versa*, these results can help planners interested in the costs and benefits of conservation better identify who are included in the "local" communities described in Figure 7.1.

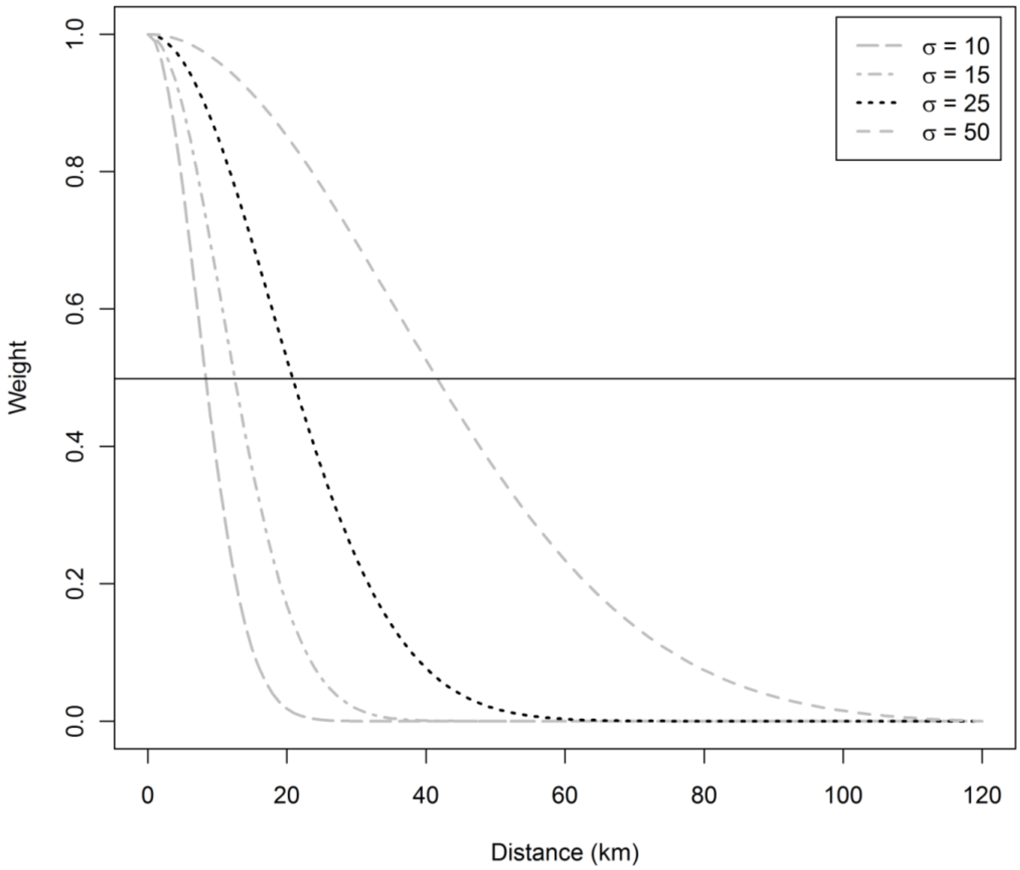


Figure 7.2. Calculating population pressure. Population pressure is hypothesised to impact a particular point in space according to some distance-weighted function. Nearby populations are expected to exhibit highest pressure, which decreases rapidly once the distance to the PA is beyond walking distance. Modifying the sigma value changes the shape of the curve. Higher sigma values give greater weight to relatively distant populations, while smaller sigma 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: distance decay is plotted in black for a sigma value of 25 (found to be the best predictor of protected area management costs and opportunity cost of forest), the impact decreases by 50% at just under 25 km. Distance decay functions for other sigma values are plotted in grey.

7.3. Implications for Systematic Conservation Planning elsewhere

The degree to which including information on costs can improve the efficiency with which conservation targets are met depends very much upon the constraints within which SCP is conducted (Figures 6.11, 6.13). I identified two important constraints – spatial and financial - for which my findings have general relevance to SCP. Spatial constraints include features such as migration corridors (which must be protected in their entirety) and current protected areas (assuming that they will remain in a solution whether they are efficient or not). As these constraints increase, more of the conservation targets are met within them, up to the point at which all targets are met within current spatial constraints. In such a situation, there could be no difference between an area-minimising approach and a cost-minimising approach. Although such a scenario is unlikely, it is possible, as in the EAM, that a large proportion of conservation targets are met within the current protected area network. If this is the case, then researchers should carefully consider the utility of investing time and resources into developing spatially explicit models of cost that may not substantially influence the spatial priorities of an SCP exercise.

Financial constraints have the opposite effect (Figure 6.12). As conservation budgets decrease and SCP analyses are unable to meet all conservation targets, so the utility of using cost data to identify efficient reserve networks increased. However, this effect depends on spatial constraints and on the spatial overlap between conservation targets (Figure 6.13).

Ultimately, cost data are most usefully incorporated into an SCP exercise when the number of conservation targets met within current spatial constraints is low and when resources are limited to achieving less than 100% of those targets. But even when there is no spatial flexibility over where to conserve, it is still useful to quantify costs. It can help identify particularly threatened sites and hence provide useful information for the temporal scheduling of priorities. Moreover, it can help conservation planners consider compensation, payments for ecosystem services or other such schemes to redress the imbalance between those who bear the costs of conservation and those who benefit. Overall the benefits of conservation to society, at local, national or global scales, might outweigh the costs, but asking for such a large proportion of the costs to be borne at the local level is not just unethical, it is also unlikely to succeed in generating effective conservation of biodiversity.

7.4. Imperatives for the future

Future work on conservation planning for the EAM should focus on four points in particular: updates to the land cover data, mapping and inclusion of EEPs in SCP analyses; updates to these analyses to account for future increases in cost and threat; and inclusion of ecosystem services in priority setting. These are justified and described in more detail here.

7.4.1 Improvements to land cover data

The quality of spatially explicit information for use in GIS analyses is rapidly improving for the study area. However, one of the most fundamental GIS layers is the land cover map. It is from this layer that my estimates of opportunity cost, damage cost and land cover change are derived. Furthermore, other analyses pertinent to conservation of biodiversity and ecosystem services also depend on the accuracy of the land cover map (Swetnam et al. 2011; Willcock et al. Submitted). Despite its fundamental role, the land cover map for the EAM has many inaccuracies due to the difficulties of remotely classifying land use classes and because land use changes over time (Sedano et al. 2005; Swetnam et al. 2011). Therefore, the land cover map for the study area needs to be updated and regularly reviewed to enable accurate estimations of the spatial distribution of conservation costs and benefits, to investigate patterns of recent land use change and to predict future changes. A particularly useful inclusion would be for future land cover datasets to include a measure of natural habitat degradation.

7.4.2 Mapping important Ecological and Evolutionary Processes

Inclusion of EEPs into SCP is difficult, mostly because it is an approach that has traditionally, at least in practice, relied heavily on quantitative data. However, this is not going to be enough to ensure species persist in the long-term. Experienced conservationists are often aware of the processes that need to be protected (e.g. see World Bank 2006; Jones et al. 2009a for work on corridors between protected areas) but this has yet to become mainstream within SCP analyses. Collection of information on EEPs in the EAM – in particular focusing on the proxies by which they can be mapped – should be prioritised. For instance, identifying intact forested altitudinal gradients could be much improved if future land cover datasets were to include information on forest quality. This should be undertaken alongside considerations of whether restoration might be a useful tool and, if so, where it can most usefully be applied. Unless these kinds of spatial data can be improved, the ability of SCP to generate long-term solutions will be limited.

7.4.3 Future increases in cost and threat

There are several reasons that conservation costs might increase into the future. Yields are likely to increases through increased use of fertilisers and improved seeds (Denning et al. 2009; Sánchez 2010) raising the value of agricultural areas. This will increase the opportunity cost of conservation and probably increase the cost of managing protected areas in places of high agricultural potential. The cost of damage by wild animals will also be higher. Damage costs might be further exacerbated if increased fragmentation of forest and woodland causes the loss of large carnivores, resulting in meso-predator release of pest species, such as baboons (Prugh et al. 2009; Brashares et al. 2010; Estes et al. 2011). Lastly, climate change predictions for Tanzania suggest the EAM will become increasingly important for agriculture, relative to lowland areas (Jones and Thornton 2009; Thornton et al. 2009; Thornton et al. 2010), particularly as demand increases with projected population increases (Thornton et al. 2010). This will increase the value of crops grown in the EAM, likely increasing all three of the cost types considered in my analyses. More importantly for spatial prioritisation is the potential for changes in the distribution of cost. This is likely if there is significant population migration. For instance, a rural to urban migration as the economy shifts away from its traditional agricultural base could lower the pressure on protected areas in more remote rural areas.

7.4.4 Ecosystem services

The work presented here contrasts biodiversity benefits against conservation costs; however, it would also be useful to quantify some of the direct benefits of conservation to people (Reyers et al. 2012). The natural vegetation of the EAM provides benefits that can be directly linked to human health and wellbeing such as water flow regulation, carbon storage and sustainable flows of timber and Non-Timber Forest Products (NTFPs; Burgess et al. 2009; Fisher et al. 2010; Swetnam et al. 2011). Further analyses which consider costs in the light of such benefits (and the spatial distribution of both) should put the relatively high costs identified here into context. It will be particularly useful to consider how the direct ecosystem service benefits of conservation are distributed in comparison to the costs. Just as I have found costs to correlate with human population pressure, it is likely that benefits will co-vary too. Fine-scale analyses that identify where costs are low and benefits are high are planned as part of the Valuing the Arc project (Burgess et al. 2009).

7.5. Conclusion

The analyses presented in the preceding chapters are not intended as a blueprint for conservation in the EAM. Rather, they are a decision aid for conservation planners based on data that incorporate a diversity of new biological information but also include socio-economic information that has neither been calculated nor mapped before. Nevertheless, costs and threats are dynamic and are particularly subject to change. Therefore, if these analyses are to be used to aid conservation planners they should be updated regularly.

My more general findings – that indirect costs are substantially greater than direct costs, that spatial and financial constraints are useful considerations when planning to gather cost data, and that population pressure is a promising tool to for estimating the effect of conservation interventions on local communities (and *vice versa*) - are of importance to those considering implementing SCP in their region, particularly in developing countries.

ABPmer (2011). ABPMarine Environmental Research Limited, London: Boundary Tool. [www.abpmer.net].
Adams, V. M., Pressey, R. L. and Naidoo, R. (2010). Opportunity costs: Who really pays for conservation?, *Biological Conservation,* 143(2), 439-448.
Ahrends, A., Burgess, N. D., Milledge, S. A. H., Bulling, M. T., Fisher, B., Smart, J. C. R., Clarke, G. P., Mhoro, B. E. and Lewis, S. L. (2010). Predictable waves of sequential forest degradation and biodiversity loss spreading from an African city, *Proceedings of the National Academy of Science,* 107(33), 14556-14561.
Ahrends, A., Burgess, N. D., Gereau, R. E., Marchant, R., Bulling, M. T., Lovett, J. C., Platts, P. J., Kindemba, V. W., Owen, N., Fanning, E. and Rahbek, C. (2011). Funding begets biodiversity, *Diversity and Distributions,* 17(2), 191-200.
Ando, A., Camm, J., Polasky, S. and Solow, A. (1998). Species Distributions, Land Values, and Efficient Conservation, *Science,* 279(5359), 2126-2128.
Ardron, J. A., Possingham, H. P. and Klein, C. J. (2008). *Marxan Good Practices Handbook*, Vancouver, BC: Pacific Marine Analysis and Research Association.
Arponen, A. (2012). Prioritizing species for conservation planning, *Biodiversity and Conservation,* 21(4), 875-893.
Ball, I. and Possingham, H. (2000). *Marxan (v1.8.2): Marine Reserve Design using Spatially Explicit Annealing*, Townsville: The Great Barrier Reef Marine Park Authority.
Ball, I. R., Possingham, H. P. and Watts, M. E. (2009). Marxan and Relatives: Software for Spatial Conservation Prioritization in Moilanen, A., Wilson, K. A. and Possingham, H. P., eds., *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*, Oxford: Oxford University Press, 185-195.
Balmford, A. and Whitten, T. (2003). Who Should Pay for Tropical Conservation, and How Could the Costs Be Met?, *Oryx,* 37(2), 238-250.
Balmford, A., Mace, G. M. and Ginsberg, J. R. (1998). The challenges to conservation in a changing world: putting processes on the map in Mace, G. M., Balmford, A. and Ginsberg, J. R., eds., *Conservation in a changing world*, Cambridge: Cambridge University Press, 1-28.
Balmford, A., Gaston, K. J., Rodrigues, A. S. L. and James, A. (2000). Integrating Costs of Conservation into International Priority Setting, *Conservation Biology,* 14(3), 597-605.
Balmford, A., Moore, J. L., Brooks, T., Burgess, N., Hansen, L. A., Williams, P. and Rahbek, C. (2001). Conservation Conflicts Across Africa, *Science,* 291(5513), 2616-2619.
Balmford, A., Gaston, K. J., Simon, B., James, A. and Kapos, V. (2003). Global Variation in Terrestrial Conservation Costs, Conservation Benefits, and Unmet Conservation Needs, *Proceedings of the National Academy of Sciences of the United States of America,* 100(3), 1046-1050.
Balmford, A., Gravestock, P., Hockley, N., McClean, C. J. and Roberts, C. M. (2004). The worldwide costs of marine protected areas, *Proceedings of the National Academy of Sciences of the United States of America,* 101(26), 9694-9697.
Ban, N. C. and Klein, C. J. (2009). Spatial socioeconomic data as a cost in systematic marine conservation planning, *Conservation Letters,* 2, 206-215.
Ban, N. C., Hansen, G. J. A., Jones, M. and Vincent, A. C. J. (2009). Systematic marine conservation planning in data-poor regions: Socioeconomic data is essential, *Marine Policy,* 33(5), 794-800.
Bank of Tanzania (2006). *Monthly Economic Review December 2006*.
Bank of Tanzania (2007a). *Monthly Economic Review March 2007*.
Bank of Tanzania (2007b). *Monthly Economic Review December 2007*.
Bank of Tanzania (2008a). *Monthly Economic Review December 2008*.
Bank of Tanzania (2008b). *Monthly Economic Review April 2008*.
Bank of Tanzania (2009). *Monthly Economic Review December 2009*.
Bank of Tanzania (2010a). *Monthly Economic Review December 2010*.
Bank of Tanzania (2010b). *Monthly Economic Review May 2010*.
Bauer, D. M., Swallow, S. K. and Paton, P. W. C. (2010). Cost-effective species conservation in exurban communities: A spatial analysis, *Resource and Energy Economics,* 32(2), 180-202.
BBC (2010). The decade's top ten new species, available: http://news.bbc.co.uk/earth/hi/earth_news/newsid_9269000/9269338.stm [accessed 12/12/10].
Bell, R. H. V. and McShane-Caluzi, E. (1986). Conservation and wildlife management in Africa. The proceedings of a workshop organized by the U.S. Peace Corps at Kasungu National Park, Malawi, October 1984,
Beresford, A. E., Buchanan, G. M., Donald, P. F., Butchart, S. H. M., Fishpool, L. D. C. and Rondinini, C. (2011a). Minding the protection gap: estimates of species' range sizes and holes in the Protected Area network, *Animal Conservation,* 14(2), 114-116.
Beresford, A. E., Buchanan, G. M., Donald, P. F., Butchart, S. H. M., Fishpool, L. D. C. and Rondinini, C. (2011b). Poor overlap between the distribution of Protected Areas and globally threatened birds in Africa, *Animal Conservation,* 14(2), 99-107.
BirdLife International (2008). Threatened birds of the world, available: www.birdlife.org/datazone/info/spcdownload [accessed 7/1/2008].
Blanc, J. J., Barnes, R. F. W., Craig, G. C., Dublin, H. T., Thouless, C. R., Douglas-Hamilton, I. and Hart, J. A. (2007). *African Elephant Status Report 2007: an update from the African Elephant Database*, Gland: SSC African Elephant Specialist Group, IUCN.
Blom, A. (2004). An estimate of the costs of an effective system of protected areas in the Niger Delta - Congo Basin Forest Region, *Biodiversity and Conservation,* 13, 2661-2678.
Blomley, T., Pfliegner, K., Isango, J., Zahabu, E., Ahrends, A. and Burgess, N. (2008). Seeing the wood for the trees: an assessment of the impact of participatory forest management on forest condition in Tanzania, *Oryx,* 42(3), 380-391.

Boitani, L., Sinibaldi, I., Corsi, F., De Biase, A., Carranza, I., Ravagli, M., Reggiani, G., Rondinini, C. and Trapanese, P. (2008). Distribution of medium- to large-sized African mammals based on habitat suitability models, *Biodiversity and Conservation,* 17(3), 605-621.

Boitani, L., Maiorano, L., Baisero, D., Falcucci, A., Visconti, P. and Rondinini, C. (2011). What spatial data do we need to develop global mammal conservation strategies?, *Philosophical Transactions of the Royal Society B: Biological Sciences,* 366(1578), 2623-2632.

Bolt, K., Ruta, G. and Sarraf, M. (2005). *Estimating the Cost of Environmental Degradation: A Training Manual in English, French and Arabic*, Washington DC: World Bank.

Börner, J., Mburu, J., Guthiga, P. and Wambua, S. (2009). Assessing opportunity costs of conservation: Ingredients for protected area management in the Kakamega Forest, Western Kenya, *Forest Policy and Economics,* 11(7), 459-467.

Brashares, J. S., Prugh, L. R., Stoner, C. J. and Epps, C. W. (2010). Ecological and conservation implications of mesopredator release in Terborgh, J. and Estes, J. A., eds., *Trophic Cascades: Predators, Prey, and The Changing Dynamics of Nature*, Washington DC: Island Press, 221-240.

Brechin, S. R., Wilshusen, P. R., Fortwangler, C. L. and West, P. C. (2002). Beyond the square wheel: toward a more comprehensive understanding of biodiversity conservation as social and political process, *Society & Natural Resources,* 15(1), 41-64.

Brooks, T., Balmford, A., Burgess, N., Jon, F., Hansen, L. A., Moore, J., Rahbek, C. and Williams, P. (2001). Toward a Blueprint for Conservation in Africa, *BioScience,* 51(8), 613-624.

Brooks, T., Mittermeier, R., Mittermeier, C., da Fonseca, G., Rylands, A., Konstant, W., Flick, P., Pilgrim, J., Oldfield, S., Magin, G. and Hilton-Taylor, C. (2002). Habitat Loss and Extinction in the Hotspots of Biodiversity, *Conservation Biology,* 16(4), 909-923.

Brooks, T., Hoffmann, M., Burgess, N., Plumptre, A., Williams, S., Gereau, R. E., Mittermeier, R. A., Stuart, S. and Robles-Gil, P. (2005). Eastern Afromontane in Mittermeier, R. A., Robles-Gil, P., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C. G., Lamoreux, J. and da Fonseca, G. A. B., eds., *Hotspots Revisited: Earth's Biologically Richest and Most Endangered Ecoregions*, 2nd ed., Chicago: Chicago University Press, 241-242.

Bruner, A., Gullison, R. E. and Balmford, A. (2004). Financial Costs and Shortfalls of Managing and Expanding Protected-Area Systems in Developing Countries, *BioScience,* 54(12), 1119-1126.

Bryan, B. A., King, D. and Ward, J. R. (2011). Modelling and mapping agricultural opportunity costs to guide landscape planning for natural resource management, *Ecological Indicators,* 11(1), 199-208.

Burgess, N. and Kilahama, F. (2004). Is enough being invested in Tanzania's Eastern Arc Mountians?, *The Arc Journal,* 17, 2-4.

Burgess, N. D. and Mlingwa, C. O. F. (2000). Evidence for altitudinal migration of forest birds between montane Eastern Arc and lowland forests in East Africa, *Ostrich,* 71, 184-190.

Burgess, N. D., Fjeldså, J. and Botterweg, R. (1998). Faunal Importance of the Eastern Arc Mountains of Kenya and Tanzania, *Journal of East African Natural History,* 87(1), 37-58.

Burgess, N. and Rodgers, A. (2004). Protected Area Categories and why they matter for the Eastern Arc and coastal forests in Tanzania, *The Arc Journal,* 17, 16-18.

Burgess, N., Doggart, N. and Lovett, J. C. (2002a). The Uluguru Mountains of Eastern Tanzania: The Effect of Forest Loss on Biodiversity, *Oryx,* 36(02), 140-152.

Burgess, N. D., Rahbek, C., Larsen, F. W., Williams, P. and Balmford, A. (2002b). How much of the vertebrate diversity of sub-Saharan Africa is catered for by recent conservation proposals?, *Biological Conservation,* 107(3), 327-339.

Burgess, N., Hales, J. D. A., Underwood, E. and Dinerstein, E. (2004a). *Terrestrial Ecoregions of Africa and Madagascar: A Conservation Assessment,* Washington DC: Island Press.

Burgess, N., Lovett, J., Rodgers, A., Kilahama, F., Nashanda, E., Davenport, T. and Butynski, T. (2004b). Eastern Arc Mountains and Southern Rift in Mittermeier, R. A., P., R. G., Hoffmann, M., Pilgrim, J. D., Brooks, T. M., Mittermeier, C. G., Lamoreux, J. L. and Fonseca, G. A. B., eds., *Hotspots revisited: earth's biologically richest and most endangered ecoregions*, 2nd ed., Chicago: University of Chicago Press, 244–255.

Burgess, N., Küper, W., Mutke, J., Brown, J., Westaway, S., Turpie, S., Meshack, C., Taplin, J., McClean, C. and Lovett, J. (2005). Major gaps in the distribution of protected areas for threatened and narrow range Afrotropical plants, *Biodiversity and Conservation,* 14(8), 1877-1894.

Burgess, N. D., Hales, J. D. A., Ricketts, T. H. and Dinerstein, E. (2006). Factoring species, non-species values and threats into biodiversity prioritisation across the ecoregions of Africa and its islands, *Biological Conservation,* 127(4), 383-401.

Burgess, N. D., Loucks, C., Stolton, S. and Dudley, N. (2007a). The potential of forest reserves for augmenting the protected area network in Africa., *Oryx,* 41(2), 151-159.

Burgess, N. D., Balmford, A., Cordeiro, N. J., Fjeldsa, J., Kuper, W., Rahbek, C., Sanderson, E. W., Scharlemann, J. P. W., Sommer, J. H. and Williams, P. H. (2007b). Correlations among species distributions, human density and human infrastructure across the high biodiversity tropical mountains of Africa, *Biological Conservation,* 134(2), 164-177.

Burgess, N. D., Butynski, T. M., Cordeiro, N. J., Doggart, N. H., Fjeldså, J., Howell, K. M., Kilahama, F. B., Loader, S. P., Lovett, J. C., Mbilinyi, B., Menegon, M., Moyer, D. C., Nashanda, E., Perkin, A., Rovero, F., Stanley, W. T. and Stuart, S. N. (2007c). The biological importance of the Eastern Arc Mountains of Tanzania and Kenya, *Biological Conservation,* 134(2), 209-231.

Burgess, N. D., Mwakalila, S., Madoffe, S., Ricketts, T., Olwero, N., Swetnam, R., Mbilinyi, B., Marchant, R., Mtalo, F., White, S., Munishi, P., Marshall, A., Malimbwi, R., Jambiya, G., Fisher, B., Kajembe, G., Morse-Jones, S., Kulindwa, K., Green, J. and Balmford, A. (2009). Valuing the Arc - A Programme to Map and Value Ecosystem Services in Tanzania, *Mountain Research Initiative Newsletter*, October 2009, 18-21.

Burgman, M. A., Breininger, D. R., Duncan, B. W. and Ferson, S. (2001). Setting Reliability Bounds on Habitat Suitability Indices, *Ecological Applications,* 11(1), 70-78.

Burnham, K. P. and Anderson, D. R. (2002). *Model selection and multimodel inference: a practical information-theoretic approach,* 2nd ed.*,* New York: Springer Verlag.

Busch, J., Dave, R., Hannah, L. E. E., Cameron, A., Rasolohery, A., Roehrdanz, P. and Schatz, G. (2012). Climate Change and the Cost of Conserving Species in Madagascar, *Conservation Biology,* 26(3), 408-419.

Carlson, A. and Hartman, G. (2001). Tropical forest fragmentation and nest predation – an experimental study in an Eastern Arc montane forest, Tanzania, *Biodiversity and Conservation,* 10(7), 1077-1085.

Caro, T. (1999). Densities of mammals in partially protected areas: the Katavi ecosystem of western Tanzania, *Journal of Applied Ecology,* 36(2), 205-217.

Carroll, C., Dunk, J. R. and Moilanen, A. (2010). Optimizing resiliency of reserve networks to climate change: multispecies conservation planning in the Pacific Northwest, USA, *Global Change Biology,* 16(3), 891-904.

Carvalho, S. B., Brito, J. C., Crespo, E. J. and Possingham, H. P. (2011). Incorporating evolutionary processes into conservation planning using species distribution data: a case study with the western Mediterranean herpetofauna, *Diversity and Distributions,* 17(3), 408-421.

Carwardine, J., Rochester, W., Richardson, K., Williams, K., Pressey, R. and Possingham, H. (2007). Conservation planning with irreplaceability: does the method matter?, *Biodiversity and Conservation,* 16(1), 245-258.

Carwardine, J., Wilson, K. A., Ceballos, G., Ehrlich, P. R., Naidoo, R., Iwamura, T., Hajkowicz, S. A. and Possingham, H. P. (2008). Cost-effective priorities for global mammal conservation, *Proceedings of the National Academy of Science,* 105(32), 11446-11450.

Ceballos, G., Ehrlich, P. R., Soberón, J., Salazar, I. and Fay, J. P. (2005). Global Mammal Conservation: What Must We Manage?, *Science,* 309(5734), 603-607.

Chambers, J. M., Cleveland, W. S., Kleiner, B. and Tukey, P. A. (1983). *Graphical methods for data analysis,* Boston, MA: Duxbury Press.

Chan, K. M. A., Shaw, M. R., Cameron, D. R., Underwood, E. C. and Daily, G. C. (2006). Conservation Planning for Ecosystem Services, *PLoS Biology,* 4(11), e379.

Chetkiewicz, C.-L. B., Cassady St. Clair, C. and Boyce, M. S. (2006). Corridors for Conservation: Integrating Pattern and Process, *Annual Review of Ecology, Evolution, and Systematics,* 37, 317-342.

Chiozza, F. (2006). *The Cost of Conservation in Uganda*, unpublished thesis (M.Sc.), Università degli Studi di Roma "La Sapienza".

Chomitz, K. M., Alger, K., Thomas, T. S., Orlando, H. and Nova, P. V. (2005). Opportunity Costs of Conservation in a Biodiversity Hotspot: The Case of Southern Bahia, *Environment and Development Economics,* 10(03), 293-312.

CIMMYT (n.d.). http://apps.cimmyt.org/agricdb/price/Default.aspx, available: [accessed 19/01/11].

Cincotta, R. P., Wisnewski, J. and Engelman, R. (2000). Human population in the biodiversity hotspots, *Nature,* 404(6781), 990-992.

Conrad, J., Gomes, C. P., van Hoeve, W. J., Sabharwal, A. and Suter, J. F. (2012). Wildlife corridors as a connected subgraph problem, *Journal of Environmental Economics and Management,* 63(1), 1-18.

Cordeiro, N. and Howe, H. (2001). Low recruitment of trees dispersed by animals in African forest fragments, *Conservation Biology,* 15(6), 1733-1741.

Cordeiro, N. J. and Howe, H. F. (2003). Forest fragmentation severs mutualism between seed dispersers and an endemic African tree, *Proceedings of the National Academy of Sciences of the United States of America,* 100(24), 14052-14056.

Couvreur, T. h. L. P. and Luke, W. R. Q. (2010). A new species of *Uvariopsis* (Annonaceae), endemic to the Eastern Arc Mountains of Tanzania, *Blumea - Biodiversity, Evolution and Biogeography of Plants,* 55(1), 68-72.

Cowling, R. (2003). A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa, *Biological Conservation,* 112(1-2), 191-216.

Cowling, R. M. and Pressey, R. L. (2001). Rapid plant diversification: Planning for an evolutionary future, *Proceedings of the National Academy of Sciences of the United States of America,* 98(10), 5452-5457.

Cowling, R., Pressey, R., Lombard, A., Desmet, P. and Ellis, A. (1999). From representation to persistence: requirements for a sustainable system of conservation areas in the species-rich mediterranean-climate desert of southern Africa, *Diversity and Distributions,* 5, 51-71.

Craigie, I. D. (2010). *An Assessment of the Performance of Africa's Protected Areas*, unpublished thesis (Ph.D.), University of Cambridge.

Cronk, Q. C. B. and Fuller, J. L. (1995). *Plant invaders: the threat to natural ecosystems,* London: Chapman & Hall.

Culverwell, J. (1997). *Long term recurrent costs of protected area management in Cameroon*, Yaoundé, Cameroon: WWF Cameroon.

Cunneyworth, P. and Stubblefield, L. (1996). *Bamba Ridge Forest Reserve: A biodiversity survey*, Dar es Salaam: Frontier-Tanzania.

Cushman, S. A., McKelvey, K. S. and Schwartz, M. K. (2009). Use of Empirically Derived Source-Destination Models to Map Regional Conservation Corridors, *Conservation Biology,* 23(2), 368-376.

da Fonseca, G. A. B., Balmford, A., Bibby, C., Boitani, L., Corsi, F., Brooks, T., Gascon, C., Olivieri, S., Mittermeier, R. A., Burgess, N., Dinerstein, E., Olson, D., Hannah, L., Lovett, J., Moyer, D., Rahbek, C., Stuart, S. and Williams, P. (2000). ... following Africa's lead in setting priorities, *Nature,* 405(6785), 393-394.

Davenport, T. R. B., Stanley, W. T., Sargis, E. J., De Luca, D. W., Mpunga, N. E., Machaga, S. J. and Olson, L. E. (2006). A New Genus of African Monkey, *Rungwecebus*: Morphology, Ecology, and Molecular Phylogenetics, *Science,* 312(5778), 1378-1381.

Dawson, W., Burslem, D. F. R. P. and Hulme, P. E. (2009). The suitability of weed risk assessment as a conservation tool to identify invasive plant threats in East African rainforests, *Biological Conservation,* 142(5), 1018-1024.
De Boer, W. F. and Baquete, D. S. (1998). Natural Resource Use, Crop Damage and Attitudes of Rural People in the Vicinity of the Maputo Elephant Reserve, Mozambique, *Environmental Conservation,* 25(03), 208-218.
De Klerk, H. M., Fjeldså, J., Blyth, S. and Burgess, N. D. (2004). Gaps in the protected area network for threatened Afrotropical birds, *Biological Conservation,* 117(5), 529-537.
DeFries, R. S., Rudel, T., Uriarte, M. and Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century, *Nature Geoscience,* 3(3), 178-181.
Denning, G., Kabambe, P., Sanchez, P., Malik, A., Flor, R., Harawa, R., Nkhoma, P., Zamba, C., Banda, C. and Magombo, C. (2009). Input subsidies to improve smallholder maize productivity in Malawi: Toward an African Green Revolution, *PLoS Biology,* 7(1), e1000023.
Dixon, J. A. and Sherman, P. B. (1991). *Economics of protected areas: a new look at benefits and costs,* 2nd ed., London: Earthscan Publications Ltd.
Doggart, N., Perkin, A., Kiure, J., Fjeldså, J., Poynton, J. and Burgess, N. (2006). Changing places: How the results of new field work in the Rubeho Mountains influence conservation priorities in the Eastern Arc Mountains of Tanzania, *African Journal of Ecology,* 44(2), 134-144.
Drummond, S., Wilson, K. A., Meijaard, E., Watts, M., Dennis, R., Christy, L. and Possingham, H. (2010). Influence of a Threatened-Species Focus on Conservation Planning, *Conservation Biology,* 24(2), 441-449.
EAMCEF (2008). *Memorandum of agreement between the Eastern Arc Mountains Conservation Endowment Fund on one part and Nilo Nature Reserve on the other part: For the project: General management and implementation of routine activities in Nilo nature reserve*, Morogoro: EAMCEF.
EAMCEF (2009). *Annual Progress Report, Financial Year 2008/2009*, Morogoro: Eastern Arc Mountains Conservation Endowment Fund.
EAMCEF (2010). *Brief Status of Funded Projects August 2006 - March 2010*, Morogoro: Eastern Arc Mountains Conservation Endowment Fund.
Eberhart, S. A. (1969). *Spring Review of the New Cereal Varieties: Global Crop Paper: Corn*, Washington DC: Agency for International Development.
Egoh, B. N., Reyers, B., Carwardine, J., Bode, M., O'Farrell, P. J., Wilson, K. A., Possingham, H. P., Rouget, M., Lange, W. d., Richardson, D. M. and Cowling, R. M. (2010). Safeguarding Biodiversity and Ecosystem Services in the Little Karoo, South Africa, *Conservation Biology,* 24(4), 1021-1030.
Elhorst, J. P. (1993). The estimation of investment equations at the farm level, *European Review of Agricultural Economics,* 20(2), 167-182.
Elith, J., Graham, C. H., Anderson, R. P., Dudík, M., Ferrier, S., Guisan, A., Hijmans, R. J., Huettmann, F., Leathwick, J. R., Lehmann, A., Li, J., Lohmann, L. G., Loiselle, B. A., Manion, G., Moritz, C., Nakamura, M., Nakazawa, Y., Overton, J. M. M., Peterson, A. T., Phillips, S. J., Richardson, K., Scachetti-Pereira, R., Schapire, R. E., Soberón, J., Williams, S., Wisz, M. S. and Zimmermann, N. E. (2006). Novel methods improve prediction of species' distributions from occurrence data, *Ecography,* 29(2), 129-151.
Ellis, F. (1993). *Peasant economics: farm households and agrarian development,* Cambridge: Cambridge University Press.
Environmental Systems Research Institute (2009). *ArcGIS,* Redlands: Environmental Systems Research Institute.
Epps, C. W., Mutayoba, B., Wasser, S. K. and Brashares, J. (2008). *Past and present connectivity of wildlife populations in Tanzania, East Africa*, Morogoro: Sokoine University of Agriculture.
Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., Carpenter, S. R., Essington, T. E., Holt, R. D., Jackson, J. B. C., Marquis, R. J., Oksanen, L., Oksanen, T., Paine, R. T., Pikitch, E. K., Ripple, W. J., Sandin, S. A., Scheffer, M., Schoener, T. W., Shurin, J. B., Sinclair, A. R. E., Soulé, M. E., Virtanen, R. and Wardle, D. A. (2011). Trophic Downgrading of Planet Earth, *Science,* 333(6040), 301-306.
Exchange Rates UK (n.d.). http://www.exchangerates.org.uk, available: [accessed 2010-2012].
FAO (2005). *Livestock Sector Brief: United Republic of Tanzania*, Rome: Food and Agriculture Organization of the United Nations.
Farler, J. P. (1879). The Usambara Country in East Africa, *Proceedings of the Royal Geographical Society and Monthly Record of Geography,* 1(2), 81-97.
FBD (2007). *Eastern Arc Mountains Strategy: Coding Forest Reserves as Protected Areas.*, Morogoro: Conservation and Management of the Eastern Arc Mountain Forests, Forestry and Beekeeping Division.
FBD (2008). *Amani Nature Reserve: Workplan July to June 2008-2009*, Dar es Salaam: Forestry and Beekeeping Division, Ministry of Natural Resources and Tourism.
Ferraro, P. J. (2002). The local costs of establishing protected areas in low-income nations: Ranomafana National Park, Madagascar, *Ecological Economics,* 43(2-3), 261-275.
Ferraro, P. J. and Kiss, A. (2002). Direct Payments to Conserve Biodiversity, *Science,* 298(5599), 1718-1719.
Ferrier, S. and Wintle, B. A. (2009). Quantitative Approaches to Spatial Conservation Prioritization: matching the Solution to the Need in Moilanen, A., Wilson, K. A. and Possingham, H. P., eds., *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*, Oxford: Oxford University Press, 1-15.
Finch, J. and Marchant, R. (2011). A palaeoecological investigation into the role of fire and human activity in the development of montane grasslands in East Africa, *Vegetation History and Archaeobotany,* 20, 109-124.
Fisher, B. (2010). African exception to drivers of deforestation, *Nature Geoscience,* 3(6), 375-376.
Fisher, B. and Christopher, T. (2007). Poverty and biodiversity: Measuring the overlap of human poverty and the biodiversity hotspots, *Ecological Economics,* 62(1), 93-101.

Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R. K. and Burgess, N. D. (2010). Common pool resource management and PES: Lessons and constraints for water PES in Tanzania, *Ecological Economics,* 69(6), 1253-1261.
Fisher, B., Edwards, D. P., Giam, X. and Wilcove, D. S. (2011a). The high costs of conserving Southeast Asia's lowland rainforests, *Frontiers in Ecology and the Environment,* 9, 329–334.
Fisher, B., Lewis, S. L., Burgess, N. D., Malimbwi, R. E., Munishi, P. K., Swetnam, R. D., Kerry Turner, R., Willcock, S. and Balmford, A. (2011b). Implementation and opportunity costs of reducing deforestation and forest degradation in Tanzania, *Nature Climate Change,* 1(3), 161-164.
Fjeldså, J. (2003). Patterns of endemism in African birds: how much does taxonomy matter?, *Ostrich,* 74, 30-38.
Fjeldså, J. (2007). How Broad-Scale Studies of Patterns and Processes Can Serve to Guide Conservation Planning in Africa, *Conservation Biology,* 21(3), 659-667.
Fjeldså, J. and Burgess, N. D. (2008). The coincidence of biodiversity patterns and human settlement in Africa, *African Journal of Ecology,* 46(Suppl. 1), 33–42.
Fjeldså, J. and Lovett, J. C. (1997a). Geographical patterns of old and young species in African forest biota: the significance of specific montane areas as evolutionary centres, *Biodiversity and Conservation,* 6(3), 325-346.
Fjeldså, J. and Lovett, J. C. (1997b). Biodiversity and environmental stability, *Biodiversity and Conservation,* 6, 315-323.
Fjeldså, J. and Tushabe, H. (2005). Complementarity of Species Distributions as a Tool for Prioritising Conservation Actions in Africa: Testing the Efficiency of Using Coarse-Scale Distribution Data in Huber, B. A., Sinclair, B. J. and Lampe, K.-H., eds., *African Biodiversity: Molecules, Organisms, Ecosystems*, New York: Springer Science, 1-24.
Fjeldså, J., Ehrlich, D., Lambin, E. and Prins, E. (1997). Are biodiversity 'hotspots' correlated with current ecoclimatic stability? A pilot study using the NEA-AVHRR remote sensing data, *Biodiversity and Conservation,* 6(3), 401-422.
Fjeldså, J., Burgess, N. D., Blyth, S. and de Klerk, H. M. (2004). Where Are the Major Gaps in the Reserve Network for Africa's Mammals?, *Oryx,* 38(01), 17-25.
Fjeldså, J., Kiure, J., Doggart, N., Hansen, L. A. and Perkins, A. (2010). Distribution of highland forest birds across a potential dispersal barrier in the Eastern Arc Mountains of Tanzania, *Steenstrupia,* 32(1), 1-43.
Foxcroft, L. C., Jarošík, V., Pyšek, P., Richardson, D. M. and Rouget, M. (2011). Protected-Area Boundaries as Filters of Plant Invasions, *Conservation Biology,* 25(2), 400–405.
Frazee, S. R., Cowling, R. M., Pressey, R. L., Turpie, J. K. and Lindenberg, N. (2003). Estimating the costs of conserving a biodiversity hotspot: a case-study of the Cape Floristic Region, South Africa, *Biological Conservation,* 112, 275-290.
Fuller, R. A., McDonald-Madden, E., Wilson, K. A., Carwardine, J., Grantham, H. S., Watson, J. E. M., Klein, C. J., Green, D. C. and Possingham, H. P. (2010). Replacing underperforming protected areas achieves better conservation outcomes, *Nature,* 466(7304), 365-367.
Gaston, K. J. (1991). How Large Is a Species' Geographic Range?, *Oikos,* 61(3), 434-438.
Gaston, K. J. (1996). Species-range-size distributions: patterns, mechanisms and implications, *Trends in Ecology & Evolution,* 11(5), 197-201.
Geist, H. J. and Lambin, E. F. (2002). Proximate causes and underlying driving forces of tropical deforestation, *BioScience,* 52(2), 143-150.
Gentry, A. H. (1988). Changes in plant community diversity and floristic composition on environmental and geographical gradients, *Annals of the Missouri Botanical Garden*, 1-34.
Gereau, R. (unpublished data). Plant threat status according to IUCN criteria (work in progress),
Gereau, R. E., Hall, J., Hemp, A. and Platts, P. J. (in prep.). Distribution and endemism of vascular plants in the Eastern Arc Mountains.
Ghiselin, M. T., ed. (1969). *The triumph of the Darwinian method,* Berkeley, CA: University of California Press.
Gillingham, S. and Lee, P. C. (1999). The Impact of Wildlife-Related Benefits on the Conservation Attitudes of Local People Around the Selous Game Reserve, Tanzania, *Environmental Conservation,* 26(03), 218-228.
Gillingham, S. and Lee, P. C. (2003). People and Protected Areas: A Study of Local Perceptions of Wildlife Crop-Damage Conflict in an Area Bordering the Selous Game Reserve, Tanzania, *Oryx,* 37(03), 316-325.
Government of Tanzania (2011). http://www.tanzania.go.tz/livestock.html, available: [accessed 24/08/2011].
Graham, M. (2006). *Coexistence in a land use mosaic? Land use, risk and elephant ecology in Laikipia District, Kenya*, unpublished thesis (Ph.D.), University of Cambridge.
Grantham, H. S., Wilson, K. A., Moilanen, A., Rebelo, T. and Possingham, H. P. (2009). Delaying conservation actions for improved knowledge: how long should we wait?, *Ecology Letters,* 12(4), 293-301.
Green, J. M. H., Burgess, N. D., Green, R. E., Madoffe, S. S., Munishi, P. K. T., Turner, R. K. and Balmford, A. (2012). Estimating management costs of protected areas: A novel approach from the Eastern Arc Mountains, Tanzania, *Biological Conservation,* 150(1), 5-14.
Grueber, C. E., Nakagawa, S., Laws, R. J. and Jamieson, I. G. (2011). Multimodel inference in ecology and evolution: challenges and solutions, *Journal of Evolutionary Biology,* 24(4), 699-711.
Guerrero, A. M., Knight, A. T., Grantham, H. S., Cowling, R. M. and Wilson, K. A. (2010). Predicting willingness-to-sell and its utility for assessing conservation opportunity for expanding protected area networks, *Conservation Letters,* 3(5), 332-339.
Gunn, J., Hawkins, D., Mofulu, F., Critton, C., Augustine, J. and Norton, G. (2005). Human-wildlife conflict around Mikumi national park, Tanzania, *Proceedings of the Fifth TAWIRI Scientific Conference,* 17-29.
Hamilton, A. C. and Bensted-Smith, R. (1989). *Forest conservation in the East Usambara Mountains, Tanzania,* Gland: IUCN.
Haule, K. S., Johnsen, F. H. and Maganga, S. L. S. (2002). Striving for sustainable wildlife management: the case of Kilombero Game Controlled Area, Tanzania, *Journal of Environmental Management,* 66(1), 31-42.

Henschel, P., Hunter, L., Coad, L., Abernethy, K. and Mühlenberg, M. (2011). Leopard prey choice in the Congo Basin rainforest suggests exploitative competition with human bushmeat hunters, *Journal of Zoology,* 285(1), 11-20.
Hess, S., van Beukering, P., Kayharara, G. and Geofrey, V. (2008). *Livelihoods in the Uluguru mountains of Tanzania: survey report and choice model*, Cambridge: Valuing the Arc Project, University of Cambridge.
Hijmans, R. J., Cameron, S. E., Parra, J. L., Jones, P. G. and Jarvis, A. (2005). Very high resolution interpolated climate surfaces for global land areas, *International Journal of Climatology,* 25(15), 1965-1978.
Hill, C. (2000). Conflict of Interest Between People and Baboons: Crop Raiding in Uganda, *International Journal of Primatology,* 21(2), 299-315.
Holding, C., Njuguna, P. and Gatundu, C. (2001). *Farm Sourced Timber: the Restructuring of the Timber Industry in Kenya-Opportunities and Challenges*, Vienna: International Union of Forestry Research Organizations.
Hopper, S. D. (2009). OCBIL theory: towards an integrated understanding of the evolution, ecology and conservation of biodiversity on old, climatically buffered, infertile landscapes, *Plant and Soil,* 322(1-2), 49-86.
Howard, P. C. (1995). *The Economics of Protected Areas in Uganda: Costs, Benefits and Policy Issues*, unpublished thesis (M.Sc.), University of Edinburgh.
Hymas, O. (2001). *Assessment of the Remaining Forests on the Uluguru Mountains and the Pressures that they Face*, Morogoro: CARE Tanzania and Uluguru Mountains Biodiversity Conservation Project.
Index Mundi (2010). http://www.indexmundi.com/tanzania/inflation_rate_%28consumer_prices%29.html,
IUCN (2001). IUCN Red List Categories and Criteria: Version 3.1, available: http://www.iucnredlist.org/technical-documents/categories-and-criteria/2001-categories-criteria [accessed 17/05/2008].
IUCN (2010a). *World Database on Protected Areas (WDPA),* Cambridge: UNEP-WCMC.
IUCN (2010b). *IUCN Red List of Threatened Species,* Cambridge: IUCN Red List Unit.
Jack, B. K., Kousky, C. and Sims, K. R. E. (2008). Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms, *Proceedings of the National Academy of Sciences,* 105(28), 9465-9470.
Jack, B. K., Leimona, B. and Ferraro, P. J. (2009). A Revealed Preference Approach to Estimating Supply Curves for Ecosystem Services: Use of Auctions to Set Payments for Soil Erosion Control in Indonesia, *Conservation Biology,* 23(2), 359-367.
James, A., Gaston, K. J. and Balmford, A. (2001). Can We Afford to Conserve Biodiversity?, *BioScience,* 51(1), 43-52.
James, A. N., Green, M. J. B. and Paine, J. R. (1999). *A global review of protected area budgets and staff*, Cambridge: World Conservation Monitoring Centre.
Janzen, D. H. (1974). The deflowering of Central America. La deforestación de Centroamérica, *Natural History.,* 83, 48-53.
Jarvis, A., Reuter, H., Nelson, A. and Guevara, E. (2008). Hole-filled SRTM for the globe Version 4, [online], available: [accessed 20/03/09].
Jennings, M. D. (2000). Gap analysis: concepts, methods, and recent results, *Landscape Ecology,* 15(1), 5-20.
Jetz, W. and Rahbek, C. (2002). Geographic range size and determinants of avian species richness, *Science,* 297(5586), 1548.
Jetz, W., Rahbek, C. and Colwell, R. K. (2004). The coincidence of rarity and richness and the potential signature of history in centres of endemism, *Ecology Letters,* 7(12), 1180-1191.
Jhala, Y. (1993). Damage to Sorghum crop by blackbuck, *International Journal of Pest Management,* 39(1), 23-27.
Jones, P. G. and Thornton, P. K. (2009). Croppers to livestock keepers: livelihood transitions to 2050 in Africa due to climate change, *Environmental Science & Policy,* 12(4), 427-437.
Jones, T., Rovero, F. and Msirikale, J. (2007). *Vanishing Corridors: A Last Chance to Preserve Ecological Connectivity between the Udzungwa and Selous-Mikumi Ecosystems of Southern Tanzania.*, Arlington, VA: Conservation International.
Jones, T., Caro, T. and Davenport, T. R. B. (2009a). *Wildlife corridors in Tanzania*, Arusha: Tanzania Wildlife Research Institute (TAWIRI).
Jones, T., Epps, C., Mbano, B., Coppolillo, P., Mutayoba, B. and Rovero, F. (2009b). *Maintaining ecological connectivity between the Protected Areas of south-central Tanzania: evidence and challenges*, Cambridge: Anglia Ruskin University.
Joppa, L. N., Roberts, D. L., Myers, N. and Pimm, S. L. (2011). Biodiversity hotspots house most undiscovered plant species, *Proceedings of the National Academy of Sciences,* 108(32), 13171-13176.
Kagoro-Rugunda, G. (2004). Crop raiding around Lake Mburo National Park, Uganda, *African Journal of Ecology,* 42(1), 32-41.
Kati, V., Devillers, P., Dufrêne, M., Legakis, A., Vokou, D. and Lebrun, P. (2004). Hotspots, complementarity or representativeness? designing optimal small-scale reserves for biodiversity conservation, *Biological Conservation,* 120(4), 471-480.
Kelly, R. C., Youngblood-Coleman, D., Azevedo, M. A., Colley, J., Zhu, J., Jennings, R., Holliway, R., Surnson, A. M., Hughes, M., Rosa, C., Roye, S. and Torres, J. (2008). *Tanzania: 2008 Country Review*, Houston, TX: CountryWatch.
Kideghesho, J. and Mtoni, P. (2008). Who compensates for wildlife conservation in Serengeti?, *International Journal of Biodiversity Science, Ecosystems Services & Management,* 4(2), 112-125.
Kier, G. and Barthlott, W. (2001). Measuring and mapping endemism and species richness: a new methodological approach and its application on the flora of Africa, *Biodiversity and Conservation,* 10(9), 1513-1529.
Klein, C., Wilson, K., Watts, M., Stein, J., Berry, S., Carwardine, J., Stafford Smith, M., Mackey, B. and Possingham, H. (2009). Incorporating ecological and evolutionary processes into continental scale conservation planning, *Ecological Applications,* 19(1), 206-217.

Knight, A. T., Cowling, R. M. and Campbell, B. M. (2006a). An Operational Model for Implementing Conservation Action, *Conservation Biology,* 20(2), 408-419.

Knight, A. T., Driver, A., Cowling, R. M., Maze, K., Desmet, P. G., Lombard, A. T., Rouget, M., Botha, M. A., Boshoff, A. F., Castley, J. G., Goodman, P. S., Mackinnon, K., Pierce, S. M., Sims-Castley, R., Stewart, W. I. and von Hase, A. (2006b). Designing Systematic Conservation Assessments that Promote Effective Implementation: Best Practice from South Africa, *Conservation Biology,* 20(3), 739-750.

Knight, A. T., Cowling, R. M., Possingham, H. P. and Wilson, K. A. (2009). From Theory to Practice: Designing and Situating Spatial Prioritization Approaches to Better Implement Conservation Action in Moilanen, A., Wilson, K. A. and Possingham, H. P., eds., *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*, Oxford: Oxford University Press, 249-259.

Knight, A. T., Grantham, H. S., Smith, R. J., McGregor, G. K., Possingham, H. P. and Cowling, R. M. (2011). Land managers' willingness-to-sell defines conservation opportunity for protected area expansion, *Biological Conservation,* 144(11), 2623-2630.

Kniivilä, M. and Saastamoinen, O. (2002). The Opportunity Costs of Forest Conservation in a Local Economy, *Silva Fennica,* 36(4), 853-865.

Lagabrielle, E., Rouget, M., Payet, K., Wistebaar, N., Durieux, L., Baret, S., Lombard, A. and Strasberg, D. (2009). Identifying and mapping biodiversity processes for conservation planning in islands: A case study in Réunion Island (Western Indian Ocean), *Biological Conservation,* 142(7), 1523-1535.

Lahti, D. (2001). The "edge effect on nest predation" hypothesis after twenty years, *Biological Conservation,* 99(3), 365-374.

Lahti, D. C. (2009). Why we have been unable to generalize about bird nest predation, *Animal Conservation,* 12(4), 279-281.

LandScan (2006). *High resolution global population data set.,* Oak Ridge, TN: Oak Ridge National Laboratory.

Langhammer, P. F., Bakarr, M. I., Bennun, L. and Brooks, T. M. (2007). *Identification and gap analysis of key biodiversity areas,* Gland, Switzerland: IUCN.

Larsen, F. W. and Rahbek, C. (2003). Influence of scale on conservation priority setting – a test on African mammals, *Biodiversity and Conservation,* 12(3), 599-614.

Larsen, F. W., Bladt, J., Balmford, A. and Rahbek, C. (2012). Birds as biodiversity surrogates: will supplementing birds with other taxa improve effectiveness?, *Journal of Applied Ecology,* 49(2), 349–356.

Lawler, J. J., White, D. and Master, L. L. (2003). Integrating representation and vulnerability: two approaches for prioritizing areas for conservation, *Ecological Applications,* 13(6), 1762-1772.

Linkie, M., Dinata, Y., Nofrianto, A. and Leader-Williams, N. (2007). Patterns and perceptions of wildlife crop raiding in and around Kerinci Seblat National Park, Sumatra, *Animal Conservation,* 10(1), 127-135.

Linnell, J. D. C., Rondeau, D., Reed, D. H., Williams, R., Altwegg, R., Raxworthy, C. J., Austin, J. D., Hanley, N., Fritz, H., Evans, D. M., Gordon, I. J., Reyers, B., Redpath, S. and Pettorelli, N. (2010). Confronting the costs and conflicts associated with biodiversity, *Animal Conservation,* 13(5), 429-431.

Liu, J., Fritz, S., van Wesenbeeck, C. F. A., Fuchs, M., You, L., Obersteiner, M. and Yang, H. (2008). A spatially explicit assessment of current and future hotspots of hunger in Sub-Saharan Africa in the context of global change, *Global and Planetary Change,* 64, 222-235.

Loader, S. P., Gower, D. J., Ngalason, W. and Menegon, M. (2010). Three new species of *Callulina* (Amphibia: Anura: Brevicipitidae) highlight local endemism and conservation plight of Africa's Eastern Arc forests, *Zoological Journal of the Linnean Society,* 160(3), 496-514.

Loader, S. P., Poynton, J. C., Lawson, L. P., Blackburn, D. C. and Menegon, M. (2011). Herpetofauna of Montane Areas of Tanzania. 3. Amphibian Diversity in the Northwestern Eastern Arc Mountains, with the Description of a New Species of *Arthroleptis* (Anura: Arthroleptidae), *Fieldiana Life and Earth Sciences,* 4, 90-102.

Loarie, S. R., Duffy, P. B., Hamilton, H., Asner, G. P., Field, C. B. and Ackerly, D. D. (2009). The velocity of climate change, *Nature,* 462(7276), 1052-1055.

Lovett, J. (1985). Moist forests of eastern Tanzania, *Swara,* 8(5), 8-9.

Lovett, J. C. (1998). Eastern Tropical African Centre of Endemism: A Candidate for World Heritage Status, *Journal of East African Natural History,* 87(1), 359-366.

Lovett, J. C. (2003). The Forest Act, 2002 (Tanzania), *Journal of African Law,* 47(1), 133-135.

Lovett, J. C. and Moyer, D. C. (1992). A new nature reserve in the Eastern Arc mountains, Tanzania, *Oryx,* 26(2), 115-118.

Loyola, R. D., Oliveira-Santos, L. G. R., Almeida-Neto, M., Nogueira, D. M., Kubota, U., Diniz-Filho, J. A. F. and Lewinsohn, T. M. (2009). Integrating Economic Costs and Biological Traits into Global Conservation Priorities for Carnivores, *PLoS ONE,* 4(8), e6807.

Mackenzie, C. A. and Ahabyona, P. (2012). Elephants in the garden: Financial and social costs of crop raiding, *Ecological Economics,* 75, 72-82.

Madoffe, S. S. and Munishi, P. K. T. (2010). *Forest condition assessment in the Eastern Arc Mountain Forests of Tanzania*, Dar es Salaam: Forestry and Beekeeping Division, Ministry of Natural Resources and Tourism.

Makundi, W. R. and Okiting'ati, A. (1995). Carbon flows and economic evaluation of mitigation options in Tanzania's forest sector, *Biomass and Bioenergy,* 8(5), 381-393.

Marchal, V. and Hill, C. (2009). Primate Crop-raiding: A Study of Local Perceptions in Four Villages in North Sumatra, Indonesia, *Primate Conservation,* 24, 107-116.

Margules, C., Pressey, R. and Williams, P. (2002). Representing biodiversity: Data and procedures for identifying priority areas for conservation, *Journal of Biosciences,* 27(4), 309-326.

Margules, C. R. and Pressey, R. L. (2000). Systematic conservation planning, *Nature,* 405(6783), 243-53.
Mariaux, J. and Tilbury, C. R. (2006). The Pygmy Chameleons of the Eastern Arc Range (Tanzania): Evolutionary Relationships and the Description of Three New Species of *Rhampholeon* (Sauria: Chamaeleonidae), *Herpetological Journal,* 16, 315-331.
Mascarenhas, A. (1971). Agricultural vermin in Tanzania in Ominde, S. and Baker, S., eds., *Studies in East African geography and development*, Berkeley: University of California Press, 259-267.
Mascia, M. B. and Pailler, S. (2011). Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications, *Conservation Letters,* 4(1), 9-20.
Mbilinyi, B. P., Malimbwi, R. E., Shemwetta, D. T. K., Songorwa, A. N., Zahabu, E., Katani, J. Z. and Kashaigili, J. (2006). *Forest Area Baseline for the Eastern Arc Mountains*, Dar es Salaam: Conservation and Management of the Eastern Arc Mountain Forests, Forestry and Beekeeping Division.
McCrea-Strub, A., Zeller, D., Rashid Sumaila, U., Nelson, J., Balmford, A. and Pauly, D. (2010). Understanding the cost of establishing marine protected areas, *Marine Policy,* 35(1), 1-9.
Mduma, S. R., Lobora, A. L., Foley, C. and Jones, T. (2010). *Tanzania Elephant Management Plan: 2010-2015*, Arusha: Tanzania Wildlife Research Institute (TAWIRI).
Menegon, M., Salvidio, S. and Tilbury, C. (2002). A New Dwarf Forest Chameleon from the Udzungwa Mountains of Tanzania, East Africa, (Squamata: Rhampholeon Günther, 1874), *Journal of Herpetology,* 36(1), 51-57.
Menegon, M., Doggart, N. H. and Owen, N. (2008). The Nguru mountains of Tanzania, an outstanding hotspot of herpetofaunal diversity, *Acta Herpetologica,* 3(2), 107-127.
Meuser, E., Harshaw, H. W. and Mooers, A. Ø. (2009). Public Preference for Endemism over Other Conservation-Related Species Attributes, *Conservation Biology,* 23(4), 1041-1046.
Midgley, G. F., Hannah, L., Millar, D., Thuiller, W. and Booth, A. (2003). Developing regional and species-level assessments of climate change impacts on biodiversity in the Cape Floristic Region, *Biological Conservation,* 112(1-2), 87-97.
Mittermeier, R., Gil, P. R., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C. G., Lamoreux, J. and da Fonseca, G. A. B. (2004). *Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions,* 2nd ed., Chicago: Chicago University Press.
MNRT (1997). *National reconnaissance level land use and natural resources mapping project*, Dar es Salaam: Ministry of Natural Resources and Tourism, The United Republic of Tanzania.
MNRT (2010). *Nomination of Properties for Inclusion on the World Heritage List Serial Nomination: Eastern Arc Mountains Forests of Tanzania*, Dar es Salaam: Ministry of Natural Resources and Tourism, United Republic of Tanzania.
Moilanen, A. and Wintle, B. A. (2007). The boundary-quality penalty: a quantitative method for approximating species responses to fragmentation in reserve selection, *Conservation Biology,* 21(2), 355-364.
Moilanen, A. and Ball, I. (2009). Heuristic and Approximate Optimization Methods for Spatial Conservation Prioritization in Moilanen, A., Wilson, K. A. and Possingham, H. P., eds., *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*, Oxford: Oxford University Press, 58-69.
Moilanen, A., Wilson, K. A. and Possingham, H. (2009a). *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools,* Oxford: Oxford University Press.
Moilanen, A., Possingham, H. P. and Polasky, S. (2009b). A mathematical classification of conservation prioritization problems in Moilanen, A., Wilson, K. A. and Possingham, H. P., eds., *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*, Oxford: Oxford University Press, 28-42.
Moore, J., Balmford, A., Allnutt, T. and Burgess, N. (2004). Integrating costs into conservation planning across Africa, *Biological Conservation,* 117(3), 343-350.
Morrison, S. A. and Boyce, W. M. (2009). Conserving Connectivity: Some Lessons from Mountain Lions in Southern California, *Conservation Biology,* 23(2), 275-285.
Mortelliti, A., Santulli Sanzo, G. and Boitani, L. (2009). Species' surrogacy for conservation planning: caveats from comparing the response of three arboreal rodents to habitat loss and fragmentation, *Biodiversity and Conservation,* 18(5), 1131-1145.
Mouillot, D., Culioli, J. M., Pelletier, D. and Tomasini, J. A. (2008). Do we protect biological originality in protected areas? A new index and an application to the Bonifacio Strait Natural Reserve, *Biological Conservation,* 141(6), 1569-1580.
Mumbi, C. T., Marchant, R., Hooghiemstra, H. and Wooller, M. J. (2008). Late Quaternary vegetation reconstruction from the Eastern Arc Mountains, Tanzania, *Quaternary Research,* 69(2), 326-341.
Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B. and Kent, J. (2000). Biodiversity hotspots for conservation priorities, *Nature,* 403(6772), 853-858.
Naidoo, R. and Adamowicz, W. L. (2005). Economic benefits of biodiversity exceed costs of conservation at an African rainforest reserve, *Proceedings of the National Academy of Sciences of the United States of America,* 102(46), 16712.
Naidoo, R. and Adamowicz, W. L. (2006). Modeling Opportunity Costs of Conservation in Transitional Landscapes, *Conservation Biology,* 20(2), 490-500.
Naidoo, R. and Ricketts, T. H. (2006). Mapping the Economic Costs and Benefits of Conservation, *PLoS Biology,* 4(11), e360.
Naidoo, R. and Iwamura, T. (2007). Global-scale mapping of economic benefits from agricultural lands: Implications for conservation priorities, *Biological Conservation,* 140(1-2), 40-49.
Naidoo, R., Balmford, A., Ferraro, P. J., Polasky, S., Ricketts, T. H. and Rouget, M. (2006). Integrating economic costs into conservation planning, *Trends in Ecology & Evolution,* 21(12), 681-687.

Naughton, L., Rose, R. and Treves, A. (1999). *The social dimensions of human-elephant conflict in Africa: A literature review and case studies from Uganda and Cameroon*, Gland: IUCN.

Naughton-Treves, L. (1997). Farming the Forest Edge: Vulnerable Places and People around Kibale National Park, Uganda, *Geographical Review,* 87(1), 27-46.

Naughton-Treves, L. (1998). Predicting Patterns of Crop Damage by Wildlife around Kibale National Park, Uganda, *Conservation Biology,* 12(1), 156-168.

Naughton-Treves, L. and Treves, A. (2005). Socio-ecological factors shaping local support for wildlife: crop raiding by elephants and other wildlife in Africa in Woodroffe, R., Thirgood, S. and Rabinowitz, A., eds., *People and wildlife: conflict or co-existence?* , Cambridge: Cambridge University Press, 252-277.

Naughton-Treves, L., Treves, A., Chapman, C. and Wrangham, R. (1998). Temporal patterns of crop-raiding by primates: linking food availability in croplands and adjacent forest, *Journal of Applied Ecology,* 35(4), 596-606.

NBS (2002). *Tanzania Population and Housing Census 2002*, Dar es Salaam: National Bureau of Statistics, Ministry of Planning, Econnomy and Empowerment.

Nelson, A. (2008). *Estimated travel time to the nearest city of 50,000 or more people in year 2000*, Ispra: Global Environment Monitoring Unit - Joint Research Centre of the European Commission.

Nelson, A. and Chomitz, K. M. (2009). *Protected Area Effectiveness in Reducing Tropical Deforestation. A Global Analysis of the Impact of Protection Status*, Washington D. C: The World Bank.

Nelson, F., Nshala, R. and Rodgers, W. (2007). The evolution and reform of Tanzanian wildlife management, *Conservation and Society,* 5(2), 232.

Neumann, R. P. (1992). Political ecology of wildlife conservation in the Mt. Meru area of Northeast Tanzania, *Land Degradation & Development,* 3(2), 85-98.

Neumann, R. P. (2002). *Imposing wilderness: Struggles over livelihood and nature preservation in Africa,* Berkeley: University of California Press.

Newmark, W. D. (1991). Tropical Forest Fragmentation and the Local Extinction of Understory Birds in the Eastern Usambara Mountains, Tanzania, *Conservation Biology,* 5(1), 67-78.

Newmark, W. D. (2002). *Conserving Biodiversity in East African Forests: A Study of the Eastern Arc Mountains,* Berlin Heidelberg New York: Springer-Verlag.

Newmark, W. D. (2009). *Recommended Land Use Management Guidelines for Corridors in the Eastern Arc Mountains*, Washington, D.C.: Critical Ecosystem Partnership Fund.

Newmark, W. D. and Stanley, T. R. (2011). Habitat fragmentation reduces nest survival in an Afrotropical bird community in a biodiversity hotspot, *Proceedings of the National Academy of Sciences,* 108(28), 11488-11493.

Newmark, W. D., Manyanza, D. N., Gamassa, D.-G. M. and Sariko, H. I. (1994). The Conflict between Wildlife and Local People Living Adjacent to Protected Areas in Tanzania: Human Density as a Predictor, *Conservation Biology,* 8(1), 249-255.

Ngambeki, D. S. (1985). Economic evaluation of alley cropping *Leucaena* with maize-maize and maize-cowpea in Southern Nigeria, *Agricultural Systems,* 17(4), 243-258.

Nhancale, B. A. and Smith, R. J. (2011). The influence of planning unit characteristics on the efficiency and spatial pattern of systematic conservation planning assessments, *Biodiversity and Conservation,* 20(8), 1821-1835.

Nicholson, E., Keith, D. A. and Wilcove, D. S. (2009). Assessing the Threat Status of Ecological Communities, *Conservation Biology,* 23(2), 259-274.

Nicholson, E., Westphal, M. I., Frank, K., Rochester, W. A., Pressey, R. L., Lindenmayer, D. B. and Possingham, H. P. (2006). A new method for conservation planning for the persistence of multiple species, *Ecology Letters,* 9(9), 1049-1060.

Nielsen, M. R. (2006). Importance, cause and effect of bushmeat hunting in the Udzungwa Mountains, Tanzania: Implications for community based wildlife management, *Biological Conservation,* 128(4), 509-516.

Nijman, V. and Nekaris, K. A.-I. (2010). Testing a model for predicting primate crop-raiding using crop- and farm-specific risk values, *Applied Animal Behaviour Science,* 127(3-4), 125-129.

Norton-Griffiths, M. and Southey, C. (1995). The opportunity costs of biodiversity conservation in Kenya, *Ecological Economics,* 12(2), 125-139.

Noss, R. F., Carroll, C., Vance-Borland, K. and Wuerthner, G. (2002). A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem, *Conservation Biology,* 16(4), 895-908.

Ochoa-Ochoa, L. M., Bezaury-Creel, J. E., Vázquez, L.-B. and Flores-Villela, O. (2011). Choosing the survivors? A GIS-based triage support tool for micro-endemics: Application to data for Mexican amphibians, *Biological Conservation,* 144(11), 2710-2718.

Ogra, M. V. (2008). Human-wildlife conflict and gender in protected area borderlands: A case study of costs, perceptions, and vulnerabilities from Uttarakhand (Uttaranchal), India, *Geoforum,* 39(3), 1408-1422.

Orme, C. D. L., Davies, R. G., Olson, V. A., Thomas, G. H., Ding, T.-S., Rasmussen, P. C., Ridgely, R. S., Stattersfield, A. J., Bennett, P. M., Owens, I. P. F., Blackburn, T. M. and Gaston, K. J. (2006). Global Patterns of Geographic Range Size in Birds, *PLoS Biology,* 4(7), e208.

Osborn, F. V. and Hill, C. M. (2005). Techniques to reduce crop loss: human and technical dimensions in Africa in Woodroffe, R., Thirgood, S. and Rabinowitz, A., eds., *People and wildlife: conflict or co-existence?* , Cambridge: Cambridge University Press, 72-85.

Parks, P. J. (1995). Explaining "Irrational" Land Use: Risk Aversion and Marginal Agricultural Land, *Journal of Environmental Economics and Management,* 28(1), 34-47.

Perz, S., Brilhante, S., Brown, F., Caldas, M., Ikeda, S., Mendoza, E., Overdevest, C., Reis, V., Reyes, J. F., Rojas, D., Schmink, M., Souza, C. and Walker, R. (2008). Road building, land use and climate change: prospects for

environmental governance in the Amazon, *Philosophical Transactions of the Royal Society B: Biological Sciences,* 363(1498), 1889–1895.
Peterson, A. T. (2001). Predicting species' geographic distributions based on ecological niche modeling, *The Condor,* 103, 599-605.
Platts, P. (2011). *Population Pressure Grids*, Cambridge: Valuing the Arc project, University of Cambridge.
Platts, P. J. (2012). *Spatial modelling, phytogeography and conservation in the Eastern Arc Mountains of Tanzania and Kenya*, unpublished thesis (Ph.D.), University of York.
Platts, P. J., McClean, C. J., Lovett, J. C. and Marchant, R. (2008). Predicting tree distributions in an East African biodiversity hotspot: model selection, data bias and envelope uncertainty, *Ecological Modelling,* 218(1-2), 121-134.
Platts, P. J., Burgess, N. D., Gereau, R. E., Lovett, J. C., Marshall, A. R., McClean, C. J., Pellikka, P. K. E., Swetnam, R. D. and Marchant, R. (2011). Delimiting tropical mountain ecoregions for conservation, *Environmental Conservation,* 38, 312-324.
Platts, P. J., Ahrends, A., Gereau, R. E., McClean, C. J., Lovett, J. C., Marshall, A. R., Pellikka, P. K. E., Mulligan, M., Fanning, E. and Marchant, R. (2010). Can distribution models help refine inventory-based estimates of conservation priority? A case study in the Eastern Arc forests of Tanzania and Kenya, *Diversity and Distributions,* 16(4), 628-642.
PNB (n.d.). Maize Basic, available: http://www.pnbkrishi.com/maize.htm [accessed 21/02/2011].
Pócs, T. (1998). Bryophyte diversity along the Eastern Arc, *Journal of East African Natural History,* 87(1), 75-84.
Polasky, S. (2008). Why conservation planning needs socioeconomic data, *Proceedings of the National Academy of Sciences of the United States of America,* 105(18), 6505-6506.
Polasky, S., Camm, J. D. and Garber-Yonts, B. (2001). Selecting Biological Reserves Cost-Effectively: An Application to Terrestrial Vertebrate Conservation in Oregon, *Land Economics,* 77(1), 68-78.
Polasky, S., Nelson, E., Lonsdorf, E., Fackler, P. and Starfield, A. (2005). Conserving species in a working landscape: land use with biological and economic objectives, *Ecological Applications,* 15(4), 1387-1401.
Pontius Jr., R. G. and Schneider, L. C. (2001). Land-cover change model validation by an ROC method for the Ipswich watershed, Massachusetts, USA, *Agriculture, Ecosystems & Environment,* 85(1–3), 239-248.
Pontius Jr., R. G., Huffaker, D. and Denman, K. (2004). Useful techniques of validation for spatially explicit land-change models, *Ecological Modelling,* 179(4), 445-461.
Porter, P. and Sheppard, E. (1998). *A world of difference: Society, nature, development,* New York: Guilford Press.
Possingham, H., Ball, I. and Andelman, S. (2000). Mathematical Methods for Identifying Representative Reserve Networks in Ferson, S. and Burgman, M., eds., *Quantitative Methods for Conservation Biology*, New York: Springer-Verlag, 291-306.
Possingham, H., Ward, T., Stewart, R., Segan, D. and Kircher, L. (2009a). *Wild Australia Program - South West Australia Science Project: PEW Log No. 2006-000202*, Washington DC: PEW Environment Group.
Possingham, H. P., Moilanen, A. and Wilson, K. A. (2009b). Accounting for habitat dynamics in conservation planning in Moilanen, A., Wilson, K. A. and Possingham, H. P., eds., *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*, Oxford: Oxford University Press, 135-144.
Pratt, D. J., Greenway, P. J. and Gwynne, M. D. (1966). A Classification of East African Rangeland, with an Appendix on Terminology, *Journal of Applied Ecology,* 3(2), 369-382.
Pressey, R. and Taffs, K. H. (2001). Scheduling conservation action in production landscapes: priority areas in western New South Wales defined by irreplaceability and vulnerability to vegetation loss, *Biological Conservation,* 100(3), 355-376.
Pressey, R. L. (1994). *Ad Hoc* Reservations: Forward or Backward Steps in Developing Representative Reserve Systems?, *Conservation Biology,* 8(3), 662-668.
Pressey, R. L. (2004). Conservation Planning and Biodiversity: Assembling the Best Data for the Job, *Conservation Biology,* 18(6), 1677-1681.
Pressey, R. L., Cowling, R. M. and Rouget, M. (2003). Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa, *Biological Conservation,* 112(1-2), 99-127.
Pressey, R. L., Cabeza, M., Watts, M. E., Cowling, R. M. and Wilson, K. A. (2007). Conservation planning in a changing world, *Trends in Ecology & Evolution (Personal Edition),* 22(11), 583-92.
Pringle, R. M. (2008). Elephants as agents of habitat creation for small vertebrates at the patch scale, *Ecology,* 89(1), 26-33.
Priston, N. (2009). Exclosure plots as a mechanism for quantifying damage to crops by primates, *International Journal of Pest Management,* 55, 243-249.
Priston, N. E. C. and Underdown, S. J. (2009). A simple method for calculating the likelihood of crop damage by primates: an epidemiological approach, *International Journal of Pest Management,* 55(1), 51-56.
Prugh, L. R., Stoner, C. J., Clinton, W. E., Bean, W. T., Ripple, W. J., Laliberte, A. S. and Brashares, J. S. (2009). The Rise of the Mesopredator, *BioScience,* 59(9), 779-791.
Purvis, A., Gittleman, J. L., Cowlishaw, G. and Mace, G. M. (2000). Predicting Extinction Risk in Declining Species, *Proceedings: Biological Sciences,* 267(1456), 1947-1952.
Qian, H. and Ricklefs, R. E. (2000). Large-scale processes and the Asian bias in species diversity of temperate plants, *Nature,* 407(6801), 180-182.
R Development Core Team (2009). *R: A language and environment for statistical computing,* Vienna, Austria: R Foundation for Statistical Computing.
Raxworthy, C. J., Pearson, R. G., Rabibisoa, N., Rakotondrazafy, A. M., Ramanamanjato, J.-B., Raselimanana, A. P., Wu, S., Nussbaum, R. A. and Stone, D. A. (2008). Extinction vulnerability of tropical montane endemism from warming and

upslope displacement: a preliminary appraisal for the highest massif in Madagascar, *Global Change Biology,* 14(8), 1703-1720.
Reardon, T. and Vosti, S. A. (1995). Links between rural poverty and the environment in developing countries: Asset categories and investment poverty, *World Development,* 23(9), 1495-1506.
Reyers, B., Polasky, S., Tallis, H., Mooney, H. A. and Larigauderie, A. (2012). Finding Common Ground for Biodiversity and Ecosystem Services, *BioScience,* 62(5), 503-507.
Ricardo, D. (1821). *On The Principles of Political Economy and Taxation,* Third Edition ed.*,* Kitchener, Ontario: Batoche Books.
Rodrigues, A. S. L. and Brooks, T. M. (2007). Shortcuts for Biodiversity Conservation Planning: The Effectiveness of Surrogates, *Annual Review of Ecology, Evolution, and Systematics,* 38, 713–737.
Rodrigues, A. S. L., Andelman, S. J., Bakarr, M. I., Boitani, L., Brooks, T. M., Cowling, R. M., Fishpool, L. D. C., da Fonseca, G. A. B., Gaston, K. J., Hoffmann, M., Long, J. S., Marquet, P. A., Pilgrim, J. D., Pressey, R. L., Schipper, J., Sechrest, W., Stuart, S. N., Underhill, L. G., Waller, R. W., Watts, M. E. J. and Yan, X. (2004a). Effectiveness of the global protected area network in representing species diversity, *Nature,* 428(6983), 640-643.
Rodrigues, A. S. L., Akçakaya, H. R., Andelman, S. J., Bakarr, M. I., Boitani, L., Brooks, T. M., Chanson, J. S., Fishpool, L. D. C., da Fonseca, G. A. B., Gaston, K. J., Hoffmann, M., Marquet, P. A., Pilgrim, J. D., Pressey, R. L., Schipper, J., Sechrest, W., Stuart, S. N., Underhill, L. G., Waller, R. W., Watts, M. E. J. and Yan, X. (2004b). Global Gap Analysis: Priority Regions for Expanding the Global Protected-Area Network, *BioScience,* 54(12), 1092-1100.
Rondinini, C., Stuart, S. and Boitani, L. (2005). Habitat Suitability Models and the Shortfall in Conservation Planning for African Vertebrates, *Conservation Biology,* 19(5), 1488-1497.
Rondinini, C., Wilson, K. A., Boitani, L., Grantham, H. and Possingham, H. P. (2006). Tradeoffs of different types of species occurrence data for use in systematic conservation planning, *Ecology Letters,* 9(10), 1136-1145.
Rondinini, C., Di Marco, M., Chiozza, F., Santulli, G., Baisero, D., Visconti, P., Hoffmann, M., Schipper, J., Stuart, S. N., Tognelli, M. F., Amori, G., Falcucci, A., Maiorano, L. and Boitani, L. (2011). Global habitat suitability models of terrestrial mammals, *Philosophical Transactions of the Royal Society B: Biological Sciences,* 366(1578), 2633-2641.
Rosenberg, D. K., Noon, B. R. and Meslow, E. C. (1997). Biological corridors: form, function, and efficacy, *BioScience,* 47(10), 677-687.
Rouget, M., Richardson, D. M. and Cowling, R. M. (2003a). The current configuration of protected areas in the Cape Floristic Region, South Africa--reservation bias and representation of biodiversity patterns and processes, *Biological Conservation,* 112(1-2), 129-145.
Rouget, M., Richard, M. C., Robert, L. P. and David, M. R. (2003b). Identifying spatial components of ecological and evolutionary processes for regional conservation planning in the Cape Floristic Region, South Africa, *Diversity & Distributions,* 9(3), 191-210.
Rouget, M., Cowling, R. M., Lombard, A. T., Knight, A. T. and Kerley, G. I. H. (2006). Designing Large-Scale Conservation Corridors for Pattern and Process, *Conservation Biology,* 20(2), 549-561.
Rovero, F., Rathbun, G. B., Perkin, A., Jones, T., Ribble, D. O., Leonard, C., Mwakisoma, R. R. and Doggart, N. (2008). A new species of giant sengi or elephant-shrew (genus Rhynchocyon) highlights the exceptional biodiversity of the Udzungwa Mountains of Tanzania, *Journal of Zoology,* 274(2), 126-133.
Rudel, T. K. (2007). Changing agents of deforestation: From state-initiated to enterprise driven processes, 1970-2000, *Land Use Policy,* 24(1), 35-41.
Rudel, T. K., Defries, R., Asner, G. P. and Laurance, W. F. (2009). Changing Drivers of Deforestation and New Opportunities for Conservation, *Conservation Biology,* 23(6), 1396-1405.
Saj, T. L., Sicotte, P. and Paterson, J. D. (2001). The conflict between vervet monkeys and farmers at the forest edge in Entebbe, Uganda, *African Journal of Ecology,* 39, 195-199.
Sánchez, P. A. (2010). Tripling crop yields in tropical Africa, *Nature Geoscience,* 3(5), 299-300.
Sandel, B., Arge, L., Dalsgaard, B., Davies, R., Gaston, K., Sutherland, W. and Svenning, J. C. (2011). The influence of Late Quaternary climate-change velocity on species endemism, *Science,* 334(6056), 660-664.
Sappington, J., Longshore, K. M. and Thompson, D. B. (2007). Quantifying landscape ruggedness for animal habitat analysis: a case study using bighorn sheep in the Mojave Desert, *The Journal of Wildlife Management,* 71(5), 1419-1426.
Sarkar, S., Pressey, R. L., Faith, D. P., Margules, C. R., Fuller, T., Stoms, D. M., Moffett, A., Wilson, K. A., Williams, K. J., Williams, P. H. and Andelman, S. (2006). Biodiversity Conservation Planning Tools: Present Status and Challenges for the Future, *Annual Review of Environmental Resources,* 31, 123-159.
Scharff, N. (1992). The linyphiid fauna of eastern Africa (Araneae: Linyphiidae)—distribution patterns, diversity and endemism, *Biological Journal of the Linnean Society,* 45(2), 117-154.
Schmitt, C. B., Burgess, N. D., Coad, L., Belokurov, A., Besançon, C., Boisrobert, L., Campbell, A., Fish, L., Gliddon, D., Humphries, K., Kapos, V., Loucks, C., Lysenko, I., Miles, L., Mills, C., Minnemeyer, S., Pistorius, T., Ravilious, C., Steininger, M. and Winkel, G. (2009). Global analysis of the protection status of the world's forests, *Biological Conservation,* 142(10), 2122-2130.
Sedano, F., Gong, P. and Ferrão, M. (2005). Land cover assessment with MODIS imagery in southern African Miombo ecosystems, *Remote Sensing of Environment,* 98(4), 429-441.
Shaw, M. R., Klausmeyer, K., Cameron, D. R., Mackenzie, J. and Roehrdanz, P. (2012). Economic Costs of Achieving Current Conservation Goals in the Future as Climate Changes, *Conservation Biology,* 26(3), 385-396.
Sillero-Zubiri, C. and Switzer, D. (2001). *Crop raiding primates: Searching for alternative, humane ways to resolve conflict with farmers in Africa*, Oxford: Wildlife Conservation Research Unit, Oxford University.

SIPRI (2010). The SIPRI Military Expenditure Database, [online], available: http://milexdata.sipri.org/ [accessed 14/12/10].
Smith, R. J., Minin, E. D., Linke, S., Segan, D. B. and Possingham, H. P. (2010). An approach for ensuring minimum protected area size in systematic conservation planning, *Biological Conservation,* 143(11), 2525-2531.
Sodhi, N. S., Bickford, D., Diesmos, A. C., Lee, T. M., Koh, L. P., Brook, B. W., Sekercioglu, C. H. and Bradshaw, C. J. A. (2008). Measuring the Meltdown: Drivers of Global Amphibian Extinction and Decline, *PLoS ONE,* 3(2), e1636.
Spawls, S., Howell, K., Drewes, R. and Ashe, J. (2004). *A Field Guide to the Reptiles of East Africa,* 3rd ed., London: A & C Black Publishers Ltd.
Stanley, W., Rogers, M. and Hutterer, R. (2005). A new species of Congosorex from the Eastern Arc Mountains, Tanzania, with significant biogeographical implications, *Journal of Zoology,* 265(3), 269-280.
Stattersfield, A., Crosby, M., Long, A. and Wege, D. (1998). *Endemic Bird Areas of the World: Priorities for Biodiversity Conservation,* Cambridge: BirdLife International.
Stewart, R., Noyce, T. and Possingham, H. (2003). Opportunity cost of ad hoc marine reserve design decisions: an example from South Australia, *Marine Ecology Progress Series,* 253, 25-38.
Stuart, S. N. (1983). *Biogeographical and ecological aspects of forest bird communities in Eastern Tanzania*, unpublished thesis (Ph.D.), University of Cambridge.
Swetnam, R. D., Fisher, B., Mbilinyi, B. P., Munishi, P. K. T., Willcock, S., Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N. D. and Marshall, A. R. (2011). Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling, *Journal of Environmental Management,* 92(3), 563-574.
TANAPA (2001). *Udzungwa Mountains National Park: General Management Plan / Environmental Impact Assessment*, Arusha: Department of Planning and Development Projects, Tanzania National Parks.
TANAPA (2009). *Tanzania National Parks Annual Report 2007/08*, Arusha: Tanzania National Parks.
Thirgood, S., Woodroffe, R. and Rabinowitz, A. (2005). The impact of human-wildlife conflict on human lives and livelihoods in Woodroffe, R., Thirgood, S. and Rabinowitz, A., eds., *People and wildlife: conflict or co-existence?* , Cambridge: Cambridge University Press, 13-26.
Thornton, P. K., Jones, P. G., Alagarswamy, G. and Andresen, J. (2009). Spatial variation of crop yield response to climate change in East Africa, *Global Environmental Change,* 19(1), 54-65.
Thornton, P. K., Jones, P. G., Alagarswamy, G., Andresen, J. and Herrero, M. (2010). Adapting to climate change: Agricultural system and household impacts in East Africa, *Agricultural Systems,* 103(2), 73-82.
Turpie, J. K. (2000). *Environmental Management and Biodiversity Conservation of Forests, Woodlands, and Wetlands of the Rufiji Delta and Floodplain: The Use and Value of Natural Resources of the Rufiji Floodplain and Delta, Tanzania*, Dar es Salaam: Rufiji Environment Management Project.
Tweheyo, M., Hill, C. M. and Obua, J. (2005). Patterns of crop raiding by primates around the Budongo Forest Reserve, Uganda, *Wildlife Biology,* 11(3), 237-247.
Underwood, E. C., Shaw, M. R., Wilson, K. A., Kareiva, P., Klausmeyer, K. R., McBride, M. F., Bode, M., Morrison, S. A., Hoekstra, J. M. and Possingham, H. P. (2008). Protecting Biodiversity when Money Matters: Maximizing Return on Investment, *PLoS ONE,* 3(1), e1515.
United Nations (2011). *World Population Prospects: The 2010 Revision. Volume I: Comprehensive Tables*, New York: United Nations.
US Department of State (2008). *Human Rights Report: Tanzania*, Washington DC: United States Department of State.
Vanthomme, H., Bellé, B. and Forget, P. M. (2010). Bushmeat Hunting Alters Recruitment of Large-seeded Plant Species in Central Africa, *Biotropica,* 42(6), 672-679.
Venter, O., Meijaard, E., Possingham, H., Dennis, R., Sheil, D., Wich, S., Hovani, L. and Wilson, K. (2009). Carbon payments as a safeguard for threatened tropical mammals, *Conservation Letters,* 2(3), 123-129.
Vera-Diaz, M. d. C., Kaufmann, R. K., Nepstad, D. C. and Schlesinger, P. (2008). An interdisciplinary model of soybean yield in the Amazon Basin: The climatic, edaphic, and economic determinants, *Ecological Economics,* 65(2), 420-431.
Walsh, P. D., Abernethy, K. A., Bermejo, M., Beyersk, R., Wachter, P. D., Akou, M. E., Huijbregts, B., Mambounga, D. I., Toham, A. K., Kilbournk, A. M., Lahmq, S. A., Latourk, S., Maiselsk, F., Mbinak, C., Mihindouk, Y., Obiang, S. n. N., Effa, E. N., Starkeyk, M. P., Telfer, P., Thibault, M., Tutin, C. E. G., White, L. J. T. and Wilkie, D. S. (2003). Catastrophic ape decline in western equatorial Africa, *Nature,* 422(6932), 611-614.
Walsh, S., Crawford, T., Welsh, W. and Crews-Meyer, K. (2001). A multiscale analysis of LULC and NDVI variation in Nang Rong district, northeast Thailand, *Agriculture, Ecosystems & Environment,* 85(1-3), 47-64.
Warman, L. D., Sinclair, A., Scudder, G., Klinkenberg, B. and Pressey, R. L. (2004). Sensitivity of systematic reserve selection to decisions about scale, biological data, and targets: case study from southern British Columbia, *Conservation Biology,* 18(3), 655-666.
Warren, Y., Buba, B. and Ross, C. (2007). Patterns of crop-raiding by wild and domestic animals near Gashaka Gumti National Park, Nigeria, *International Journal of Pest Management,* 53(3), 207-216.
Watts, M. E., Ball, I. R., Stewart, R. S., Klein, C. J., Wilson, K., Steinback, C., Lourival, R., Kircher, L. and Possingham, H. P. (2009). Marxan with Zones: Software for optimal conservation based land- and sea-use zoning, *Environmental Modelling & Software,* 24(12), 1513-1521.
Whyte, I., Biggs, H., Gaylard, A. and Braack, L. (2008). A new policy for the management of the Kruger National Park's elephant population, *Koedoe-African Protected Area Conservation and Science,* 42(1), 111-132.
Wilcove, D. S. (1985). Nest predation in forest tracts and the decline of migratory songbirds, *Ecology,* 66(4), 1211-1214.
Wilkie, D. S., Carpenter, J. F. and Zhang, Q. (2001). The under-financing of protected areas in the Congo Basin: so many parks and so little willingness-to-pay, *Biodiversity and Conservation,* 10(5), 691-709.

Willcock, S., Phillips, O. L., Platts, P. J., Balmford, A., Burgess, N. D., Lovett, J. C., Ahrends, A., Bayliss, J., Doggart, N. D., Doody, K., Fanning, E., Green, J., Hall, J., Howell, K. L., Marchant, R., Marshall, A. R., Mbilinyi, B., Munishi, P. K. T., Owen, N., Swetnam, R. D., Topp-Jorgensen, E. J. and Lewis, S. L. (Submitted). Towards regional, error-bounded landscape carbon storage estimates for data defficient areas of the world, *PLoS ONE*.

Wilson, E. O. (1994). *The diversity of life,* 3rd ed., London: Penguin Books Ltd.

Wilson, K. A., Cabeza, M. and Klein, C. J. (2009). Fundamental Concepts of Spatial Conservation Prioritization in Moilanen, A., Wilson, K. A. and Possingham, H. P., eds., *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*, Oxford: Oxford University Press, 16-27.

Wise, R. M., Reyers, B., Guo, C., Midgley, G. F. and De Lange, W. (2012). Costs of Expanding the Network of Protected Areas as a Response to Climate Change in the Cape Floristic Region, *Conservation Biology,* 26(3), 397-407.

World Bank (2006). *Resettlement Action Plan for Farm Plots Displaced for Biodiversity Conservation in the Derema Forest Corridor*, Dar es Salaam: Ministry of Natural Resources and Tourism, Forestry and Beekeeping Division.

Yudelman, M., Ratta, A. and Nygaard, D. (1998). *Pest Management and Food production: Looking to the Future*, Discussion Paper 25, Washington DC: International Food Policy Research Institute.

Zorita, E. and Tilya, F. F. (2002). Rainfall variability in Northern Tanzania in the March-May season (long rains) and its links to large-scale climate forcing, *Climate Research,* 20(1), 1-7.

Appendix A: Species list

Species used in these analyses are listed by taxonomic group and binomial classification. The reason for their inclusion as a species of conservation concern is also given (threat status, restricted range or endemism) and the size of their Extent of Occurrence (EO) is given where available. In addition, the size of their modelled Area of Occupancy (AO) is given with the percentage of this that is currently within protected areas in square brackets. The targets used in these analyses are also given along with the percentage of the target that is already within the current protected area network.

Genus species	Threatened	Restricted Range	Endemic	Notes	EO (km2)	AO (km2)	Target (km2)
Amphibians							
Afrixalus dorsimaculatus	VU				610	236 [47 %]	100 [111 %]
Afrixalus morerei	VU		E		9108	134 [2 %]	100 [3 %]
Afrixalus sylvaticus	EN				8018	20 [28 %]	20 [28 %]
Afrixalus uluguruensis	EN		NE		1000	691 [32 %]	100 [222 %]
Amietophrynus brauni	EN		E		4212	2333 [44 %]	233 [438 %]
Arthroleptis affinis	LC		NE		11065	9083 [37 %]	908 [371 %]
Arthroleptis lonnbergi	DD	RR			21	4 [6 %]	4 [6 %]
Arthroleptis nikeae	EN	RR	E	1	251	246 [11 %]	100 [26 %]
Arthroleptis reichei	NT		NE		11064	4814 [36 %]	481 [359 %]
Arthroleptis stridens	DD	RR		1	12	13 [25 %]	13 [25 %]
Arthroleptis tanneri	VU		E		780	367 [54 %]	100 [200 %]
Arthroleptis xenodactylus	VU		E		1491	567 [41 %]	100 [230 %]
Boulengerula boulengeri	LC		E		1617	491 [24 %]	100 [119 %]
Boulengerula uluguruensis	LC		E		858	475 [61 %]	100 [290 %]
Callulina kisiwamsitu	EN		E		1208	903 [30 %]	100 [275 %]
Callulina kreffti	LC		E		4932	3599 [55 %]	360 [545 %]
Churamiti maridadi	CR	RR	E	1	95	95 [50 %]	95 [50 %]
Hoplophryne rogersi	EN		E		497	254 [60 %]	100 [152 %]
Hoplophryne uluguruensis	VU		E		1352	1011 [58 %]	101 [579 %]
Hyperolius kihangensis	EN		E		420	140 [23 %]	100 [32 %]

Hyperolius minutissimus	VU		NE		7448	17 [0 %]	17 [0 %]
Hyperolius spinigularis	LC		NE		6692	4744 [27 %]	474 [267 %]
Hyperolius tannerorum	EN	RR	E	1	1	1 [100 %]	1 [100 %]
Leptopelis barbouri	VU		NE		2485	1152 [19 %]	115 [189 %]
Leptopelis parkeri	VU		E		3668	490 [50 %]	100 [244 %]
Leptopelis uluguruensis	VU		E		1605	177 [83 %]	100 [147 %]
Leptopelis vermiculatus	VU		NE		4356	2156 [49 %]	216 [493 %]
Mertensophryne usambarae	EN	RR	E		43	15 [86 %]	15 [86 %]
Mertensophryne uzunguensis	VU		NE		14850	2 [0 %]	2 [0 %]
Nectophrynoides cryptus	EN	RR	E		132	95 [70 %]	95 [70 %]
Nectophrynoides frontierei	DD	RR	E	1	1	1 [0 %]	1 [0 %]
Nectophrynoides laevis	DD	RR	E		232	123 [87 %]	100 [107 %]
Nectophrynoides laticeps	EN	RR			17	15 [95 %]	15 [95 %]
Nectophrynoides minutus	EN		E		640	197 [39 %]	100 [76 %]
Nectophrynoides paulae	CR	RR			18	14 [95 %]	14 [95 %]
Nectophrynoides poyntoni	CR	RR	E	1	3	3 [100 %]	3 [100 %]
Nectophrynoides pseudotornieri	EN	RR	E		38	20 [53 %]	20 [53 %]
Nectophrynoides tornieri	LC		NE		3077	1690 [49 %]	169 [490 %]
Nectophrynoides vestergaardi	EN		E		778	640 [34 %]	100 [215 %]
Nectophrynoides viviparus	VU		NE		8225	3883 [27 %]	388 [268 %]
Nectophrynoides wendyae	CR	RR	E		15	11 [100 %]	11 [100 %]
Parhoplophryne usambarica	CR	RR	E		11	9 [31 %]	9 [31 %]
Petropedetes martiensseni	EN		NE		1251	749 [43 %]	100 [319 %]
Petropedetes yakusini	EN		E		2240	1512 [70 %]	151 [703 %]
Phlyctimantis keithae	VU		E		3021	2518 [19 %]	252 [186 %]
Phrynobatrachus breviceps	DD	RR		1	49	50 [0 %]	50 [0 %]
Phrynobatrachus krefftii	EN		E		1808	940 [24 %]	100 [223 %]
Phrynobatrachus uzungwensis	VU		E		592	421 [70 %]	100 [293 %]

Probreviceps durirostris	EN	RR			95	54 [67 %]	54 [67 %]
Probreviceps loveridgei	VU				1261	354 [66 %]	100 [233 %]
Probreviceps macrodactylus	VU		E		1320	782 [50 %]	100 [395 %]
Probreviceps rungwensis	VU		NE		2511	1548 [22 %]	155 [221 %]
Probreviceps uluguruensis	VU		E	2	447	66 [89 %]	66 [89 %]
Scolecomorphus kirkii	LC		NE		18352	7306 [39 %]	731 [395 %]
Scolecomorphus uluguruensis	LC		E		435	220 [63 %]	100 [139 %]
Scolecomorphus vittatus	LC		E		3594	1864 [34 %]	186 [337 %]
Spelaeophryne methneri	LC		NE		158235	5133 [12 %]	513 [122 %]

Birds							
Alethe usambarae	nr	RR			45333	3530 [67 %]	353 [672 %]
Andropadus chlorigula	nr	RR	NE		79930	29 [0 %]	29 [0 %]
Andropadus masukuensis	LC		NE		119876	2858 [76 %]	286 [762 %]
Andropadus milanjensis	LC		NE		134409	778 [63 %]	100 [489 %]
Andropadus neumanni	nr	RR	E		3105	220 [89 %]	100 [196 %]
Anthreptes neglectus	LC	RR			56091	541 [57 %]	100 [310 %]
Anthreptes pallidigaster	EN	RR	NE		6535	591 [77 %]	100 [456 %]
Anthreptes reichenowi	NT				158997	159 [60 %]	100 [95 %]
Anthreptes rubritorques	VU	RR	E		20397	908 [73 %]	100 [663 %]
Apalis chapini	LC		NE		136021	1811 [77 %]	181 [773 %]
Apalis chariessa	VU	RR	NE	3	80645	787 [82 %]	100 [647 %]
Apalis moschi	nr	RR			22977	1376 [68 %]	138 [683 %]
Apalis udzungwensis	nr	RR			4968	513 [84 %]	100 [431 %]
Artisornis metopias	LC	RR	NE		31671	2551 [76 %]	255 [760 %]
Artisornis moreaui	CR	RR	NE		950	63 [80 %]	63 [80 %]
Bathmocercus winifredae	VU	RR	E	3	6800	135 [84 %]	100 [113 %]
Batis crypta	LC	RR			47817	1588 [73 %]	159 [731 %]

Batis mixta	LC		NE		104573	1035 [54 %]	104 [542 %]
Bradypterus cinnamomeus nyassae	nr	RR			29187	805 [38 %]	100 [305 %]
Bradypterus mariae	nr	RR			60976	1754 [72 %]	175 [721 %]
Bubo vosseleri	VU	RR	E		19496	1589 [74 %]	159 [736 %]
Bucorvus cafer	VU				8670000	70 [18 %]	70 [18 %]
Caprimulgus guttifer	LC	RR			38502	962 [31 %]	100 [296 %]
Cinnyricinclus femoralis	VU	RR	NE		19473	169 [28 %]	100 [47 %]
Circaetus fasciolatus	NT				600762	257 [67 %]	100 [172 %]
Cisticola nigriloris	LC		NE		95606	424 [17 %]	100 [73 %]
Cisticola njombe	LC		NE		61261	2 [43 %]	2 [43 %]
Cossypha grotei	nr	RR			16767	293 [90 %]	100 [265 %]
Falco fasciinucha	NT				1707571	2169 [19 %]	217 [188 %]
Francolinus usambarensis	nr	RR			33534	1872 [67 %]	187 [672 %]
Hirundo atrocaerulea	VU				243032	1 [0 %]	1 [0 %]
Hyliota usambara	EN	RR	E		7700	381 [66 %]	100 [253 %]
Illadopsis distans	nr	RR			7452	636 [50 %]	100 [315 %]
Illadopsis mpwapwensis	nr	RR			621	29 [0 %]	29 [0 %]
Illadopsis puguensis	nr	RR			4968	0 [75 %]	0 [75 %]
Illadopsis udzungwensis	nr	RR			11799	1224 [73 %]	122 [731 %]
Laniarius fuelleborni	nr	RR	NE		68969	29 [0 %]	29 [0 %]
Lanius marwitzi	LC		NE	4	76235	3117 [15 %]	312 [154 %]
Malaconotus alius	CR	RR	E	3	2151	354 [3 %]	100 [12 %]
Modulatrix orostruthus	VU		NE		29824	814 [83 %]	100 [672 %]
Modulatrix stictigula	LC	RR	NE		63767	1588 [72 %]	159 [724 %]
Nectarinia loveridgei	EN	RR	E		1871	135 [84 %]	100 [113 %]
Nectarinia moreaui	NT	RR	E		6376	867 [75 %]	100 [651 %]
Nectarinia nyikae	nr	RR			4347	622 [72 %]	100 [449 %]
Nectarinia rufipennis	VU	RR	E		6592	189 [94 %]	100 [177 %]

Nectarinia usambarae	nr	RR	E		5589	287 [80 %]	100 [230 %]
Oriolus chlorocephalus	LC	RR	NE		21987	383 [65 %]	100 [247 %]
Otus ireneae	EN	RR			4535	126 [59 %]	100 [74 %]
Phyllastrephus albigula	nr	RR			4968	468 [72 %]	100 [339 %]
Phyllastrephus alfredi	nr	RR			50688	0 [37 %]	0 [37 %]
Phyllastrephus rabai	nr	RR			35397	5 [14 %]	5 [14 %]
Phyllastrephus udzungwensis	nr	RR			13041	29 [10 %]	29 [10 %]
Ploceus nicolli	EN	RR	E		34041	379 [71 %]	100 [269 %]
Poeoptera kenricki	LC	RR	NE		42317	1238 [77 %]	124 [773 %]
Polemaetus bellicosus	NT				12933722	161 [18 %]	100 [30 %]
Pseudoalcippe abyssinica	LC	RR			70794	1741 [72 %]	174 [724 %]
Schoutedenapus myoptilus	LC	RR			36018	1367 [74 %]	137 [744 %]
Serinus melanochrous	NT	RR	NE		36719	518 [74 %]	100 [384 %]
Sheppardia aurantiithorax	EN	RR	E		550	467 [58 %]	100 [270 %]
Sheppardia gunningi	NT	RR	NE		77750	427 [64 %]	100 [274 %]
Sheppardia lowei	VU	RR	NE		42622	854 [83 %]	100 [704 %]
Sheppardia montana	EN	RR	E		4779	250 [63 %]	100 [156 %]
Sheppardia sharpei	LC	RR	NE		70494	1522 [75 %]	152 [746 %]
Sheppardia usambarae	nr	RR			7452	641 [75 %]	100 [483 %]
Stactolaema howelli	nr	RR			35397	1684 [72 %]	168 [719 %]
Stactolaema olivacea	LC		NE		122920	516 [44 %]	100 [227 %]
Stactolaemus rungweensis	nr	RR			40365	2278 [73 %]	228 [727 %]
Swynnertonia swynnertoni	VU	RR	NE		33227	664 [84 %]	100 [560 %]
Tauraco fischeri	NT	RR		3	47070	639 [49 %]	100 [312 %]
Telacanthura ussheri	LC	RR			75141	211 [80 %]	100 [170 %]
Terathopius ecaudatus	NT			3	13847469	8465 [27 %]	847 [266 %]
Turdus roehli	nr	RR	E		6831	507 [71 %]	100 [360 %]
Xenoperdix obscurata	nr	RR	E		621	21 [35 %]	21 [35 %]

Xenoperdix udzungwensis	EN	RR	E	3457	214 [100 %]	100 [214 %]
Zosterops poliogastrus	LC	RR		249000	46 [51 %]	46 [51 %]
Zosterops winifredae	nr	RR	E	249000	79 [95 %]	79 [95 %]

Chameleons

Chamaeleo deremensis	not listed		E		1182 [42 %]	118 [423 %]
Chamaeleo goetzei	not listed		NE		20 [61 %]	20 [61 %]
Chamaeleo laterispinis	not listed		E		540 [48 %]	100 [259 %]
Chamaeleo tavetanus	not listed	RR			717 [34 %]	100 [244 %]
Chamaeleo tempeli	not listed		NE		912 [35 %]	100 [318 %]
Chamaeleo werneri	not listed		E		1029 [81 %]	103 [807 %]
Kinyongia fischeri	not listed		E		1671 [49 %]	167 [487 %]
Kinyongia oxyrhinum	not listed		E		2226 [65 %]	223 [647 %]
Kinyongia tenue	not listed		E		102 [79 %]	100 [81 %]
Rhampholeon moyeri	not listed		E		1184 [67 %]	118 [666 %]
Rhampholeon spinosum	not listed		E		302 [32 %]	100 [97 %]
Rhampholeon temporalis	not listed		E		32 [74 %]	32 [74 %]
Rhampholeon uluguruensis	not listed		E		104 [85 %]	100 [88 %]
Rieppeleon brevicaudatus	not listed		NE		2363 [62 %]	236 [622 %]

Mammals

Bdeogale jacksoni	NT			64384	1077 [88 %]	108 [882 %]
Bdeogale omnivora	VU			34656	2670 [23 %]	267 [225 %]
Cephalophus spadix	EN	RR	NE	6366	3455 [72 %]	346 [723 %]
Cercocebus sanjei	EN	RR	E	189	52 [96 %]	52 [96 %]
Crocidura desperata	EN	RR	NE	16999	5299 [20 %]	530 [198 %]
Crocidura monax	LC		NE	54105	3430 [19 %]	343 [186 %]
Crocidura tansaniana	EN	RR	E	2851	397 [33 %]	100 [132 %]

Crocidura telfordi	EN	RR	E	4285	1593 [60 %]	159 [596 %]
Crocidura usambarae	EN	RR	E	5096	897 [40 %]	100 [363 %]
Dendrohyrax validus	LC		NE	38059	16161 [36 %]	1616 [356 %]
Diceros bicornis	CR			6733399	38783 [22 %]	3878 [224 %]
Eidolon helvum	NT			11874485	29576 [24 %]	2958 [240 %]
Galagoides cocos	LC	RR		7064	0 [0 %]	0 [0 %]
Galagoides orinus	NT	RR	E	19141	8177 [28 %]	818 [281 %]
Galagoides zanzibaricus	LC	RR	NE	7147	1134 [35 %]	113 [349 %]
Hippopotamus amphibius	VU			1893643	117 [24 %]	100 [29 %]
Hipposideros vittatus	NT			4029230	23319 [23 %]	2332 [231 %]
Hyaena hyaena	NT			23619817	2764 [14 %]	276 [141 %]
Kerivoula africana	EN			29178	828 [22 %]	100 [185 %]
Kobus ellipsiprymnus	NT			9005433	4102 [16 %]	410 [158 %]
Kobus vardonii	NT			373675	25 [10 %]	25 [10 %]
Litocranius walleri	NT			1427700	333 [37 %]	100 [122 %]
Loxodonta africana	NT			6385229	5661 [25 %]	566 [248 %]
Myonycteris relicta	VU		NE	142840	14124 [34 %]	1412 [335 %]
Myosorex geata	EN	RR	E	466	97 [67 %]	97 [67 %]
Myosorex kihaulei	EN	RR	E	1626	1107 [11 %]	111 [110 %]
Otomops martiensseni	NT			7355933	48670 [21 %]	4867 [208 %]
Otomys lacustris	VU	RR		18341	53 [1 %]	53 [1 %]
Panthera leo	VU			5273749	7731 [18 %]	773 [182 %]
Panthera pardus	NT			22307144	36519 [23 %]	3652 [234 %]
Praomys delectorum	NT			56358	5224 [17 %]	522 [170 %]
Procolobus gordonorum	EN	RR	E	5868	4566 [65 %]	457 [648 %]
Rhinolophus deckenii	NT		NE	159243	6444 [17 %]	644 [171 %]
Rhinolophus maendeleo	DD	RR	NE	2716	693 [29 %]	100 [202 %]
Rhynchocyon cirnei	NT			1675280	22859 [25 %]	2286 [246 %]

Rhynchocyon petersi	VU		NE	49118	7462 [25 %]	746 [252 %]
Rhynchocyon udzungwensis	VU	RR		322	88 [83 %]	88 [83 %]
Rungwecebus kipunji	CR	RR	NE	72	21 [100 %]	21 [100 %]
Sylvisorex howelli	EN	RR	E	3841	1796 [36 %]	180 [355 %]
Taphozous hildegardeae	VU			45049	1584 [16 %]	158 [161 %]
Tragelaphus imberbis	NT			1498807	5220 [20 %]	522 [200 %]

Plants

Achyrospermum scandens					561 [85 %]	100 [476 %]
Acridocarpus scheffleri					6 [81 %]	6 [81 %]
Adenia kigogoensis					0 [0 %]	0 [0 %]
Aframomum alpinum					125 [74 %]	100 [93 %]
Aframomum laxiflorum					242 [79 %]	100 [191 %]
Aframomum usambarense					1 [79 %]	1 [79 %]
Afrocanthium siebenlistii					272 [87 %]	100 [238 %]
Agelanthus atrocoronatus					26 [62 %]	26 [62 %]
Agelanthus validus					49 [90 %]	49 [90 %]
Aidia crassifolia					2 [38 %]	2 [38 %]
Allanblackia ulugurensis					761 [83 %]	100 [634 %]
Allophylus grotei					7 [100 %]	7 [100 %]
Allophylus melliodorus					238 [74 %]	100 [176 %]
Aloe brachystachys					93 [73 %]	93 [73 %]
Aloe leptosiphon					3 [93 %]	3 [93 %]
Alsodeiopsis schumannii					1063 [75 %]	106 [749 %]
Ancistrocladus tanzaniensis					20 [74 %]	20 [74 %]
Ancistrorhynchus laxiflorus					623 [78 %]	100 [486 %]
Anisophyllea obtusifolia					4 [74 %]	4 [74 %]
Anisotes spectabilis					11 [100 %]	11 [100 %]

Anisotes tangensis	50 [85 %]	50 [85 %]
Annickia kummerae	28 [71 %]	28 [71 %]
Argomuellera basicordata	21 [65 %]	21 [65 %]
Artabotrys rupestris	133 [77 %]	100 [102 %]
Asparagus usambarensis	38 [64 %]	38 [64 %]
Asystasia schliebenii	11 [66 %]	11 [66 %]
Asystasia tanzaniensis	40 [76 %]	40 [76 %]
Baphia pauloi	5 [50 %]	5 [50 %]
Beilschmiedia kweo	491 [83 %]	100 [405 %]
Bersama rosea	348 [68 %]	100 [237 %]
Bertiera pauloi	359 [81 %]	100 [289 %]
Blotiella hieronymi	110 [55 %]	100 [60 %]
Bothriocline argentea	243 [84 %]	100 [205 %]
Brillantaisia stenopteris	191 [74 %]	100 [141 %]
Bulbophyllum concatenatum	13 [80 %]	13 [80 %]
Callipteris ulugurica	110 [76 %]	100 [84 %]
Casearia engleri	165 [87 %]	100 [144 %]
Chamaecrista mwangokae	28 [55 %]	28 [55 %]
Chamaepentas hindsioides	297 [72 %]	100 [213 %]
Chamaepentas longituba	1490 [80 %]	149 [795 %]
Chamaepentas pseudomagnifica	101 [87 %]	100 [88 %]
Chassalia albiflora	201 [74 %]	100 [149 %]
Chassalia christineae	2 [100 %]	2 [100 %]
Chassalia violacea	133 [81 %]	100 [107 %]
Chassalia zimmermannii	158 [73 %]	100 [116 %]
Cheilanthes deboeri	0 [11 %]	0 [11 %]
Coccinia ulugurensis	22 [55 %]	22 [55 %]
Coffea bridsoniae	1 [23 %]	1 [23 %]

Coffea fadenii	149 [81 %]	100 [120 %]
Coffea kimbozensis	2 [40 %]	2 [40 %]
Coffea mongensis	980 [80 %]	100 [784 %]
Cola scheffleri	146 [87 %]	100 [128 %]
Cola stelechantha	362 [91 %]	100 [331 %]
Cola usambarensis	0 [73 %]	0 [73 %]
Cordia peteri	1 [55 %]	1 [55 %]
Craterispermum longipedunculatum	745 [81 %]	100 [600 %]
Crossandra cephalostachya	258 [90 %]	100 [231 %]
Crotalaria hemsleyi	7 [81 %]	7 [81 %]
Crotalaria inopinata	137 [91 %]	100 [125 %]
Crotalaria mwangulangoi	1 [0 %]	1 [0 %]
Croton dictyophlebodes	26 [79 %]	26 [79 %]
Cryptotaenia calycina	1501 [80 %]	150 [801 %]
Cryptotaenia polygama	0 [0 %]	0 [0 %]
Cyathea fadenii	260 [83 %]	100 [216 %]
Cynometra engleri	2 [28 %]	2 [28 %]
Cynometra longipedicellata	0 [73 %]	0 [73 %]
Cynorkis uncata	387 [67 %]	100 [259 %]
Cyperus longiinvolucratus	278 [89 %]	100 [246 %]
Cyperus purpureoviridis	667 [81 %]	100 [537 %]
Cyphostemma njegerre	0 [73 %]	0 [73 %]
Cyphostemma schliebenii	2 [62 %]	2 [62 %]
Danais xanthorrhoea	875 [79 %]	100 [690 %]
Dichapetalum eickii	1227 [79 %]	123 [787 %]
Dicliptera grandiflora	28 [70 %]	28 [70 %]
Dioscorea longicuspis	38 [51 %]	38 [51 %]
Diospyros uzungwaensis	4 [26 %]	4 [26 %]

Diplazium pseudoporrectum	587 [81 %]	100 [474 %]
Dissotis dichaetantheroides	11 [77 %]	11 [77 %]
Dissotis polyantha	1403 [80 %]	140 [797 %]
Dolichometra leucantha	8 [81 %]	8 [81 %]
Dorstenia bicaudata	1 [62 %]	1 [62 %]
Dorstenia ulugurensis	43 [90 %]	43 [90 %]
Dorstenia variifolia	301 [82 %]	100 [248 %]
Drypetes gerrardinoides	37 [87 %]	37 [87 %]
Duhaldea stuhlmannii	1644 [78 %]	164 [777 %]
Endostemon usambarensis	1 [3 %]	1 [3 %]
Englerina longiflora	113 [94 %]	100 [106 %]
Englerodendron usambarense	2 [91 %]	2 [91 %]
Eragrostis pseudopoa	16 [66 %]	16 [66 %]
Erythrococca sanjensis	148 [90 %]	100 [133 %]
Eugenia mufindiensis	4 [0 %]	4 [0 %]
Eugenia toxanatolica	1237 [80 %]	124 [798 %]
Garcinia bifasciculata	1 [100 %]	1 [100 %]
Garcinia semseii	242 [76 %]	100 [185 %]
Gomphia scheffleri	1599 [79 %]	160 [792 %]
Gouania ulugurica	3 [91 %]	3 [91 %]
Gravesia hylophila	34 [84 %]	34 [84 %]
Gravesia pulchra	897 [80 %]	100 [719 %]
Gravesia riparia	468 [78 %]	100 [364 %]
Gymnosiphon usambaricus	1590 [79 %]	159 [793 %]
Gymnosporia schliebenii	186 [81 %]	100 [151 %]
Harveya tanzanica	12 [92 %]	12 [92 %]
Helixanthera verruculosa	1 [56 %]	1 [56 %]
Hydrostachys angustisecta	163 [96 %]	100 [157 %]

Impatiens cinnabarina	1 [100 %]	1 [100 %]
Impatiens engleri	1141 [77 %]	114 [775 %]
Impatiens hamata	68 [79 %]	68 [79 %]
Impatiens joachimii	67 [95 %]	67 [95 %]
Impatiens kentrodonta	1133 [79 %]	113 [793 %]
Impatiens lukwangulensis	243 [85 %]	100 [208 %]
Impatiens mahengeensis	2 [10 %]	2 [10 %]
Impatiens palliderosea	452 [78 %]	100 [351 %]
Impatiens polhillii	4 [95 %]	4 [95 %]
Impatiens serpens	41 [81 %]	41 [81 %]
Impatiens teitensis	210 [83 %]	100 [174 %]
Impatiens thamnoidea	56 [79 %]	56 [79 %]
Impatiens ukagurensis	3 [74 %]	3 [74 %]
Impatiens ulugurensis	175 [83 %]	100 [145 %]
Impatiens usambarensis	71 [55 %]	71 [55 %]
Impatiens uzungwaensis	1 [19 %]	1 [19 %]
Isoglossa asystasioides	1 [100 %]	1 [100 %]
Isoglossa bondwaensis	12 [92 %]	12 [92 %]
Isoglossa candelabrum	13 [77 %]	13 [77 %]
Isoglossa oreacanthoides	2 [74 %]	2 [74 %]
Isolona linearis	344 [80 %]	100 [275 %]
Ixora albersii	68 [90 %]	68 [90 %]
Juncus engleri	11 [99 %]	11 [99 %]
Justicia beloperonoides	8 [90 %]	8 [90 %]
Justicia bridsoniana	28 [100 %]	28 [100 %]
Justicia lukei	23 [43 %]	23 [43 %]
Justicia mkungweensis	1 [100 %]	1 [100 %]
Justicia oblongifolia	0 [0 %]	0 [0 %]

Justicia roseobracteata	2 [100 %]	2 [100 %]
Justicia sulphuriflora	1045 [77 %]	105 [770 %]
Justicia ukagurensis	1075 [74 %]	108 [744 %]
Keetia carmichaelii	81 [94 %]	81 [94 %]
Keetia koritschoneri	122 [68 %]	100 [82 %]
Keetia lulandensis	1 [94 %]	1 [94 %]
Lasianthus cereiflorus	119 [88 %]	100 [105 %]
Lasianthus glomeruliflorus	228 [81 %]	100 [184 %]
Lasianthus macrocalyx	103 [89 %]	100 [92 %]
Lasianthus microcalyx	43 [89 %]	43 [89 %]
Lasianthus pedunculatus	1466 [80 %]	147 [797 %]
Lasianthus wallacei	118 [82 %]	100 [97 %]
Lefebvrea droopii	398 [76 %]	100 [304 %]
Leptoderris harmsiana	30 [58 %]	30 [58 %]
Lijndenia brenanii	184 [84 %]	100 [153 %]
Lijndenia procteri	80 [93 %]	80 [93 %]
Lingelsheimia sylvestris	1 [100 %]	1 [100 %]
Lobelia gilgii	179 [77 %]	100 [138 %]
Lobelia longisepala	854 [81 %]	100 [695 %]
Lobelia lukwangulensis	68 [87 %]	68 [87 %]
Mammea usambarensis	14 [98 %]	14 [98 %]
Medinilla engleri	1140 [80 %]	114 [796 %]
Meineckia acuminata	70 [56 %]	70 [56 %]
Meineckia paxii	40 [71 %]	40 [71 %]
Memecylon cogniauxii	1236 [80 %]	124 [796 %]
Memecylon deminutum	82 [93 %]	82 [93 %]
Memecylon greenwayi	193 [65 %]	100 [126 %]
Memecylon myrtilloides	586 [85 %]	100 [497 %]

Memecylon semseii	63 [66 %]	63 [66 %]
Memecylon teitense	7 [81 %]	7 [81 %]
Microlepia fadenii	269 [80 %]	100 [214 %]
Millettia sacleuxii	20 [71 %]	20 [71 %]
Mitriostigma usambarense	81 [94 %]	81 [94 %]
Monadenium heteropodum	5 [94 %]	5 [94 %]
Monanthotaxis dictyoneura	9 [60 %]	9 [60 %]
Monanthotaxis discrepantinervia	91 [90 %]	91 [90 %]
Monodora globiflora	256 [96 %]	100 [245 %]
Mwasumbia alba	1 [100 %]	1 [100 %]
Mystacidium pulchellum	571 [79 %]	100 [453 %]
Neohemsleya usambarensis	1 [99 %]	1 [99 %]
Octoknema orientalis	518 [85 %]	100 [441 %]
Oldenlandia oxycoccoides	61 [73 %]	61 [73 %]
Omphalocarpum strombocarpum	114 [91 %]	100 [103 %]
Oncella gracilis	74 [50 %]	74 [50 %]
Palisota orientalis	25 [71 %]	25 [71 %]
Parapentas silvatica	1673 [77 %]	167 [766 %]
Pauridiantha coalescens	353 [84 %]	100 [295 %]
Pauridiantha hirsuta	288 [82 %]	100 [236 %]
Pavetta amaniensis	131 [58 %]	100 [76 %]
Pavetta axillipara	33 [97 %]	33 [97 %]
Pavetta bruceana	20 [89 %]	20 [89 %]
Pavetta coelophlebia	5 [93 %]	5 [93 %]
Pavetta constipulata	12 [78 %]	12 [78 %]
Pavetta diversicalyx	0 [0 %]	0 [0 %]
Pavetta filistipulata	30 [81 %]	30 [81 %]
Pavetta holstii	721 [74 %]	100 [537 %]

Pavetta manyanguensis	104 [93 %]	100 [97 %]
Pavetta mazumbaiensis	234 [85 %]	100 [200 %]
Pavetta mufindiensis	305 [80 %]	100 [243 %]
Pavetta nitidissima	192 [92 %]	100 [177 %]
Pavetta olivaceonigra	3 [56 %]	3 [56 %]
Pavetta sparsipila	716 [73 %]	100 [525 %]
Peddiea lanceolata	1 [54 %]	1 [54 %]
Peddiea puberula	56 [85 %]	56 [85 %]
Peddiea subcordata	1379 [79 %]	138 [790 %]
Phyllanthus mittenianus	2 [81 %]	2 [81 %]
Phyllanthus thulinii	12 [59 %]	12 [59 %]
Phyllopentas ionolaena	444 [79 %]	100 [351 %]
Pittosporum goetzei	24 [95 %]	24 [95 %]
Placodiscus amaniensis	5 [62 %]	5 [62 %]
Platypterocarpus tanganyikensis	1 [94 %]	1 [94 %]
Plectranthus bracteolatus	325 [85 %]	100 [278 %]
Plectranthus dichotomus	13 [72 %]	13 [72 %]
Plectranthus scopulicola	0 [100 %]	0 [100 %]
Plectranthus strangulatus	2 [89 %]	2 [89 %]
Plectranthus triangularis	559 [83 %]	100 [465 %]
Plectranthus trullatus	103 [93 %]	100 [95 %]
Pneumatopteris usambarensis	11 [92 %]	11 [92 %]
Pollia bracteata	415 [78 %]	100 [323 %]
Polyceratocarpus scheffleri	878 [81 %]	100 [711 %]
Polygala multifurcata	0 [0 %]	0 [0 %]
Polyscias stuhlmannii	625 [82 %]	100 [515 %]
Polysphaeria macrantha	412 [85 %]	100 [350 %]
Polystachya canaliculata	2 [84 %]	2 [84 %]

Polystachya caudata	53 [43 %]	53 [43 %]
Polystachya longiscapa	47 [77 %]	47 [77 %]
Polystachya mazumbaiensis	14 [47 %]	14 [47 %]
Polystachya pudorina	14 [62 %]	14 [62 %]
Polystachya serpentina	0 [0 %]	0 [0 %]
Polystachya shega	2 [80 %]	2 [80 %]
Polystachya uluguruensis	18 [78 %]	18 [78 %]
Pseuderanthemum campylosiphon	1184 [80 %]	118 [803 %]
Psychotria brevicaulis	13 [47 %]	13 [47 %]
Psychotria brucei	574 [86 %]	100 [492 %]
Psychotria castaneifolia	2 [100 %]	2 [100 %]
Psychotria diploneura	230 [75 %]	100 [173 %]
Psychotria elachistantha	647 [82 %]	100 [528 %]
Psychotria griseola	543 [70 %]	100 [378 %]
Psychotria iringensis	54 [93 %]	54 [93 %]
Psychotria megalopus	447 [81 %]	100 [361 %]
Psychotria megistantha	167 [85 %]	100 [142 %]
Psychotria pandurata	242 [75 %]	100 [181 %]
Psychotria peteri	32 [71 %]	32 [71 %]
Psychotria pocsii	5 [53 %]	5 [53 %]
Psychotria porphyroclada	154 [68 %]	100 [104 %]
Psychotria triclada	172 [60 %]	100 [103 %]
Psychotria usambarensis	316 [84 %]	100 [266 %]
Psychotria verdcourtii	17 [53 %]	17 [53 %]
Pycnocoma macrantha	61 [70 %]	61 [70 %]
Pyrostria uzungwaensis	426 [85 %]	100 [363 %]
Pyrrosia liebuschii	11 [79 %]	11 [79 %]
Rhipidantha chlorantha	69 [79 %]	69 [79 %]

Rinorea scheffleri	17 [74 %]	17 [74 %]
Rytigynia caudatissima	384 [81 %]	100 [313 %]
Rytigynia hirsutiflora	569 [75 %]	100 [427 %]
Rytigynia longicaudata	144 [87 %]	100 [125 %]
Rytigynia longituba	2 [100 %]	2 [100 %]
Rytigynia pseudolongicaudata	1121 [78 %]	112 [779 %]
Saintpaulia pusilla	822 [80 %]	100 [661 %]
Saintpaulia shumensis	24 [84 %]	24 [84 %]
Sanrafaelia ruffonammari	71 [79 %]	71 [79 %]
Sclerochiton glandulosissimus	449 [87 %]	100 [393 %]
Sclerochiton uluguruensis	42 [90 %]	42 [90 %]
Seychellaria africana	3 [100 %]	3 [100 %]
Solanecio buchwaldii	69 [42 %]	69 [42 %]
Sorindeia calantha	745 [84 %]	100 [623 %]
Sorindeia usambarensis	14 [61 %]	14 [61 %]
Stapfiella ulugurica	17 [89 %]	17 [89 %]
Stapfiella usambarica	48 [98 %]	48 [98 %]
Stenandrium afromontanum	350 [72 %]	100 [253 %]
Stenandrium warneckei	49 [80 %]	49 [80 %]
Stolzia atrorubra	55 [80 %]	55 [80 %]
Stolzia christopheri	55 [89 %]	55 [89 %]
Stolzia leedalii	83 [74 %]	83 [74 %]
Stolzia moniliformis	40 [79 %]	40 [79 %]
Stolzia viridis	62 [77 %]	62 [77 %]
Streptocarpus albus	23 [81 %]	23 [81 %]
Streptocarpus bambuseti	81 [94 %]	81 [94 %]
Streptocarpus bullatus	8 [100 %]	8 [100 %]
Streptocarpus gonjaensis	3 [83 %]	3 [83 %]

Streptocarpus heckmannianus	24 [93 %]	24 [93 %]
Streptocarpus hirsutissimus	12 [92 %]	12 [92 %]
Streptocarpus inflatus	195 [89 %]	100 [173 %]
Streptocarpus kimbozanus	1 [100 %]	1 [100 %]
Streptocarpus parensis	1 [100 %]	1 [100 %]
Streptocarpus schliebenii	795 [80 %]	100 [638 %]
Streptocarpus stomandrus	132 [90 %]	100 [118 %]
Streptocarpus thysanotus	4 [100 %]	4 [100 %]
Syzygium parvulum	47 [82 %]	47 [82 %]
Tarenna roseicosta	779 [72 %]	100 [563 %]
Ternstroemia polypetala	853 [83 %]	100 [707 %]
Tetrorchidium ulugurense	42 [90 %]	42 [90 %]
Thunbergia hamata	0 [0 %]	0 [0 %]
Thunbergia schliebenii	12 [94 %]	12 [94 %]
Tournefortia usambarensis	83 [80 %]	83 [80 %]
Toussaintia patriciae	38 [100 %]	38 [100 %]
Tricalysia aciculiflora	354 [89 %]	100 [314 %]
Trichilia lovettii	69 [92 %]	69 [92 %]
Tridactyle minuta	11 [46 %]	11 [46 %]
Tridactyle tanneri	124 [57 %]	100 [71 %]
Turraea kimbozensis	2 [60 %]	2 [60 %]
Urogentias ulugurensis	151 [76 %]	100 [114 %]
Uvaria dependens	34 [78 %]	34 [78 %]
Uvariodendron oligocarpum	67 [64 %]	67 [64 %]
Uvariodendron pycnophyllum	64 [66 %]	64 [66 %]
Uvariodendron usambarense	60 [70 %]	60 [70 %]
Uvariopsis bisexualis	116 [99 %]	100 [115 %]
Uvariopsis lovettiana	407 [87 %]	100 [354 %]

Vangueria bicolor	25 [80 %]	25 [80 %]
Vangueria fuscosetulosa	233 [87 %]	100 [203 %]
Vangueria rufescens	301 [85 %]	100 [255 %]
Vangueriopsis longiflora	32 [100 %]	32 [100 %]
Vernonia amaniensis	205 [60 %]	100 [122 %]
Vernonia bruceae	2 [100 %]	2 [100 %]
Vernonia luhomeroensis	40 [100 %]	40 [100 %]
Vernonia nuxioides	7 [79 %]	7 [79 %]
Vernonia ruvungatundu	114 [81 %]	100 [92 %]
Viscum engleri	675 [75 %]	100 [505 %]
Viscum luisengense	1 [0 %]	1 [0 %]
Vitex amaniensis	69 [86 %]	69 [86 %]
Warneckea erubescens	23 [75 %]	23 [75 %]
Warneckea microphylla	14 [57 %]	14 [57 %]
Zenkerella capparidacea	628 [78 %]	100 [493 %]
Zygophlebia major	1 [100 %]	1 [100 %]

Codes: 1: no suitable habitat within EO, so classed all EO as suitable; **2**: used score of 1 as threshold, as this was maximum; **3**: used score of 3 as threshold, as this was maximum; **4**: used score of 2 as threshold, as this was maximum; **LC**: least concern; **NT**: near-threatened, **VU**: vulnerable; **EN**: endangered; **CR**: critically endangered; **DD**: data deficient; **nr**: not recognised; **RR**: restricted range; **E**: endemic; **NE**: near-endemic.

Appendix B: Species used to validate proxies for biological processes

Old taxa. Species' names (Fjeldså and Tushabe 2005; Fjeldså 2007; Fjeldså et al. 2010) and endemism status (according to Burgess et al. 2007c) are given. A taxon is considered restricted-range if its extent of occurrence was less than 82,000 km^2 (see chapter 2). Taxon ages are based on Fjeldså and Lovett (1997a) and J. Fjeldså (pers. comm.). This list is a subset of those species of conservation concern that were mapped as part of chapter two; therefore, there will be other old species that are not of conservation concern and are consequently not included here.

Species name	Endemism status	Restricted range
Anas sparsa		
Andropadus importunus		
Guttera pucherani		
Hyliota usambara	Endemic	Yes
Indicator variegatus		
Macrosphenus kretschmeri		
Malaconotus alius	Endemic	Yes
Modulatrix orostruthus	Near Endemic	
Modulatrix stictigula	Near Endemic	Yes
Nicator gularis		
Pitta angolensis		
Smithornis capensis		
Xenoperdix obscurata	Endemic	Yes
Xenoperdix udzungwensis	Endemic	Yes

Young, restricted-range taxa. Species' names (Fjeldså and Tushabe 2005; Fjeldså 2007; Fjeldså et al. 2010) and endemism status (according to Burgess et al. 2007c) are given. A taxon is considered restricted-range if its extent of occurrence is less than 82,000 km^2 (see chapter 2). Taxon ages are based on Fjeldså and Lovett (1997a) and J. Fjeldså (pers. comm.). This list is a subset of those species of conservation concern that were mapped as part of chapter two. Restricted-range species were included as species of conservation concern, so this list should include all young, restricted range taxa that are known.

Species name	Endemism status	Restricted range
Andropadus chlorigula	Near Endemic	Yes
Andropadus neumanni	Endemic	Yes
Apalis moschi		Yes
Apalis udzungwensis		Yes
Artisornis metopias	Near Endemic	Yes
Artisornis moreaui	Near Endemic	Yes
Batis crypta		Yes
Bradypterus cinnamomeus nyassae		Yes
Bradypterus mariae		Yes
Bubo vosseleri	Endemic	Yes
Caprimulgus guttifer		Yes
Cossypha anomala		Yes
Cossypha grotei		Yes
Francolinus usambarensis		Yes
Illadopsis distans		Yes
Illadopsis puguensis		Yes
Illadopsis udzungwensis		Yes
Laniarius fuelleborni	Near Endemic	Yes
Nectarinia fuelleborni	Near Endemic	Yes
Nectarinia loveridgei	Endemic	Yes
Nectarinia moreaui	Endemic	Yes
Nectarinia nyikae		Yes
Nectarinia usambarae	Endemic	Yes
Otus ireneae		Yes
Phyllastrephus alfredi		Yes
Phyllastrephus udzungwensis		Yes
Ploceus nicolli	Endemic	Yes
Poeoptera kenricki	Near Endemic	Yes
Pseudoalcippe abyssinica		Yes
Scepomycter winifredae	Endemic	Yes
Serinus melanochrous	Near Endemic	Yes
Sheppardia aurantiithorax	Endemic	Yes
Sheppardia gunningi	Near Endemic	Yes
Sheppardia lowei	Near Endemic	Yes
Sheppardia montana	Endemic	Yes
Sheppardia sharpei	Near Endemic	Yes
Sheppardia usambarae		Yes
Zosterops poliogastrus		Yes
Zosterops winifredae	Endemic	Yes

Appendix C: Targets for restoration of forested altitudinal gradients

The altitude mean, minimum, maximum and range (in metres above sea level) is given for each mountain bloc and for the forested areas within each bloc. In addition, the maximum altitudinal range for forested gradients within each bloc is also given. Blocs for which restoration of forest could be prioritised between low and high altitude forest are indicated with asterisk (*).

	Altitudinal Range of Mountain Bloc				Altitudinal Range of Forested Area				Altitudinal Range of Continuous Forested Patches		
Bloc	**Mean**	**Min.**	**Max.**	**Range**	**Mean**	**Min.**	**Max.**	**Range**	**Max. range**	**% of bloc range**	**% of forested range**
N. Pare	1130	697	2099	1402	1652	1210	2099	889	747	53	84
S. Pare	1065	459	2454	1995	1788	962	2454	1492	1244	62	83
W. Usambara	1128	290	2294	2004	1591	424	2294	1870	1451	72	78
E. Usambara	525	123	1501	1378	747	124	1501	1377	1276	93	93
Nguu	1094	676	1987	1311	1204	709	1987	1278	872	67	68*
Nguru	945	351	2382	2031	1375	413	2382	1969	1941	96	99
Ukaguru	1126	412	2259	1847	1722	1048	2259	1211	1006	54	83
Rubeho	1125	272	2345	2073	1653	515	2345	1830	1468	71	80
Uluguru	753	119	2636	2517	1592	128	2636	2508	1675	67	67*
Malundwe	834	488	1259	771	1093	842	1259	417	417	54	100
Udzungwa	1428	249	2556	2307	1389	278	2556	2278	1887	82	83
Mahenge	735	323	1501	1178	1108	519	1482	963	625	53	65*

Appendix D: Farmer Survey

Farmer questionnaire v. 8
19th July 2009

Jonathan Green

UNIVERSITY OF CAMBRIDGE

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Informed consent form

I am asking you to take part in this survey, so that I can investigate the way land value varies in the Eastern Arc mountains of Tanzania. I will show that, when planning for nature conservation, government must carefully consider how these plans affect people in the area. It is my hope that this will provide benefits for both wildlife and communities.

Some reports may be written using the information you give me. However, information will be anonymous, and it will not be possible for information to be traced back to an individual.

Please be aware that if you consent to take part in this survey now, but later change your mind, you can still withdraw your consent for any future work by contacting me. It would be very helpful to me if you can answer as many questions as you are able.

If you have any further questions, concerns or complaints about this survey, please contact Jonathan Green (contact details at the top of this page).

Please answer the following questions:

1. Have you had an opportunity to ask questions and discuss the study? yes / no
2. Do you understand that you may withdraw from this study at any time? yes / no
3. Do you agree to the use of this information in publications? yes / no
4. Do you agree to take part in this study? yes / no

Participant signature: ____________________ Researcher signature: ____________________

Participant name: ____________________ Researcher name: ____________________

Date: ____________________ Date: ____________________

Unique site code:	Initials				Date									Cell ID						Farmer #			Field	
	J	M	H	G	-	D	D	/	M	M	/	Y	Y	-	0	0	0	0	1	-	0	1	-	a

Questionnaire on the costs of forest protection to communities living in the Eastern Arc mountains of Tanzania

To be completed by the researcher

GPS coordinates - field centre: North: .... South: .... Units: ....

Altitude: ....metres above sea level (approx. is ok)

Land cover around farm? (tick any that apply)

Type	Bare soil	Bushland	Bush with sca-ttered crops	Closed woodland	Open woodland	Forest mosaic	Grassland
Tick any that apply							

Grass with sca-ttered crops	Cultivation	Monocrop	Plantation (sisal, teak, rice, tea, sugarcane, rubber)	Rock outcr ops	Urban area	Wat er	Woodland with scattered crops	Forest
Permanent swamp	Plantation forest							

If possible, walk around field and take GPS readings so that we can verify area later

If respondent is unsure, write 'unsure'. If the question is not applicable, write 'N/A'.

FARM/FIELD DETAILS – ALL QUESTIONS RELATE TO *THIS* FIELD		
HOW LARGE IS THIS FIELD? *(field refers to continuous area of land owned/rented by focal farmer, including fallow land)*	**SIZE:**	**UNITS:**
HOW DO YOU MEASURE UNITS?		
DO YOU *OWN* OR *RENT* THIS LAND? **OWN / RENT / NEITHER** (STATE: ……………………………)		
IF YOU OWN THIS LAND, HOW DID YOU AQUIRE IT? **PURCHASED / INHERITED / OTHER** (STATE: ……………………………)		
IF YOU WERE TO SELL THIS PLOT TODAY, HOW MUCH WOULD YOU SELL FOR?	**TSH**	
IF YOU WERE TO BUY THIS PLOT TODAY, HOW MUCH WOULD YOU BUY FOR?	**TSH**	
INITIAL COST:	**ANNUAL COST:**	**YEAR START/BOUGHT:**

HOW LONG IS IT SINCE THIS FIELD WAS CONVERTED FROM NATURAL VEGETATION (FOREST, MIOMBO, GRASSLAND)? (if different parts converted at different times, indicate number of hectares for each time period - nb: Lemineh et al 2005)						
☐ y < 1yr	☐ 1yr ≤ y <	☐ 3yrs ≤ y <	☐ 5yrs ≤ y <	☐ 10yrs ≤ y <	☐ y ≥ 20yrs	☐ Unknown

WHAT WAS THE VEGETATION TYPE BEFORE THIS LAND WAS CULTIVATED? ***(PLEASE CIRCLE)***
FOREST **MIOMBO** **GRASSLAND** **BUSHLAND** **UNSURE** **OTHER:** ……………………

DID YOU CLEAR THIS LAND FOR CULTIVATION YOURSELF? **YES / NO**

NOTES:

HOW MUCH OF EACH OF THE FOLLOWING WAS CLEARED AND SOLD?							
	AMOUNT	UNITS	SPECIES/CLASS(TIMBER)	OWN USE	SOLD	PRICE PER UNIT	YEAR
TIMBER 1							
TIMBER 2							
TIMBER 3							
FIREWOOD							
CHARCOAL							
POLES							
OTHER							

COSTS OF CONVERTING LAND:		
HOW MANY PEOPLE WOULD YOU EMPLOY AND FOR HOW LONG TO CLEAR THIS AREA FOR FARMING?		
NO. OF PEOPLE:	NO. OF DAYS:	COST PER PERSON PER DAY:
TOOLS USED:	TOTAL COST:	

OTHER COST	TSH	IN KIND	MAN DAYS LABOUR	SELF/HIRE
LAND CLEARING PERMIT				
TREEFELLING				
CUTTING TRUNKS				
BURNING				
PERMIT TO MAKE CHARCOAL OUT OF TREES				
COST FOR PAYING PEOPLE TO TRANSPORT POLES/CHARCOAL/TIMBER/FIREWOOD				
COST FOR PROCESSING CHARCOAL (DESCRIBE)				
COST FOR PROCESSING TIMBER (DESCRIBE)				
COST FOR PROCESSING POLES (DESCRIBE)				
COST FOR PROCESSING FIREWOOD (DESCRIBE)				
OTHER				

CAPITAL COSTS (TOOLS: AXES, HOES, MACHETES, ETC)	OWN / HIRED	NUMBER	COST	CROP USED FOR	EXPECTED LIFETIME	USED ON THIS FIELD ONLY?	
						YES / NO	IF NO, TOTAL AREA WHERE TOOL USED
1.							
2.							
3.							
4.							
5.							

FARMING ACTIVITIES – ALL QUESTIONS RELATE TO *THIS* FIELD						
WHAT ARE THE QUANTITIES, USES AND VALUES OF CROPS THAT YOU HAVE HARVESTED (OR EXPECT TO HARVEST) FROM THIS FIELD IN THE PAST OR CURRENT SEASON? (FOR CONTINUALLY HARVESTED CROPS, DO WEEKLY YIELDS)						
CROP TYPE	AGE	AREA (+ UNITS)	YIELD	WEEKLY YIELD	UNITS	NO. SEASONS / WEEKS
			1 2 3 4 5 6 7 8 9 10 11 12			
			1 2 3 4 5 6 7 8 9 10 11 12			
			1 2 3 4 5 6 7 8 9 10 11 12			
			1 2 3 4 5 6 7 8 9 10 11 12			
			1 2 3 4 5 6 7 8 9 10 11 12			
			1 2 3 4 5 6 7 8 9 10 11 12			
NOTES:						

FOR THE CROPS IDENTIFIED ABOVE, PLEASE DESCRIBE THEIR LIFECYCLES:						
CROP ->	1.	2.	3.	4.	5.	6.
EARLY GROWTH (NO HARVEST)						
IMMATURE (LOW HARVEST)						
MATURE (PRIME HARVEST)						
OLD/REPLACE (LOW HARVEST)						
OTHER						

COSTS OF INPUTS TO CROP FARMING ACTIVITIES:							
	TYPE/ DESCRIPTION	AMOUNT PER SEASON/ APPLICATION	UNITS	COST PER UNIT	NO SEASONS/ APPLICATIONS PER YEAR	MAN DAYS	
PREPARING LAND							
SEED/SEEDLING 1							
SEED/SEEDLING 2							
SEED/SEEDLING 3							
SEED/SEEDLING 4							
SEED/SEEDLING 5							
SEED/SEEDLING 6							
PEGGING (INCL. CUTTING /CARRYING PEGS)							
CARRYING/PLANTING 1							
CARRYING/PLANTING 2							
CARRYING/PLANTING 3							
CARRYING/PLANTING 4							
CARRYING/PLANTING 5							
CARRYING/PLANTING 6							
FERTILISING							
SPRAYING (HERBICIDE, FUNGICIDE, PESTICIDE)							
WEEDING							
HARVESTING 1							
HARVESTING 2							
HARVESTING 3							
HARVESTING 4							
HARVESTING 5							
HARVESTING 6							
AFTER HARVEST TIDYING/PREPARATION							

SPECIALISED EQUIPMENT TYPE	NUMBER	COST	CROPS USED FOR	EXPECTED LIFETIME	USED ON THIS FIELD ONLY?	
					YES /NO	IF NO, TOTAL AREA WHERE TOOL USED
1.						
2.						
3.						
4.						
5.						

IRRIGATION - HOW MUCH OF THIS FIELD DO YOU IRRIGATE?												
TOTAL AREA:				UNITS (AREA):					UNITS (AMOUNT PER DAY - BELOW):			
TIME OF YEAR IRRIGATED:	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
AMOUNT PER DAY:												
CROP TYPES												
AREA IRRIGATED												
METHOD:												

COSTS OF IRRIGATION:						
	TYPE/ DESCRIPTION	NUMBER	COST	EXPECTED LIFETIME	SHARED?	NOTES:
HOSE/BUCKET/ETC						
HOSE/BUCKET/ETC						
WATER USE PERMIT		N/A				
NOTES:						

DESCRIBE HOW YOU MANAGE YOUR CROPS: (E.G. INTERCROPPING, ROTATION, FALLOW LAND ETC)

DO YOU SELL YOUR CROPS?				YES / NO							
CROP	TOTAL YIELD - CHECK EARLIER	UNITS	AMOUNT LOST/ WASTED	OWN USE	SOLD	PRICE PER UNIT	DOES PRICE VARY?		PROCESSING (E.G. DEHUSKING, MILLING) PRIOR TO SELLING	COST OF PROCESSING (NOT INCLUDED IN EARLIER COSTS)	NEXT STAGE FOR YOUR CROP
							1 2 3 4 5 6 7 8 9 10 11 12				
								Indicate high low			
								Indicate when sold			
							1 2 3 4 5 6 7 8 9 10 11 12				
								Indicate high low			
								Indicate when sold			
							1 2 3 4 5 6 7 8 9 10 11 12				
								Indicate high low			
								Indicate when sold			
							1 2 3 4 5 6 7 8 9 10 11 12				
								Indicate high low			
								Indicate when sold			
							1 2 3 4 5 6 7 8 9 10 11 12				
								Indicate high low			
								Indicate when sold			
							1 2 3 4 5 6 7 8 9 10 11 12				
								Indicate high low			
								Indicate when sold			

WHERE DO YOU SELL YOUR PRODUCE?							
	YES / NO	BARTER?			LOCATION NAME	LOCATION DESCRIPTION	PRODUCTS SOLD
		ALL	SOME	NONE			
NEIGHBOURS							
LOCAL (E.G. VILLAGE) MARKET							
ROADSIDE							
DISTRICT MARKET							
REGIONAL MARKET							
TRADER AT FARM							
MARKETING COOPERATIVE							
FARMERS' ASSOCIATIONS							
OTHER							

DO YOU STORE YOUR HARVEST?							
TYPE	STORAGE PERIOD	AMOUNT STORED	STORAGE LOCATION	AREA REQUIRED	RENT PAID	HARVEST LOSS DURING STORAGE	USE AFTER STORAGE

WHAT OTHER WORK IS AVAILABLE TO YOU AND HOW MUCH IS IT PAID?					
HOUSEHOLD MEMBER	TYPE OF WORK	DAYS WORKED IN PAST YEAR	DAILY WAGE	TOTAL EXPECTED INCOME	TOTAL ACTUAL INCOME

DAMAGE BY *WILD* ANIMALS				
DOES THIS FIELD EVER GET DAMAGED BY WILD ANIMALS? YES / NO				
HOW REGULARLY IS THIS FIELD DAMGED BY WILD ANIMALS?				
DESCRIPTION OF THE *WORST* CASE OF DAMAGE BY WILD ANIMALS TO THIS FIELD IN PAST *12 MONTHS*:				
ANIMAL:		DATE (APPROX. IS OK):		
ORIGIN OF ANIMAL:		APPROX. DISTANCE TO ANIMAL'S ORIGIN:		
DAMAGE:				
CROP	AREA	ESTIMATE YIELD	ESTIMATE COST	CROP AGE

COULD ANY OF THE DAMAGED CROP BE USED OR SOLD? YES / NO			
CROP	AMOUNT USED	AMOUNT SOLD	NOTES

DESCRIPTION OF THE *WORST* CASE OF DAMAGE BY WILD ANIMALS TO THIS FIELD *EVER*:				
ANIMAL:		DATE (APPROX. IS OK):		
ORIGIN OF ANIMAL:		APPROX. DISTANCE TO ANIMAL'S ORIGIN:		
DAMAGE:				
CROP	AREA	ESTIMATE YIELD	ESTIMATE COST	CROP AGE

COULD ANY OF THE DAMAGED CROP BE USED OR SOLD? YES / NO			
CROP	AMOUNT USED	AMOUNT SOLD	NOTES

PAST 12 MONTHS: (FOR SEVERITY, USE A SCALE OF 1-3, WHERE 3 IS EQUAL TO THE WORST *EVER* INCIDENT). IF NO INCIDENTS IN THE LAST 12 MONTHS, PLEASE GIVE DETAILS OF THE *LAST* INCIDENT.				
DATE	**SEVERITY (1-3)**	**CROPS DAMAGED**	**ANIMAL**	**NOTES**

DID YOU DO ANYTHING TO REDUCE OR PREVENT WILDLIFE DAMAGE?					
ACTION	**YES /NO**	**COST MATERIALS**	**NUMBER DAYS/ NIGHTS LABOUR**	**HIRED LABOUR OR HOUSEHOLD MEMBER?**	**COST OF HIRED LABOUR PER DAY/NIGHT***
TRAPS IN FIELD					
FENCING AROUND FIELD					
I WATCHED OVER MY FIELDS DURING DAY					
I PAID SOMEONE TO WATCH OVER MY FIELDS					
OTHER					
***NOTE PAYMENTS IN KIND WITH ESTIMATE OF VALUE ASWELL AS NOTING WHAT WAS GIVEN**					

DID YOUR EFFORTS TO REDUCE OR PREVENT WILDLIFE DAMAGE INVOLVE CATCHING ANIMALS ENTERING THE FIELD?					
SPECIES	**NUMBER ADULT**	**NUMBER YOUNG**	**OWN USE**	**SOLD**	**PRICE**
NGURUWE (WILD PIG)					
NYANI (BABOON)					
NGEDELE (VERVET MONKEY)					
TANDALA/SWALA/DIGI-DIGI (ANTELOPE)					
OTHER (SPECIFY)					

PERSONAL DETAILS					
NAME:				**SEX:**	
PROFESSION:				**AGE:**	
RELATIONSHIP OF RESPONDENT TO HEAD OF HOUSEHOLD:		**SAME**	**SPOUSE**	**CHILD**	**OTHER........................**
DO YOU HAVE DEPENDENTS (PLEASE STATE)?	**WIFE:**	**CHILD < 5YR:**	**CHILD <10YRS:**	**CHILD <18YRS:**	**OTHER:**

Appendix E: Calculating net rent

Spatially explicit data on maize and bean yield (kg ha^{-1} y^{-1}) are from Thornton et al. (2009). These show predicted yield of maize across East Africa under typical current smallholder farmer practices and, where climatic conditions allow for a second crop to be harvested (i.e. a bimodal rainfall pattern), they also model bean harvest (Figure 5.10). Maize and bean farm gate prices were from farmer interviews (see section 5.2.1.1; maize: median = 0.2 USD kg^{-1}, range = 0.05 to 0.7 USD kg^{-1}, n = 58; bean: median = 0.6 USD kg^{-1}, range = 0.4 to 0.7 USD kg^{-1}, n = 12). Of 135 surveyed fields, 54% were planted, at least partly, with bought seed at an application rate of 20 kg ha^{-1} for maize and 66 kg ha^{-1} for bean (maize: range = 2 to 100 kg ha^{-1}, n = 72; bean: range = 3.5 to 328 kg ha^{-1}, n = 26). These seed application rates are supported by other studies (Eberhart 1969; PNB n.d.) and were multiplied by the crop price (see above) to derive the input cost for seed. Fertilizer application was assumed to be 5 kg ha^{-1}, as this was the amount upon which yield was modelled (Thornton et al. 2009) and the cost was set at \$0.58 kg^{-1} (CIMMYT n.d.). Finally, labour was assumed to be 55 man days ha^{-1} y^{-1} for maize farming and 49 man days ha^{-1} y^{-1} for bean farming (Ngambeki 1985). This was multiplied by the median unskilled daily labour wage available to surveyed villagers (median = 1.7 USD day^{-1}; range = 0.4 to 4 USD day^{-1}; n = 16), which was also similar to the minimum wage, in Tanzania, of 1.5 USD day^{-1} reported by the US Department of State (2008).

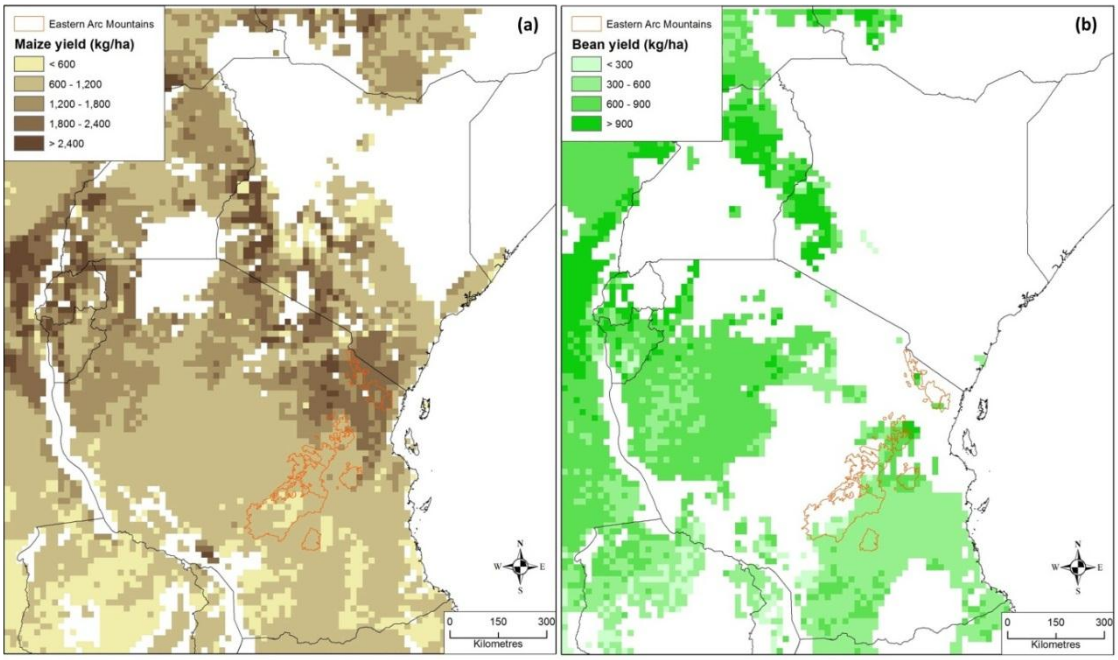

Figure 7.3. Maize (a) and bean (b) yields are mapped at a resolution of ten Arc minutes (approximately 18.5 km by 18.5 km) across East Africa under current climatic conditions and for typical smallholder farmer practices (Figure adapted from Thornton et al. (2009)).

The net returns to agriculture from production of maize and bean were calculated separately using Equation 5.15 and Equation 5.16 respectively:

$$NR_{mit} = (Y_{mit} \times P_{mt}) - (S_{mt} + F_{mt} + L_{mt}),$$

Equation 5.15.

where NR_{mit} is the net rent from maize (*m*) production on a one hectare land parcel *i* during time *t*, Y_{mit} is the yield (kilograms) of maize per hectare, P_{mt} is the price of maize per kilogram, S_{mt} is the costs of seed per hectare, F_{mt} is the cost of fertilizer per hectare and L_{mt} is the cost of labour per hectare and

$$NR_{bit} = (Y_{bit} \times P_{bt}) - (S_{bt} + F_{bt} + L_{bt}),$$

Equation 5.16.

where NR_{bit} is the net rent from bean (*b*) production on a one hectare land parcel *i* during time *t*, Y_{bit} is the yield (kilograms) of bean per hectare, P_{bt} is the price of beans per kilogram, S_{bt} is the costs of seed per hectare, F_{bt} is the cost of fertilizer per hectare and L_{bt} is the cost of labour per hectare.

On average, land will not be farmed if net rent is less than zero. Therefore, if input costs exceeded the value of the yield for that crop, net rent is set to zero. Total annual net rent was then calculated by summation of the two spatially explicit maps to give net returns to farming for maize and, where suitable, bean.