Understanding spatial priorities for conservation and restoration in Kenya

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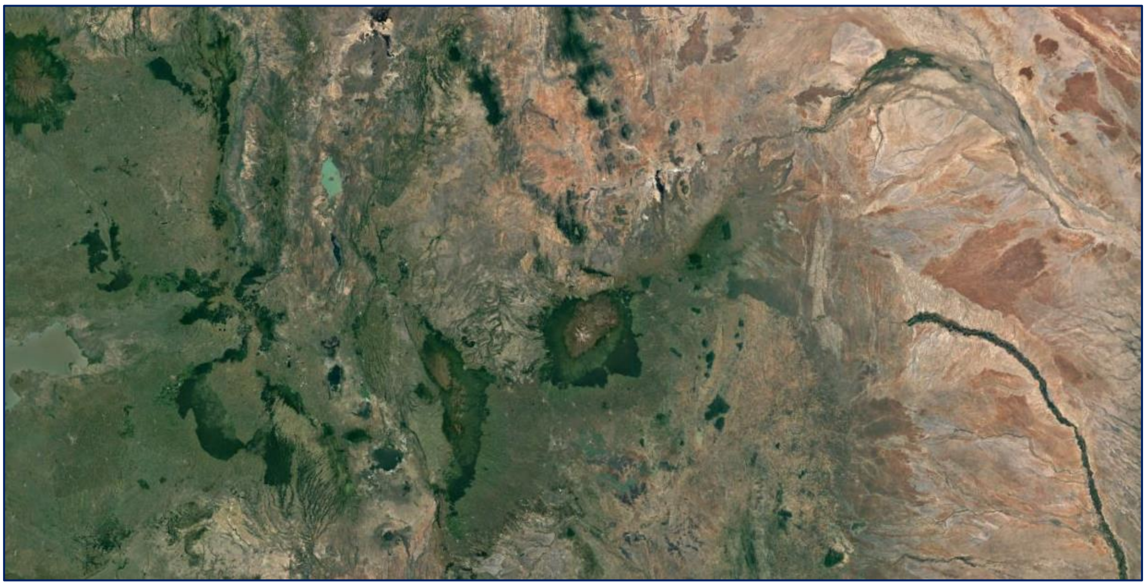

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Author declaration

G.E.M. Gibbon wrote all the chapters, with editorial suggestions made by PhD supervisors Z.G. Davies and R.J. Smith. Chapters 2-4 include collaborations with researchers outside of the supervisory team, as outlined below.

Chapter 2 originated from discussions with Z.G. Davies and R.J. Smith. The semi-structured interviews were developed by G.E.M. Gibbon with support from Z.G. Davies and R.J. Smith. The expert elicitation survey was developed by G.E.M. Gibbon with support from L.A. Evans, F.W. Ihwagi, H. Vanleeuwe, Z.G. Davies and R.J. Smith. Data collection and all analysis were led by G.E.M. Gibbon. G.E.M. Gibbon wrote the chapter with collaborative input support from Z.G. Davies and R.J. Smith.

Chapter 3 originated from discussions with Z.G. Davies, M. Dallimer, and R.J. Smith. The structured decision-making framework was developed by G.E.M. Gibbon with support from M. Dallimer and Z.G. Davies. Data collection and all analysis were led by G.E.M. Gibbon. G.E.M. Gibbon wrote the chapter with collaborative input from Z.G. Davies, M. Dallimer and R.J. Smith.

Chapter 4 originated from discussions with Z.G. Davies and R.J. Smith. G.E.M. Gibbon developed the systematic conservation planning system following discussions with L. Waringi, D. Kaelo, J.R. Allan, P. Tyrrell and R.J Smith. Data collection was led by G.E.M. Gibbon, with support from R.J. Smith. All data collection and analysis were conducted by G.E.M. Gibbon, with support from R.J. Smith. G.E.M. Gibbon wrote the chapter with collaborative input from Z.G. Davies and R.J. Smith.

Abstract

Human actions are having widespread, profound, and growing impacts on the global environment. This is converting habitats, driving biodiversity loss, and eroding ecosystem functions that support human livelihoods, health, and well-being. Protected areas remain a cornerstone of efforts to abate environmental destruction, safeguard biological diversity and maintain ecosystem functions, but they must formalise and incorporate other effective area-based conservation measures into conservation planning and monitoring frameworks. Given the rapid onset of anthropogenic climate change and the failure to halt environmental destruction, there is also a widely recognised need to restore ecosystems based on their benefits to biodiversity, ecosystems and people. This has the potential to revolutionise conservation actions, moving away from a protectionist history into a paradigm of creation and co-existence that puts local communities, industries and wider civil society at the forefront of efforts. This requires a future where multifunctional ecological networks are equitable, effective and well connected and therefore resilient to climatic and broader environmental change.

I focus this thesis on the Republic of Kenya, a country that is going through a sustained period of economic growth and committed to protecting its wealth of biodiversity and to ecological restoration. First, I provide a novel method to map functional connectivity within Kenya's Central Highlands using expert opinion validated with empirical data. I use this to identify areas of restricted movement and investigate how they would change under stakeholder-defined future land-use options, finding where restoration would be most beneficial. Second, I investigate stakeholder preference for these options using structured decision-making and a multi criteria decision analysis, finding broad support for habitat conservation and restoration that changed little as I built more consensus into the process. Third, I use a novel application based on the principles of systematic conservation planning to integrate spatial priorities for biodiversity conservation and restoration, meeting representation targets across terrestrial Kenya. I measure the contribution of Kenya's different protected area types and use scenarios to show how meeting targets by restoring large patches of habitat, instead of conserving small fragments, produces more ecologically viable but more expensive ecological networks.

This thesis helps provide an understanding of the benefits of integrating national-level priorities for conservation and restoration. However, governments must develop context-specific approaches to target interventions that put local communities at the forefront of efforts, promoting multifunctional landscapes of human-wildlife co-existence and enabling environmental stewardship.

Keywords: biodiversity conservation, ecosystem restoration, ecosystem services, landscape connectivity, local communities, scenario planning, structured decision-making, systematic conservation planning.

Covid-19 Impact Statement

G.E.M. Gibbon kindly requests his examiners to note how the COVID-19 pandemic affected the completion of this thesis.

Kenya shut its borders four days before he was due to travel in March 2020, meaning a long-planned and already piloted workshop representing the culmination of data collection for **Chapter 3** had to be adapted to occur whilst working remotely. After redesign and over 60 hours of video calls, he completed data collection six months later than planned. He was also to use this trip to collect field data and engage policymakers in designing the systematic conservation planning process undertaken in **Chapter 4**. Given the inability to do so, he instead attempted to best engage with relevant parties via video call and email and instead relied on online datasets and those shared with me by other academics and researchers. The January 2021 lockdown then again delayed him from running the final data analysis for this chapter. He is thankful to the University for granting him an extension and the support of his supervisors and other contributors that allowed him to adapt to this challenge.

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Chapter 1 Introduction

1.1 Biodiversity in a changing world

The Earth is experiencing widespread, profound, and increasing impacts on the global environment (Ellis et al., 2021; Ruckelshaus et al., 2020; Steffen et al., 2015; Williams et al., 2020), particularly driven by the consumption patterns of the increasingly affluent (Wiedmann et al., 2020). These affluence levels are also increasing, and the resultant demand for food, energy and other materials is driving land-use change, which has led to the conversion and fragmentation of natural habitats over more than 83% of our planet's land surface (Sanderson *et al.*, 2002; Venter *et al.*, 2016; Song *et al.*, 2018). The result is smaller and more isolated habitat patches, which are inherently more likely to experience declines in biodiversity and ecosystem function, something observed in both tropical forests (Haddad et al., 2015) and dry-land environments (Durant et al., 2015).

This anthropogenic change is driving high levels of biodiversity loss, with species extinctions occurring at over 1,000 times the background rate (Pimm et al., 1995). Over 37,400 species, 28% of all assessed species, are threatened with extinction (IUCN, 2020) and populations of wild mammals, birds, amphibians, reptiles, and fishes have shown an average decline of 68% between 1970 and 2016 (WWF/ZSL, 2020). This biodiversity loss risks compromising the integrity of ecosystems, which may have cascading effects across taxa (Chapin et al., 2000). It also risks the interlinked biological processes that support human societies and are known as ecosystem services (Millennium Ecosystem Assessment, 2005; Mace *et al.*, 2012). Given that human consumption patterns are already beyond planetary boundaries of replenishment (Lin et al., 2018) and destabilising the global climate (Steffen et al., 2015), widescale and urgent actions are required. These conservation activities need to protect both biodiversity and ecosystem services. However, despite growing recognition of the crises, conservation efforts to protect existing natural systems from environmental degradation and biodiversity loss have not changed the trajectory of declines (Ellis et al., 2021; Steffen et al., 2015; Williams et al., 2020). Concern has catalysed a new focus on ecosystem restoration (UNEP, 2021), which must integrate with conservation activities (Chazdon & Brancalion, 2019; Milner-Gulland et al., 2021; Wiens & Hobbs, 2015).

On the ground, this is fundamentally about rights recognition and self-determination, enabling and empowering environmental stewardship from those living within conservation or restoration landscapes (Armitage et al., 2020; Tauli-Corpuz et al., 2020). However, this also needs to be supported by accountable government institutions (Ribot *et al.*, 2006; Smith *et al.*, 2009). At a national level, the need for global action in conservation and ecosystem restoration is recognised and formalised through the Convention on Biological Diversity (CBD). For the last decade these activities were informed by the Aichi Targets (Convention on Biological Diversity, 2010) but these expired in 2020 and are due to be replaced by the 'post-2020' targets. The zero draft of these has a 2030 mission "to take urgent action across society to put biodiversity on a path to recovery for the benefit of planet and people" (Convention on Biological Diversity, 2020). Implementation will be based on meeting a series of targets, and while the exact details of the targets are still being agreed, it is certain their implementation will require rapid and profound actions. This will involve the scaling up of conservation and restoration activities, as well as the overhaul of industries and the restructuring of economies, with unprecedented ramifications for people. This has particularly large implications for countries in the tropics. These are the most biodiverse regions (Myers et al., 2000) and also where biodiversity loss is most pronounced (WWF/ZSL, 2020). These countries are therefore where some of the most urgent conservation attention and ecosystem restoration is required.

1.2 Protected areas and OECMs

Protected areas (PAs) remain a central component of conservation efforts (Bhola et al., 2021; Bingham et al., 2019; Maxwell et al., 2020). Recent estimates identify nearly a quarter of a million legally designated terrestrial PAs, covering 15.7% of the terrestrial realm (IUCN and UNEP-WCMC, 2021). Aichi Target 11 committed countries to expand area-based conservation to 17% of the terrestrial surface by 2020 'through effectively and equitably managed, ecologically representative, and well-connected systems of protected areas'. Despite increases of 2.3% on land since 2010, there is still a shortfall of 1.5% or over 7.5 million $km^{2,}$ and concerningly, the rate of increase was half that of the prior five years (Venter et al., 2018).

Protected areas represent a sizable commitment in progress towards effective conservation. They are diverse in their function and form, each set within their own socio-political and geographic context. They commonly help achieve both biodiversity and ecosystem service conservation goals but may also fulfil various other ecological, social or economic aims (Scharlemann et al., 2012). However, despite global acceptance of the worth of PAs (Butchart *et al.*, 2015), they can be contentious (Duffy et al., 2019) as further expansions will come at additional costs to many, mainly rural, poor people. Therefore their role in achieving the post-2020 targets of the CBD requires existing PAs and further conservation actions to be valued by neighbouring communities and integrated with wider conservation and land-use planning (Maxwell et al., 2020; Watson et al., 2014; Wilson et al., 2010).

As a first step, we must recognise that PA effectiveness in achieving conservation goals is mixed. For instance, while strictly protected PAs can be more likely to exhibit stable or improving large herbivore species populations (Stoner et al., 2007), there is inconclusive evidence of conservation success across broader vertebrate taxa and protection levels (Geldmann et al., 2013). They are effective at conserving forested habitats (Geldmann et al., 2013), but one study found East Africa to exhibit higher rates of forest loss within some PAs than within the surrounding unprotected areas (Bowker et al., 2016), reinforced by another study that found PAs with higher levels of protection were more likely to maintain or increase their forest area (Pfeifer et al., 2012). This research further reinforces how the specific set of challenges and human pressures that affect individual PAs and their effectiveness are highly context-specific.

PAs in lower- and middle-income countries in the tropics are at particular risk of being ineffective. This is because of financial issues such as the widespread underfunding or lack of adequate resources for management (Bottrill et al., 2009), which can compromise overarching governance processes and lead to poor PA performance (Bruner et al., 2001). One recent estimate calculated that USD 24 billion was annually devoted to PAs (Claes et al., 2020), a small proportion of what is needed. For example, one study estimated that PAs in the tropics suffer from a USD 1-1.7 billion shortfall in management spending (Bruner et al., 2004), although these costs are far less in value than the benefits they provide to local, national and global peoples (Watson et al., 2014). Another surveyed 23% of the global PA estate and found under a quarter of PAs reported that they had adequate funding for resources and staff (Coad et al., 2019). This is reinforced by an analysis of the costs and benefits of effective global conservation, of which PAs are a fundamental part, which reported a 100:1 benefit to cost ratio of effective nature conservation (Balmford et al., 2002).

Lands that fall outside of the classic definition of a PA are also essential to achieving conservation and already contribute a great deal to meeting conservation targets. There has therefore been growing recognition of the role of other effective area-based conservation measures (OECMs) (Dudley et al., 2018; Jonas et al., 2017). These are defined as "a geographically defined area other than a protected area, which is governed and managed in ways that achieve positive and sustained long-term outcomes for the in situ conservation of biodiversity, with associated ecosystem functions and services and where applicable, cultural, spiritual, socioeconomic, and other locally relevant values" and can cover a variety of land uses (Donald et al., 2019).

1.2.1 Protected areas and connectivity

One of the components of Aichi Target 11 where failure was most pronounced was related to connectivity (Saura et al., 2017), with recent estimates showing structural connectivity through intact habitat between only 10% of the world's terrestrial protected areas (Saura et al., 2019; Ward et al., 2020). Concerningly PAs containing tropical forests are particularly isolated (DeFries et al., 2005; Grantham et al., 2020) because protected area connectivity is essential for supporting healthy wildlife populations (Opdam, 1991; Woodley et al., 2019), ensuring ecosystem function (Haddad et al., 2015) and resilience to environmental change (Saunders et al., 1991; Watson et al., 2012). Research on this topic is commonly named landscape connectivity and is defined as "the degree to which the landscape facilitates or impedes movement among resource patches" (Taylor et al., 1993). This is defined by the physical features of the landscape that provide structural connectivity, which can be used for large scale analyses (e.g. Ward *et al* 2020). However, this does not consider how species interact with these features to facilitate successful movements, the study of which is referred to as functional connectivity.

Functional connectivity incorporates data on species-specific traits of landscape permeability, reflecting both landscape structure and a species' response to it (Bélisle, 2005). Least-cost modelling and circuit theory are two widely applied and complementary means to map functional connectivity (Zeller *et al.,* 2012). The first represents the most cost-effective route between two locations. Circuit theory instead treats movement like an electrical current moving omnidirectionally across a landscape. This identifies all potential routes, showing where constrained movement occurs through 'pinch points' (McRae *et al.*, 2016). Both rely on two inputs: resistance surfaces and core areas. Resistance surfaces are spatial raster layers where the permeability or physiological cost of species movement across is represented by pixels (Spear et al., 2010). Resistance surfaces are therefore commonly made by giving values to environmental or structural characteristics such as terrain, landcovers, or other features which can facilitate or inhibit movement (Pullinger & Johnson, 2010). Core areas, meanwhile, are the patches of habitat between which connectivity is modelled.

Linkage Mapper is a tool for undertaking such connectivity analyses and can be used to model species movements or gene flow across fragmented landscapes (McRae et al., 2016; McRae & Kavanagh, 2011). It has been used to create regional connectivity maps between islands (e.g. Harradine *et al.*, 2015), for conservation of priority landscapes (e.g. Bowman and Cordes, 2015; Bleyhl *et al.*, 2017; Pelletier *et al.*, 2017), for the requirements of key species (e.g. Joshi *et al.*, 2013; Reed *et al.*, 2016; Gangadharan, Vaidyanathan and St. Clair, 2017) and the requirements of guilds of species (e.g. Lechner *et al.*, 2016), as well as for identifying important or lost linkages (e.g. Dutta *et al.*, 2016). These analyses can be based on expert derived data (e.g. Zeller et al., 2012), but more robust approaches incorporate empirically observed animal movement data (e.g. Kanagaraj *et al.*, 2013).

PAs are diverse in form and many allow multiple land-uses (Scharlemann et al., 2012). Maintaining connectivity within PAs is therefore also important. Many conservation landscapes contain PA networks that consist of large core habitat blocks, which need to be connected via linkages or movement corridors to a series of smaller fragments (Laurance et al., 2001). Therefore the maintenance and restoration of habitat linkages and wildlife corridors are increasingly important (Benz et al., 2016; Caro et al., 2009). Efforts need to consider where restoring linkages is required and how this compares to the costs of conserving existing linkages. Linkage projects have also been shown to be successful at increasing wildlife dispersal within multiple taxa (Epps et al., 2011; Gilbert-Norton et al., 2010), but they need to be planned carefully as they have also been shown to increase human wildlife conflicts (e.g. Acharya et al., 2017), have the potential to deny indigenous communities access (Brockington & Wilkie, 2015) and mask hidden political dimensions (Evans & Adams, 2016). Achievement towards the Post-2020 targets must therefore consider social as well as ecological aspects of connecting PAs and OECMs as parts of larger conservation landscape.

1.3 Protected area governance and decision-making

In settler- and post-colonial nations, such as Australia, the USA, Nepal, India and those of sub-Saharan Africa, PAs have a particularly contentious past (Adams & McShane, 1996; Adams & Mulligan, 2003; Brockington, 2002). This is due in large part to a 19th and 20th century tradition of top-down management that excluded indigenous communities' and local people's rights, access, or say in management. Injustices include forced evictions (Brockington & Igoe, 2006) and other human rights abuses (Brockington et al., 2006). In a break from this mentality of 'fortress conservation' (Brockington, 2002), the World Parks Congress in 1982 outlined the birth of a global shift towards PA governance that includes local communities say in management. This has grown throughout the last few decades and today few people argue against the need for an inclusive, participatory approach to conservation. Yet, progress has in some places been slow and injustices often resurface with violent confrontations between protected area staff and local people (Duffy et al., 2019), leading to conservation efforts being resented. So, despite the move towards bottom up, inclusive management PAs are still criticised for 'distributing fortune and misfortune unevenly' (Brockington & Wilkie, 2015). They also create opportunity costs – 'the foregone benefits from other land uses' (Adams et al., 2010) – which are commonly born by local people and may exacerbate poverty levels (Brockington et al., 2006). However, multi-country analyses have shown that PAs can contribute to achieving poverty alleviation and improved human wellbeing (Ferraro et al., 2011; Naidoo et al., 2019)

PAs also tend to be located in remote regions, at higher elevation, on steeper terrain and on agriculturally unproductive land (Arturo Sánchez-Azofeifa et al., 2002; Gaston et al., 2008; Joppa & Pfaff, 2009; Moilanen et al., 2009), and this pattern of creating new PAs on land with lower opportunity costs continues (Venter et al., 2018; Watson et al., 2014). This means that the location of PAs is commonly in remote areas, often bordered by poor and marginalised farming communities. Yet, while conservation cannot solve poverty, its reduction can be linked to systems where sustainably managed PAs provide resources and other benefits to improve local livelihoods and reduce poverty levels (Naughton-Treves et al., 2005), all without compromising on biodiversity outcomes (Pfaff et al., 2014; Porter-Bolland et al., 2012).

For PAs to achieve these benefits, conservation decision-making and broader governance must engage with a range of stakeholders and account for the different ways that they value biodiversity (Pascual et al., 2021; Wheeler & Root-Bernstein, 2020). It is therefore unsurprising that conservation plans are recognised as both more legitimate and politically acceptable when set in a local context (Rodriguez et al., 2007; Smith et al., 2009; Brooks, Waylen and Mulder, 2012). Modern PA and wider governance structures also have a focus on equitability and addressing past injustices and local challenges, as this is seen as critical for ensuring protected area effectiveness (Zafra-Calvo & Geldmann, 2020). For example, one study found that community participation was the only factor that significantly positively correlated with local people complying with PA regulations (Andrade & Rhodes, 2012), and community managed forests have also been shown to be equally or even more effective than PAs at reducing deforestation (Fa et al., 2020; Hayes, 2006; Porter-Bolland et al., 2012). Management must also qualify, quantify and effectively communicate the socioeconomic and ecological effects of management actions (Adams & Mulligan, 2003; Leader-Williams et al., 2011). This means that successful conservation planning in landscapes containing PAs and OECMs often requires complex organisational structures, which are best informed by data to make management decisions and navigate the trade-off inherent within them.

1.3.1 Structured decision-making to engage and enable stakeholders

Making sensible and informed decisions in PA and OECM governance depends on understanding which options best achieve multiple and often competing goals, whilst trying to avoid or otherwise mitigate negative impacts on ecosystems and human communities. This relies on approaches that are collectively known as strategic foresight (Cook et al., 2014; Sarpong & Maclean, 2016), a valuable tool for environmental management that envision future opportunities and threats, providing similar benefits to the more narrowly defined methodology of horizon scanning (Sutherland & Woodroof, 2009).

One common practice is the creation of scenarios to explore the impacts of possible management options. Designed appropriately, exploratory scenarios can capture distinct yet divergent possible future pathways and help investigate their potential impacts (Mckenzie et al., 2012a; Peterson et al., 2003). These scenarios can then be evaluated through structured decision-making. Structured decision-making describes analytical frameworks that allow for careful and organised evaluation of choices that aim to reach clearly defined objectives (Clemen & Reilly, 2001). These processes provide an audit trail of how decisions are reached and allow the inclusion of multiple stakeholder perspectives. Structured decision-making can therefore be applied in a participatory way. This benefits decision-making by providing information on the specific aspects of problems that most concern different stakeholder groups. They are also explicit about the value judgements that underlie choices and so identify the trade-offs and synergies between the different goals of environmental management (Adams et al., 2016). When designed appropriately, being clear about the objectives of a decision and visualising the options as exploratory scenarios can naturally help foster local support for conservation and restoration plans designed to benefit diverse stakeholders.

A multi criteria decision analysis (MCDA) is a structured decision-making method that synthesises knowledge to evaluate preference for different alternative options (GoUK, 2009; Linkov & Moberg, 2012; Loro et al., 2015). These options can be exploratory scenarios. MCDAs can incorporate quantitative and qualitative measures of the performance of explicitly defined criteria and then incorporate multiple values to systematically evaluate and rank different alternative scenarios. They also allow the inclusion of multiple stakeholder perspectives that can inform all levels of the decision-making process. MCDAs have value for decision-making, as shown by a wide base of use in medicine (Adunlin et al., 2015) and forest management (Uhde et al., 2015). Given the overlaps and other similarities between these disciplines and conservation, it is no surprise that their use in broader environmental decision-making is growing (Cegan et al., 2017; Esmail & Geneletti, 2018). For example, a recent review found 86 peer reviewed examples of MCDAs being used for nature conservation

(Esmail and Geneletti, 2018). They stressed the successful application of an MCDA must include or otherwise inform stakeholders at all the following steps:

Stage 1: Decision context definition and problem structuring

(i) Criterion definition to identify the distinct objectives of the decision-making process.

(ii) Alternative option identification, which will be assessed for the above criteria.

Stage 2: Analysis

(iii) Option assessment to score its performance for each criterion.

(iv) Criterion weighting to signify their importance within the decision context.

(v) Aggregation of criterion performance scores and importance weightings to quantify preference/s for each alternative option.

(vi) Sensitivity analyses to ensure robustness of the decision model and explore relationships between the criterion performance scores and importance weightings and the outputs that define the decision outcome.

Stage 3: Decision

(vii) Ranking the alternative options based on overall preference.

(viii) Evaluating how different preferences affect the decision outcome.

These steps allow MCDAs to integrate interdisciplinary approaches and mixed methods, capturing quantitative and qualitative data such as monetary values, as well as non-monetary, ecological, and shared values (De Groot et al., 2010; Kenter et al., 2014). MCDAs can therefore help evaluate policy measures or determine the most equitable management actions (e.g. Langemeyer et al., 2016; Topping, Dalby and Valdez, 2019). They can facilitate participatory approaches to rank land-use strategies (Fontana et al., 2013) and have been used to identify the different costs and ecosystem service benefits from alternative land-use strategies (Favretto et al., 2016). Therefore, the chief value of a MCDA is the ability to build consensus whilst still facilitating the investigation of the distinct underlying perspectives. This allows them to be used to analyse complex problems in socio-politically heterogeneous landscapes in a carefully deliberated fashion.

1.4 Ecological restoration for conservation

The massive human impacts on the global environment (Ellis et al., 2021; Ruckelshaus et al., 2020; Steffen et al., 2015; Williams et al., 2020) have converted over 51 million km^2 to intensive human use and has left over 20 million km^2 of land degraded (Minnemeyer et al., 2011). Given the scale of these impacts, putting humanity on a pathway to recovery by 2030 and achieving the CBD's goal of living in harmony with nature by 2050 requires large-scale ecosystem restoration. This is recognised by the United Nations Environmental Program who label this the 'Decade on Ecosystem Restoration'. Restoration is also a massive opportunity to complement conservation activities and must be integrated and implemented as part of larger frameworks (Chazdon & Brancalion, 2019; Gardner et al., 2020).

Structured decision-making is, therefore, a way to identify shared goals and decide how best to achieve them. Integrating conservation with restoration is fundamentally about highlighting the shared goals both activities are designed to achieve (Wiens & Hobbs, 2015) and engaging those living within priority landscapes to achieve them (Chazdon & Brancalion, 2019; Erbaugh et al., 2020; Erbaugh & Oldekop, 2018; Pritchard & Brockington, 2019). In general, community managed lands are less likely to have been converted to intensive anthropogenic uses such as agriculture (Ellis et al., 2021). This makes the engagement of indigenous peoples, local communities and other actors vital to building shared values that allow individuals to work together in a group to effectively achieve a common purpose through appropriate governance (Chazdon & Brancalion, 2019; Mansourian et al., 2020; McMahen & Bommel, 2020; Pretty & Smith, 2004; Smith et al., 2009). However, community engagement is not a panacea (Ostrom, 2007; Brooks, Waylen and Mulder, 2012) and ground-up decision-making needs to be supported and monitored by well-resourced government institutions that are also accountable (Ribot et al., 2006). This is important to ensure equitable access and capacity building that avoids elite capture and other risks that can undermine long term success (Brooks, Waylen and Mulder, 2012; He and Lang, 2015).

Sound governance promoting local stewardship and the scaling up of investment in restoration is essential. Progress towards meeting Aichi target 15, which detailed the restoration of 15% of degraded land by 2020 (Convention on Biological Diversity, 2010), has been slow, and the urgency increases with ongoing environmental degradation (Ruckelshaus et al., 2020; Steffen et al., 2015; Williams et al., 2020). However, the global drive for 'natural climate solutions' (Griscom et al., 2017), although not enough to avert the climate crisis (Anderson et al., 2019), presents a clear opportunity for conservation scientists to put biodiversity conservation and local communities at the forefront of restoration efforts (Bluwstein et al., 2021). This demands equitable and effective planning of what to do and also where to do it. PAs will be essential components of these plans, as will OECMs and other indigenous peoples' and local communities' lands (Garnett et al., 2018; Maxwell et al., 2020; O'Bryan et al., 2020).

1.5 Systematic conservation planning

Systematic conservation planning is an approach for informing and guiding conservation and restoration planning by representing priorities spatially (Margules & Pressey, 2000). It centres on a transparent, iterative and defensible decision-making process, which can inform how, when and where to focus management efforts and assign limited resources (Kukkala & Moilanen, 2013). This provides insights into how different land-use decisions will affect different stakeholders and conservation aims. Much of this value comes from systematic conservation planning's ability to merge elements of strategic planning with spatial prioritisation. This generally involves defining the planning region, dividing it up into planning units, identifying and mapping a list of conservation features (species, ecosystem types, ecological processes, ecosystem services, etc.), setting quantitative conservation objectives and then using software to run a spatial conservation prioritisation to identify the best set of planning units for meeting those objectives.

Spatial conservation prioritisation software commonly belongs to one of two families – Marxan and its relatives (Ball, Possingham and Watts, 2009) and Zonation (Di Minin et al., 2011). Marxan is based on the 'minimum set' prioritisation problem, which involves setting targets for each conservation feature and identifying the best sets of planning units for meeting those targets (Moilanen et al., 2009). Whereas Zonation answers the 'max coverage' problem, which involves specifying the proportion of the planning region that should be selected and using the software to identify which planning units should be included in this selection to maximise the conservation gain. When comparing Marxan and Zonation outputs, one study found that the former commonly produced more efficient results, whereas the latter generally produced results with higher connectivity (Delavenne et al., 2012), though both identified similar arrangements of priority areas.

While hundreds of articles have been published on using systematic conservation planning to design PA systems and other ecological networks (McIntosh et al., 2016; Sinclair et al., 2018), there is also a growing literature on how it can be used for restoration planning. This includes work on the allocation of forest cover restoration using the minimum viable population size of two forest dependent mammal species and biogeographic regions as prioritisation features (Crouzeilles et al., 2015), as well as a prioritisation of restoration within a fragmented agricultural landscape using habitats, soil types and distance to important bird species (Jellinek, 2017). Marxan was also used in Japan for a national analysis using the range loss of threatened birds and ecosystem types and surrogates for agricultural opportunity cost (Yoshioka et al., 2014) and in the Caribbean using ecosystem service data as prioritisation features and the level of degradation as a cost (Adame et al., 2015). One application included combining both conservation and restoration activities into one analysis using Marxan with Zones, an extension of Marxan which allows for the allocation of areas to different management zones (Watts et al., 2009), which allocated areas for conservation using standard practices and for restoration using degraded habitats as prioritisation features and an estimation of monetary restoration costs (Barbosa et al., 2019).

1.6 Conservation and restoration in Kenya

East Africa has the highest endemism in Africa, with endemics making up 55% of mammals, 63% of birds, 49% of reptiles and 40% of amphibians (UNEP, 2000). It is also in the process of transitioning through economic development. Given that conservation and restoration planning mostly occurs at the local and national level, in this thesis I use Kenya as a national case study because it is an emergent economy and a biodiversity-rich nation recognised as a high priority for conservation (Jenkins et al., 2013; Mair et al., 2021). As a microcosm of Africa, it contains vast savannah systems, as well as deserts, and lowland and montane tropical forests. Within Kenya, I also use its Central Highlands as a sub-national case study to explore the issues related to planning for conservation and restoration at the local level.

Kenya has a long history of conservation, which was initially based on state-mandated PAs that were created by gazetting traditionally used lands, often involving evictions and other forms of exclusion and human rights violations (Brockington, 2002; Brockington et al., 2006). More recently, there has been a focus on community and privately-owned rangelands, which has led to Kenya undergoing policy shifts from the 1970s through to present (Western et al., 2015). Today area-based conservation in Kenya is based on three components of state-run PAs: i) National Parks (n = 23), managed nationally by the Kenya Wildlife Service, that cover

4.98% of its land-area; ii) National Reserves and other state PAs (n = 33), managed by county governments alongside county offices for the Kenya Wildlife Service, that cover 3.2% of the country and iii) Forest Reserves (n = 327) run by the Kenya Forest Service, these cover 2.9% of the country. These state-run PAs cover 11.1% of Kenya's land surface. Additionally, there are 'Wildlife Conservancies' (n = 108), which are private or community owned protected areas managed by private landowners or a community for wildlife conservation and other compatible land-uses that improve livelihoods (Government of Kenya, 2018b). They cover an additional 8.1% and are run by community, private or not-for-profit entities with support from the Kenya Wildlife Service. There are also larger areas of rangelands managed by indigenous peoples and local communities that also contribute to conservation but have no official conservation status (O'Bryan et al., 2020; Tyrrell et al., 2019).

Meanwhile Kenya has undergone rapid land-use change as economic development, human population growth and the needs of the increasingly affluent have driven urbanisation and agricultural expansion. Meeting these needs has led to the conversion and fragmentation of habitats, which is compromising ecological processes (UNEP, 2012) and is reflected in Kenya being recognised as a global priority for habitat restoration (Strassburg et al., 2020). This is why Kenya has committed to restoring 51,000 km^2 by 2030 through the African Forest Landscape Restoration Initiative (AFR100) as part of the Bonn Challenge as detailed in the Ministry of Environment and Natural Resources national assessment of forest and landscape restoration (Government of Kenya, 2016). This document aligned with Kenya's legal commitment to increase tree cover to 10%, which is also formalised in Kenya's Vision 2030 developmental plan (Government of Kenya, 2007). It lists interventions including afforestation and reforestation of natural forests, rehabilitation of degraded natural forests, promotion of agroforestry and silvopasture, rangeland restoration and the establishment of tree buffers along roads, railways and rivers. Kenya also underwent environmental planning and wider governance devolution in 2010, with much authority for land-use planning now under the authority of the 47 county governments, with county level plans feeding into regional and national level spatial planning (Government of Kenya, 2014). In Kenya ecosystem restoration efforts are largely decentralised (Smucker et al., 2020; UNEP, 2021), but efforts to coordinate activities exist in some counties and a national restoration strategy is being developed.

Kenya's Sixth National Report to the Convention on Biological Diversity recognises the progress being made towards its national goals and their relation to achieving the Aichi targets, Sustainable Development Goals and other such commitments (Government of Kenya, 2020). The document highlights how policymakers are mainstreaming biodiversity and ecosystem service conservation and restoration within the Kenyan constitution and other policy documents (e.g. Government of Kenya, 2007; 2014; 2016; 2017; 2018b). It makes a key point of the ecosystem service value of Kenya's montane forests following a report United Nations Environment Programme in 2012 (UNEP, 2012) and the formation of the Kenya Water Towers Agency in 2012 to coordinate and oversee the protection, rehabilitation, conservation and sustainable management of water towers (montane forests) in Kenya, where some of the most rapid human population growth is occurring (Odawa & Seo, 2019). The document also reiterates the need for national biodiversity and restoration policies that enable equitable and effective spatial and strategic planning of these activities at local, county and national scales. Given many previous efforts to prioritise areas for conservation have relied on solely bird data (e.g. Muriuki et al., 1997; Bennun & Njoroge, 2000), a handful of large mammal species (e.g. Didier et al., 2010) or for REDD+ schemes (e.g. Maukonen et al., 2016) there is a need for a more comprehensive analysis of priority areas for biodiversity and ecosystem services. This should also investigate the trade-offs when integrating restoration with conservation planning.

1.7 Kenya's Central Highlands

1.7.1 History

Kenya's Central Highlands contain a network of state, private and community run PAs linked by a matrix of mixed land-use strategies. The highlands stretch out north, west, and south from Mount Kenya and eventually drop into the Great Rift Valley. Much of the Central Highlands is the Laikipia Plateau, which are well-watered and fertile ancient lava flows. Vegetation cover is defined largely by altitude (Bussmann & Beck, 1995; Konecky et al., 2014), but also driven by the region's climatic history of variable rainfall and temperatures (Nicholson, 1996; Schmocker et al., 2016), as signalled by rapid advance and withdrawal of Mount Kenya's glaciers (Hastenrath, 2010).

Mount Kenya's first post-colonisation PA was gazetted in 1932 in the form of the Mount Kenya Forest Reserve. This was designed to protect and regulate the extraction of the commercial timber interests of the British colonial government (Emerton, 1999). Excisions occurred at the lower boundary of the PAs to government projects as well as local communities and continued until 2001 (Vanleeuwe, 2004). These excisions mainly consisted of exotic plantation blocks, which were settled under the 'shamba system', which allows peasant farmers to cultivate crops for a period of 3-5 years whilst the plantations were established (Emerton, 1999; Vanleeuwe, 2004). This method is also known as non-residential cultivation or 'taungya' and is a system adopted by the British colonial government originating in what is today Myanmar. Although this consisted of only 2.8% of the PA, much of this land was in mid-altitude linkages that are elephant movement pathways (Graham et al., 2009; Kamweya et al., 2012a; Ihwagi et al., 2015). This has played a key role in these areas becoming the main human wildlife conflict hotspots within the region (Di Minin et al., 2021; Kamweya et al., 2012b; Vanleeuwe & Gitau, 2020).

Today these PAs have a footprint of approximately 280,000 ha and are managed by government agencies at both a national and county level. These are chiefly the Kenya Wildlife Service (KWS) and the Kenya Forest Service (KFS). The Kenya Constitution (2010) has also recognised the right of local communities to have a say in management. This has led to the establishment of Community Forest Associations (CFA), which partner with the KFS and authorise members to access the PAs for grazing of livestock, firewood, timber, and honey collection, as well as to cultivate crops within the PAs during plantation establishment as under the auspices of Plantation Establishment and Livelihood Improvement System (PELIS), which is the name by which the former shamba system or non-residential cultivation is now referred to (Emerton, 1999). In early 2018 illegal clearing of a reforestation site caused a public outcry and subsequent action led to a national moratorium on the harvest of plantations (Government of Kenya, 2018a).

Despite this moratorium, the PAs still allow multiple land-use strategies – these include mixed-use areas, where sustainable grazing and non-timber forest product extraction are permitted; plantation forestry areas where PELIS is allowed and reforestation sites using the Tree Establishment and Livelihood Improvement System, which allows local communities access to farm during the establishment of indigenous trees for reforestation (Vanleeuwe & Gitau, 2020). As with many montane forest PAs, the boundary is surrounded a ‘hard’ edge of farmland with over half a million people living within 5 km of the Mount Kenya National Reserve boundary (Vanleeuwe, 2004) and over five million in the wider seven counties (Rose et al., 2019). This larger landscape consists of a matrix of differing land-use strategies, which include private and community wildlife conservancies, small and large-scale agriculture and horticulture, and community-owned pastoral rangelands (Kiteme et al., 2008). Within this matrix, fencing is becoming commonplace (Crego et al., 2020; Evans & Adams, 2016), and much of the peripheral boundary is now fenced. Habitat linkage projects within the region include one established wildlife corridor, which securely links the habitats of the MKNR to the expansive northern rangelands through a private conservancy (Nyaligu & Weeks, 2013) and other such projects being discussed (e.g. Kamweya et al. 2012b).

1.7.2 Biodiversity and connectivity

Kenya's Central Highlands host a wealth of biodiversity where the forests and moorlands of the core PAs, for example, house over 880 plant species (Bussmann & Beck, 1995). They are also home to 81 endemic species – including 12 plant species, two reptiles, and several high altitude molluscs and arachnids (Kamweya et al., 2012b). The site also contains a nationally important population of around 2,500 IUCN listed Endangered African savannah elephants – *Loxodonta africana* (Vanleeuwe, 2010; Vanleeuwe & Gitau, 2020), which are threatened both locally and globally with poaching for ivory and retaliatory killings in response to crop-raiding and property damage events (Graham et al., 2010; Wittemyer et al., 2014).

Mount Kenya's elephants are a priority species for conservation. They are surveyed using a dung-count methodology incorporating distance sampling derived data (Vanleeuwe & Gitau, 2020), with collar data (Ihwagi et al., 2019) and aerial surveys (Crego et al., 2020). Anthropogenic activities impact their movements (Graham et al., 2009), but their distribution is also determined by environmental variables such as landscape heterogeneity (Gaucherel et al., 2010); geographic features such as steep valleys and mountainous areas (Wall et al., 2006); and vegetation growth and senescing (Bohrer et al., 2014), as well as by their own traditional behaviours (Fishlock et al., 2016). The design of one established wildlife corridor was in part based on elephant occurrence and conflict data, but its final alignment was more influenced by landowner willingness (Nyaligu & Weeks, 2013). However, elephants have since adapted their movements and use this as a transit corridor and an extension of habitat (Green et al., 2018). With two more wildlife corridors proposed (Kamweya et al., 2012a), and key linkage areas of Mount Kenya's PAs under cultivation (Vanleeuwe et al., 2003), elephant movements, therefore, provide a valuable surrogate for landscape connectivity (Epps et al., 2011).

1.7.3 Ecosystem services

Kenya's Central Highlands support diverse livelihoods and human wellbeing through ecosystem service provision. The only published assessment of the ecosystem service benefits within the protected areas of Mount Kenya (Emerton, 1999) calculated that the conversion of the forests to agriculture – the 'opportunity cost' – would annually provide USD 72 million in revenue. Yet, the estimated annual economic benefits of the forests were USD 77 million. With revenues estimated at USD 300 per household received by the 40,000 households adjacent to the forest. Based on these calculations, the opportunity costs borne by the local communities were outweighed by the benefits of conversion, but the forest's overall benefits to society outweighed their returns if converted.

The wider ecosystem service benefits from Mount Kenya extend far beyond it and they are recognised as the nation's largest 'water tower' for their water provision value (Gathaara et al., 1999; Viviroli et al., 2007). The catchments on the mountain feed into Kenya's two largest rivers, which provide water to half the nation's land surface, one third of its human population and power a series of hydroelectric dams that supply some 50% of the national grid. Emerton (1999) estimated water provision as 71% of the total economic value of the forests. Yet managing abstraction is a fundamental challenge for the region (Kiteme et al., 2008; Notter et al., 2007), even with recent amendments of the Water Act (Kanda & Taragon, 2013), river water abstraction is commonly unmetered, and there are repeated claims of weak institutional governance and a lack of financial autonomy for key government agencies (Dell'Angelo et al., 2015). The cause for concern is backed up by studies documenting unsustainable levels of abstraction, often from high up in the catchments, deep within the PAs, with some rivers barely flowing once they reach the PA boundaries (Aeschbacher *et al.*, 2005).

Plantation forestry activities are a prominent feature of the multiple-use zone of the PAs and were estimated to represent over 25% of the benefits derived from the forests. The revenue also directly benefits local people, as well as the national government and its agencies (Emerton, 1999). Current plantations cover over 21,000 ha or 13.3% of the PA and allow over 6,000 people to live within the forests, with many thousand more temporarily living within the cultivated areas (Vanleeuwe, 2004). Plantations for harvest consist entirely of exotic species monocultures of *Pinus* and *Cupressaceae* species (Emerton, 1999), which are harvested approximately every 30 years. These species in similar environments have been found to raise the cost of water procurement by 30% (Wilgen et al., 1996). Also, along with indigenous forests, they have been shown to produce lower water yields than deforested habitats (Li et al., 2007), though mixed results have been found with broader habitat classifications (Bruijnzeel, 1997). Additionally, there have been criticisms of weak governance and that timber products are undervalued, with the benefits derived only to a few commercial loggers (Emerton, 1999). The establishment of Community Forest Associations in the early 2000s aimed to remedy this by directing revenue flows from forestry and other resource use activities to neighbouring communities, though problems of elite capture and abuse exist in other Kenyan forests (Witcomb & Dorward, 2009). However, food crop cultivation activities, under PELIS, also represent an important source of food and financial security in a larger landscape prone to drought and failed crops (Ulrich et al., 2012).

Revenues from tourism within the PAs are also recognised as a chief benefit, with thousands of both domestic and international visitors travelling to the region each year to climb the mountain or visit the surrounding private, community and state-owned PAs. Yet, Emerton (1999) calculated that only 1% of the total revenue of the PAs came from tourism, with recent studies of the local Afroalpine mountain tourism industry highlighting that the economic benefits are far less than commonly assumed (Steinicke & Neuburger, 2012).

Kenya's Central Highlands, therefore, represent a valid case study to map biodiversity and ecosystem service conservation and restoration values and understand what challenges and opportunities exist for improved environmental management.

1.8 Thesis structure

Given the ambitious scaling up of conservation and restoration described by the CBD's 'Post-2020 zero draft' (Convention on Biological Diversity, 2020), I use this thesis to address relevant research gaps with Kenya's Central Highlands and then wider terrestrial Kenya. These research gaps were initially identified through discussions with government officials, members of the wildlife and forestry agencies, staff of local non-governmental organisations and other stakeholders in Kenya. This means that this thesis consists of the following additional chapters:

Chapter 2 applies mixed methods to map landscape connectivity under future land-use options for Kenya's Central Highlands in the policy-relevant year of 2030. Stakeholder interviews informed this to identify the options, with expert opinion on elephant movement validated with empirical data used as a surrogate for functional connectivity.

Chapter 3 uses a multi criteria decision analysis (MCDA) as part of a structured decision-making framework to investigate stakeholder preference towards these future land-use options. Their performance was scored using mapped scenarios and ecosystem service modelling. Stakeholders then weighted the importance of the ecosystem service benefits (the criteria) through individual and consensus building discussions. These were then combined to quantify preference and understand how it differed within and across stakeholder groups.

Chapter 4 uses systematic conservation planning to look at national priorities for integrating conservation and restoration across terrestrial Kenya and investigate the cost and other implications for people in terms of the area required, the human population within this area and how they overlap with areas of importance for carbon and water. It then looks at how these change with different limits for the proportion of intact natural vegetation that was considered appropriate for conservation attention without also restoring to a minimum patch size.

Chapter 5 provides an overview of the findings of this thesis. I discuss the contributions this thesis makes to the research field, future research ideas and make recommendations for policy and practice.

Chapter 2 Trade-offs in land-use priorities define connectivity in a heavily human-modified landscape

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2.1 Abstract

Context

Maintaining and restoring functional connectivity is vital for developing ecological networks and linking conservation areas. Planning for this depends on producing measures that incorporate data on structural linkages between intact habitats with species-specific traits determining their response to this structure.

Objectives

We aimed to identify the most efficient pathways and areas of restricted movement between natural habitat patches within Kenya's Central Highlands. We then investigated how different land-use options would affect connectivity within these linkages, using the African savannah elephant (*Loxodonta africana)* as a surrogate for broader species movements.

Methods

We used semi-structured interviews with local experts to define the study system limits and identify and map the future land-use options. We then used a four-point structured question and the Delphi Technique to elicit expert opinion on landcover resistance values for elephants moving within and between protected areas, validating these estimates with empirical location data.

Results

We identify key areas of restricted movement both within and between protected areas, showing that landscape connectivity is already highly compromised. We then show where increased agriculture and silviculture would further degrade these linkages, and where habitat restoration would improve connectivity.

Conclusions

We show that expert elicitation techniques can play an important when modelling landscape connectivity, even for well-studied species, and that habitat restoration should play a key role in maintaining and restoring linkages.

Keywords: expert elicitation, structured question, Delphi method, African elephant, Linkage Mapper, functional connectivity

2.2 Introduction

There have been widespread and profound anthropogenic impacts on the global environment, which continue to expand and intensify (Venter et al., 2016b; Williams et al., 2020). Two of these impacts are the conversion and fragmentation of natural habitats (Tilman et al., 2017), which are driving biodiversity loss (Foley et al., 2005) and degradation of the associated ecosystem services on which people depend (Millennium Ecosystem Assessment, 2005; Mace *et al.,* 2012). Protected areas (PAs) are among the most important policy instruments for maintaining biodiversity (Bingham et al., 2019; Butchart et al., 2015). The Convention of Biological Diversity's Aichi Target 11 recognised this, agreeing on a global commitment to conserve 17% of the terrestrial realm 'through effectively and equitably managed, ecologically representative, and well-connected systems of protected areas' by 2020. Assessments now show that every component of this target was not met but arguably the biggest shortfall was in developing well-connected systems. This is because a number of recent analyses have shown systemic failure to measure and enhance connectivity between PAs (Saura et al., 2017), so that fewer than 10% of the world's terrestrial PAs are estimated as structurally connected through intact habitats (Saura et al., 2019; Ward et al., 2020). Such isolation prevents PAs from maintaining viable populations of many species and undermines the ecosystem services they provide.

We now need a greater focus on maintaining and restoring landscape connectivity, which is defined as "the degree to which the landscape facilitates […] movement among resource patches" for individual species (Taylor et al., 1993). The most straightforward approach to assessing landscape connectivity is identifying structural linkages of intact habitat (e.g. Ward *et al*., 2020). More sophisticated measures calculate functional connectivity by incorporating data on landscape permeability based on species-specific traits, reflecting both landscape structure and a species' response to it (Bélisle, 2005). Least-cost modelling and circuit theory are two complementary and widely applied means to map functional connectivity (Zeller *et al.,* 2012). Least-cost paths represent the singular most efficient route between two locations, whereas circuit theory treats movement as akin to an electrical current that moves omnidirectionally across the landscape, identifying all potential routes and showing where movement is constrained through 'pinch points' (McRae *et al.,* 2016). Both least-cost modelling and circuit theory rely on the same two inputs: core areas and resistance surfaces. Core areas are the habitat patches between which connectivity is modelled, whereas resistance surfaces are raster layers where each pixel represents the permeability or physiological cost of a species moving through the landscape (Spear et al., 2010).

These surfaces are created by assigning values to environmental or structural characteristics such as habitats, terrain or other physical features that can inhibit or facilitate movement (Pullinger & Johnson, 2010). However, it is not just the physical geographic features of a landscape that influence how permeable it is to a species. Human-wildlife interactions and land use will have a major role to play (Ghoddousi et al., 2020). Therefore, connectivity modelling approaches must also account for the social and socio-political factors that define species movements (Watson et al., 2016). One approach is to use exploratory scenarios (referred to as ‘land-use options’ hereafter) to evaluate the effects of different ecological and social factors on keeping the landscape connected. This can be particularly useful in areas where land-use patterns are complex (Bohensky *et al.*, 2006; Mckenzie *et al.*, 2012). For instance, modelling potential future connectivity can provide valuable insights into how land-use change may influence species movements in the unprotected matrix between PAs. Additionally, it can be helpful to assess connectivity within PAs, which might also alter as a consequence of land-use change. This is important because many PAs contain mixed-use areas that permit agriculture, plantation forestry or other land-uses that are not always well suited to the needs of specific species (Jones et al., 2018; Tucker et al., 2018).

Integrating landscape connectivity planning into broader land-use planning is especially important for African nations. Across the continent, over 8,500 PAs cover an estimated 14% (IUCN and UNEP-WCMC, 2021) of the land area. Nonetheless, structural connectivity is poor, with 0.5% of these PAs connected through intact habitat, compared to the global average of 9.7% (Ward et al., 2020). This is a serious concern in Sub-Saharan Africa, where PAs protect a wealth of biodiversity, including the greatest diversity of extant megafauna (Ripple et al., 2015, 2016; Stuart, 2015). These species need large, well-connected landscapes to persist. Here we focus on an important PA network in Kenya to show how exploratory land-use options can provide insights into how future landcover change could impact landscape connectivity, using the African savannah elephant (*Loxodonta africana*) as a surrogate species. Elephants are often used in connectivity assessments because they are well-studied, conspicuous and move long distances (Epps *et al.*, 2011; Roever *et al.*, 2013). This makes them useful surrogates for other taxa where their long-term persistence depends on creating large, connected landscapes (e.g. Estes *et al.*, 2012). However, the cost of collecting elephant movement data means that records are often limited to a few individuals for any particular region, and extrapolating from those individuals can be problematic because they are likely to exhibit idiosyncratic movement patterns and pathway fidelity (e.g. Ngene *et al.*, 2010). Thus, additional steps are often needed to interpret field data in a way that can be used to create resistance surfaces for modelling landscape connectivity for elephants.

One approach for filling the data gaps when developing landscape level resistance surfaces for a species is to use expert elicitation (e.g. van de Perre et al. 2014), and this is particularly appropriate for elephants because many people study the species. However, expert opinion processes must minimise the numerous psychological and subjective biases, with careful thought given to study design to ensure they produce robust measures. This commonly means applying structured elicitation techniques (Martin et al., 2012; Speirs-Bridge et al., 2010) and anonymising responses during consensus building approaches (Burgman et al., 2011). The Delphi Technique is one such approach popular for determining the range of opinions, where responses are then shared with other panellists to move towards an agreement. However, to ensure connectivity assessments are as robust as possible, movement estimates should be elicited from experts and then validated with empirical data wherever it exists (Zeller *et al.*, 2012).

Our study focused on Kenya's Central Highlands, a region where land-use is complex (Kiteme et al., 2008), leading to it being recognised as a national priority for landscape connectivity, both between PAs (Government of Kenya, 2017) and within them (Government of Kenya, 2018a). Our overarching goal was to identify the important linkages across the landscape, based on current land use and five future land-use options. This involved: (i) identifying core areas for elephants and producing resistance surfaces that represent the permeability of each landcover type for elephants moving across the landscape; (ii) modelling landcover under the future land-use options, and (iii) carrying out connectivity analyses to identify important linkages and pinch points and model how these would be impacted by landcover changes.

2.3 Methods

2.3.1 Study area

Our analyses focus on Kenya's Central Highlands (Figure 2.1), which we delineated based on stakeholder interviews (see Section 2.3.2). The area centred on the PAs of Mount Kenya and included the adjacent land that links them to the contiguous PAs of the larger Aberdare and Laikipia-Samburu ecosystems. These linkages pass through three isolated PAs, Nyeri Forest Reserve and Sangare and Solio. The latter two are Wildlife Conservancies, which are private- or community-owned protected areas managed for wildlife conservation and other compatible land-uses that improve local livelihoods (Government of Kenya, 2018b). This study area also includes the unprotected matrix between PAs where linkage endeavours were active or being considered, which consist largely of communally- and privately-owned agricultural lands with remnant natural vegetation patches. The study area was once heavily forested but has undergone extensive land conversion and degradation. Much of the lower altitude forest and savannah habitats have been deforested for timber, agriculture and cattle ranching (Emerton, 1999; Gathaara et al., 1999; Kiteme et al., 2008), and the primary land-uses are now farmland or exotic plantation forestry.

This study area is home to a diverse megafaunal assemblage (Didier et al., 2011; Ogutu et al., 2016) and contains three largely distinct elephant populations. The PAs of Mount Kenya are estimated to house a population of around 2,500 forest-dwelling African savannah elephants (Vanleeuwe & Gitau, 2020). These regularly move within the PAs between the main forest on Mount Kenya to the lower PA sections of Imenti and Thego (Figure 2.1) through multiple-use areas that comprise of plantation forestry, agriculture and permanent settlements. In the rainy season, at least 50-100 elephants from Mount Kenya then move between PAs to Samburu, as well as through the isolated PAs of Sangare and Solio into Ol Pejeta. These link to the contiguous PAs of Laikipia-Samburu, which are home to a further 6,000 to 8,000 elephants. Elephants have also attempted to move south from Sangare through Nyeri Forest to reach the Aberdares, an isolated mountain forest PA estimated to support another 3,000 to 5,000 elephants (Vanleeuwe & Lambrechts, 2017). These elephant populations together represent the second largest group in Kenya (Thouless et al., 2016).

Human-wildlife conflicts occur regularly in this landscape (Graham and Ochieng, 2008; Kamweya *et al.*, 2012a) and are most pronounced when elephants attempt to move seasonally between PAs. Enhancing connectivity across the study area is an ongoing discussion. It has been met with resistance from local smallholders (Kamweya *et al.*, 2012b), but received support from large agribusinesses who have set-aside land to allow the movement of elephants and other species, including African lions, leopards, wild dogs and Cape buffalo (Nyaligu & Weeks, 2013).

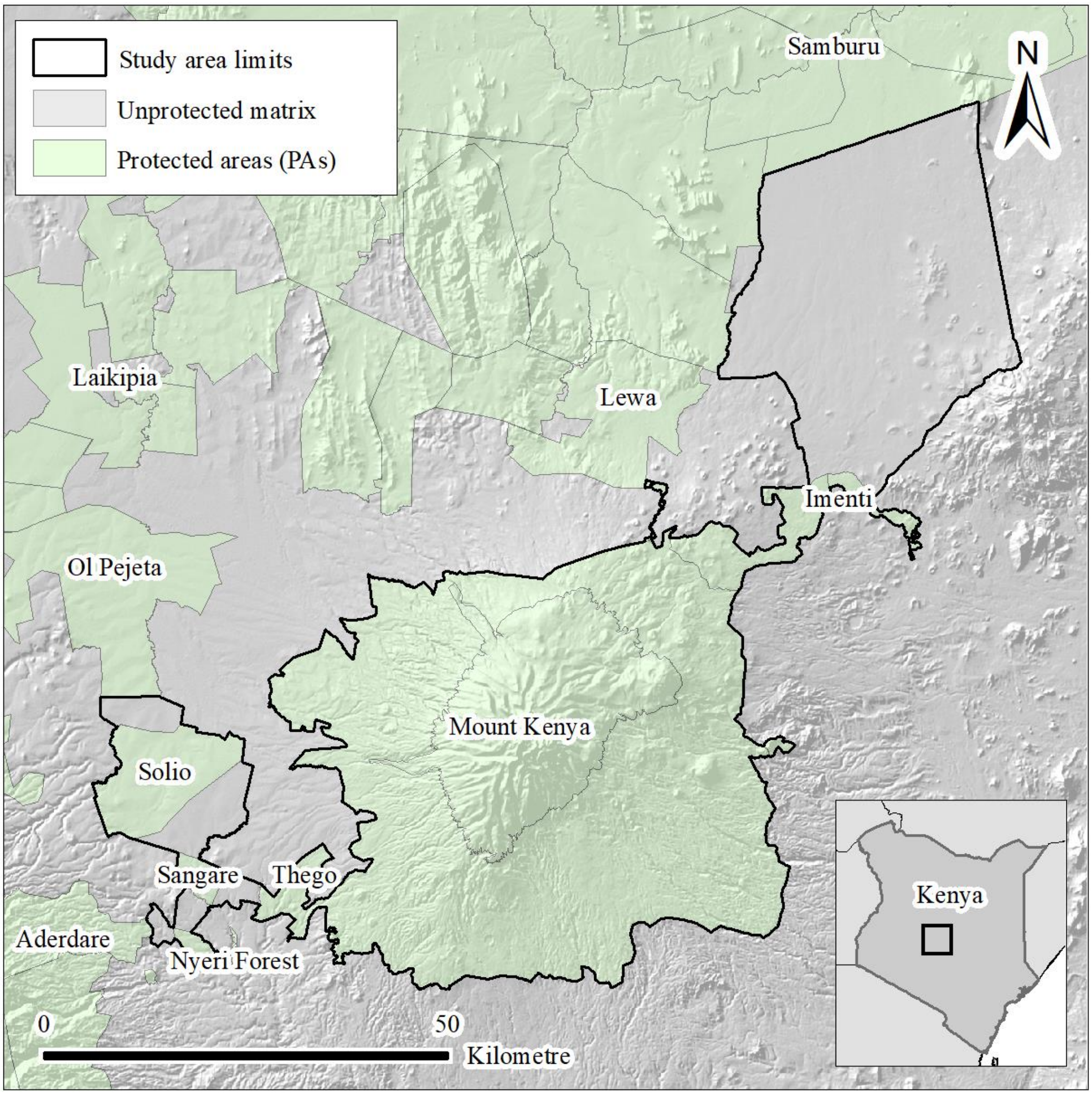


Figure 2.1: The stakeholder-defined study area (outlined in black) where we mapped landscape connectivity under current and future land-use. The study area comprises of the protected areas of Mount Kenya and the sections of the surrounding matrix of unprotected lands and additional protected areas that connect them to the contiguous protected areas of Aberdare, Laikipia and Samburu. Inset map shows the location of the study area within Kenya.

2.3.2 Semi-structured interviews

We used semi-structured interviews with stakeholders (N = 30) to identify our elephant experts, define the area relevant for ensuring landscape connectivity and map future land-use options to investigate the dynamics that drive it. We recruited stakeholders using a mixed approach to reduce biases that can arise from using a single method. First, we used snowball sampling, which asked the initial interviewees to identify further potential stakeholders. This started with five local conservation NGO staff and two protected area managers from the government wildlife agency. However, snowball sampling might over-represent similar perspectives (Sadler et al., 2010). We, therefore, additionally used the outputs of a stakeholder analysis to identify new potentially relevant interviewees, finishing conducting interviews at a point when responses reached saturation, with no new information reported.

The interviews asked a series of open questions to help define the extent of the study area where connectivity should be enhanced and to identify the different land-uses affecting elephant movement. Stakeholders were also asked to discuss and indicate the spatial areas they thought were important for connectivity and the potential land-use options following a 2018 moratorium in forest resource extraction within PAs (Government of Kenya, 2018a). These were framed within Kenya's Vision 2030 developmental plan (Government of Kenya, 2007). We then coded the interview transcripts using the software NVivo 12 (QSR International, 2018) and carried out analysis using grounded theory (Charmaz & Belgrave, 2015).

Once we had analysed all the interview transcripts, we shared a map of the study area limits and the proposed land-use options in narrative form with interviewees, so they could provide feedback and validation. This was to make sure the land-use options acted as exploratory scenarios that accurately captured distinct, divergent and plausible futures (Peterson *et al.* 2003; Mckenzie *et al.*, 2012). We next mapped the future land-use options. We did so in collaboration with the Kenya Wildlife Service, Forest Service and the Regional Centre for Mapping of Resources for Development by first mapping current landcover within the study area using a combination of remote sensing and digitising (see Supplementary Materials Section S2.1). This used the African Union's SLEEK project standard practice, applying a random sampling methodology to ground-truth the outputs (African Union, 2016). This produced a map of landcovers in January 2018, which we used as our '*Baseline*' against which we could compare the future changes in connectivity under the five land-use options. We next applied the landcover changes detailed in the narratives to transform the current landcover into future landcover maps for each of the five land-use options identified. These allowed us to map current connectivity to predict how it would change.

2.3.3 Connectivity under the different future land-use options

Expert elicitation

We elicited expert opinions (N = 14) on the permeability of the different landcover and land-use types in the study area to elephant movement. Data were collected through an online survey distributed to elephant experts identified through the interviews (see Figure S2.1). Experts scored the permeability of each landcover type to elephant movements using a four-point structured question. They first estimated the lower bound estimate of the likelihood of elephant movement from a value of 1 for no movement to 100 for free movement. They then gave an estimate for the upper bound and a best average estimate. They were next asked their confidence, from 50% for an even chance to 100% for certainty, that the actual likelihood would fall within the bounds. This procedure generates more reasoned and accurate responses than requesting a single best estimate value (Speirs-Bridge et al., 2010). To account for differences in elephant movement behaviour within and between PAs (Graham et al., 2010; Ihwagi et al., 2015), we first asked them to score the ease with which elephants would move across a specific landcover type if it were within a PA, and then if it was in the unprotected matrix between PAs. We also asked them to score the likelihood of movement for both sexes together; after discussion with experts during piloting recognised that both sexes were moving between PAs within the study area.

We then shared the scores and comments from this initial online survey among all experts as a plot with each individual's values overlaid onto the anonymised values of all other experts (Figure S2.1). Verbal discussions were then offered on a one-to-one basis with the experts to encourage reflection and the potential for revision of scores. This is a modified version of the Delphi Technique that aims to further increase the accuracy of expert opinion by building consensus (Linstone & Turoff, 1975; Mukherjee et al., 2015). Five experts joined one-on-one discussions and a total of 11 adjusted their best estimate scores.

Connectivity analyses

To run our connectivity analyses, we defined core areas as patches of contiguous natural primary vegetation more than 10 km^2 found within the PAs of our study area. Experts agreed that habitat patches of this size offered a large enough area for elephants to occupy without moving each day. We created resistance surfaces through areas of secondary vegetation within the PAs of Mount Kenya and the unprotected matrix between PAs using our *Baseline* landcover map and the five future land-use options (Figure 2.2). We maintained the 10 metre resolution from the landcover map to ensure that the finer-scale landcover features that determine elephant movement were adequately captured and to maximise the accuracy of our connectivity modelling (Zeller et al., 2012). We used the resistance values derived from the mean best estimate value of landcover permeability for elephants from our expert elicitation process. To do so, we inverted the permeability values to represent resistance, then normalised the range of values, which amplified the variation between landcovers (Zeller et al., 2012) and applied these to the maps. As elephants within the study system avoid slopes of over 30 degrees (Wall et al., 2006), we gave these areas a maximum resistance value.

To ensure the resistance surfaces were accurate, we next validated the *Baseline* resistance surface by investigating whether the locations of six tracked elephants were more likely, on average, to be in low-resistance habitats than randomly located points. We chose location data from two females and four males tracked between Jan 2016 and Dec 2017. We treated these locations as individual point selections (Zeller *et al.*, 2012). We separated the locations falling within PAs (N = 49,883) from those found in the unprotected matrix between PAs (N = 4,787) to align with our expert elicitation methodology. We then followed the validation technique developed by Osipova et al. (2018), which accounted for pseudo-replication and spatial autocorrelation by using a random number function to select one location point for each individual from the first 12 hours and one from the second 12 hours of each day. This is when most elephant movements occur (Wyatt & Eltringham, 1974) and doing so, avoided selecting multiple points from one journey or resting period. Following the Osipova method, we then selected the same number of location points from each individual to not bias our data with over-representation of animals that exhibit individualistic movement patterns (Ihwagi et al., 2019; Ngene et al., 2010). To account for positioning error (Adriaensen et al., 2003) and the species' perception of heterogeneity in the landscape, we created a buffer of 100 m radius around each point (a total of 206 from within PAs and 123 between PAs) (Weins, 1989; Zeller et al., 2012). We then calculated the median value found within these buffers from the respective resistance surface. These values were then compared to values extracted from a randomly located set of buffers of the same sample size using a Welch two-sample t-test (Koen et al., 2014). Validation was undertaken in the R (R Core Team, 2019) and analysed using the packages 'sp' (Pebesma & Bivand, 2005) and 'raster' (Hijmans, 2019).

After validation, we analysed our *Baseline*, and five future land-use option resistance surfaces through the Linkage Mapper plug-in (McRae & Kavanagh, 2011) for ArcGIS (ESRI, 2017). We first modelled connectivity between core areas, calculating the least-cost paths between our core areas. This produced a linkage-specific metric of the ratio of cost weighted distance (the least cost path over the resistance surface) to Euclidean distance (the shortest path), hereafter 'linkage quality', representing the difficulty of movement between the core areas given how close they are. It also produced a ratio of cost weight distance to least-cost path length, hereafter 'optimal path resistance', representing the mean resistance along the least-cost path between two core areas. We ran analyses with a corridor cut off length of 100 km, which our experts felt was the upper limit associated with a single continuous elephant movement. Given that the longest Euclidean distance between PAs within our study system was 40 km, this was a reasonable assumption.

We then ran Pinchpoint Mapper (McRae, 2012), which uses circuit theory to simulate omnidirectional movement across the resistance surface between two core areas, identifying multiple routes and areas of restricted movement. This approach relies on another metric, the cost-weighted width of a corridor. This represents the amount of sideways movement the species will tolerate when moving between two core areas (McRae *et al.*, 2016). We undertook a sensitivity analysis, exploring the impact of cut-off widths from one to 200 cost-weighted kilometres, which made little difference beyond a value of 10. As elephants regularly move through suboptimal habitat when crop-raiding or streaking from one area of safe habitat to the next (e.g. Graham *et al.*, 2010), we continued with the value of 10. Pinchpoint Mapper identified the 'pinch points' and provided another linkage-specific metric, the ratio of cost weight distance to effective resistance, hereafter 'alternative pathways'. Finally, we applied Centrality Mapper (McRae, 2012), a circuit theory tool that measures the importance of individual linkages and core areas for keeping the broader network of the study system connected (Carroll *et al.*, 2011). This created the last linkage-specific metric, 'linkage centrality'. We next calculated pairwise cumulative current flow across all core areas to get centrality values for the core areas, which were then corrected for the area of each core area to create a core area-specific metric, 'area-corrected centrality'. This indicates the importance of a core area in keeping the network connected in relation to their land area, with higher value areas being important 'stepping stones' for landscape connectivity across the network (Dutta et al., 2016).

2.4 Results

2.4.1 Mapping future land-use options

When asked about land-uses within our study system, 40% of interviewees identified agricultural land-uses (n =12), 37% silvicultural land-uses (n =11), 43% biodiversity conservation (n =13) and landscape connectivity land-uses, and 40% water resource conservation land-uses (n =12). We used landcover current as of January 2018 as a Baseline and then developed a ruleset to change these into five land-use options into maps of potential future landcover (Figure 2.2; Section S2.1). These were:

(i) *'Business As Usual'*, which described the continuation of planned harvesting and replanting of exotic timber plantations, as well as agricultural expansion outside of protected areas in previously cleared areas.

(ii) '*Farming Future'*, where all secondary habitats and cultivation within mixed-use areas of Mount Kenya's PAs and outside PAs became annual crops.

(iii) *'Forestry Future'*, where all secondary habitats and cultivation within mixed-use areas of Mount Kenya's PAs and outside PAs became exotic timber plantations.

(iv) *'Green Future'*, where all secondary habitats and cultivation within Mount Kenya's PAs were reforested and 300 metre wide grassland elephant corridors were established between protected areas.

(v) '*Blue Future'*, which described specific policy recommendations including the reforestation of exotic timber plantations more than 500 metres inside Mount Kenya's PA boundary, the reforestation of riparian reserves and the relocation of exotic timber plantations outside of Mount Kenya's PAs (Government of Kenya, 2018a).

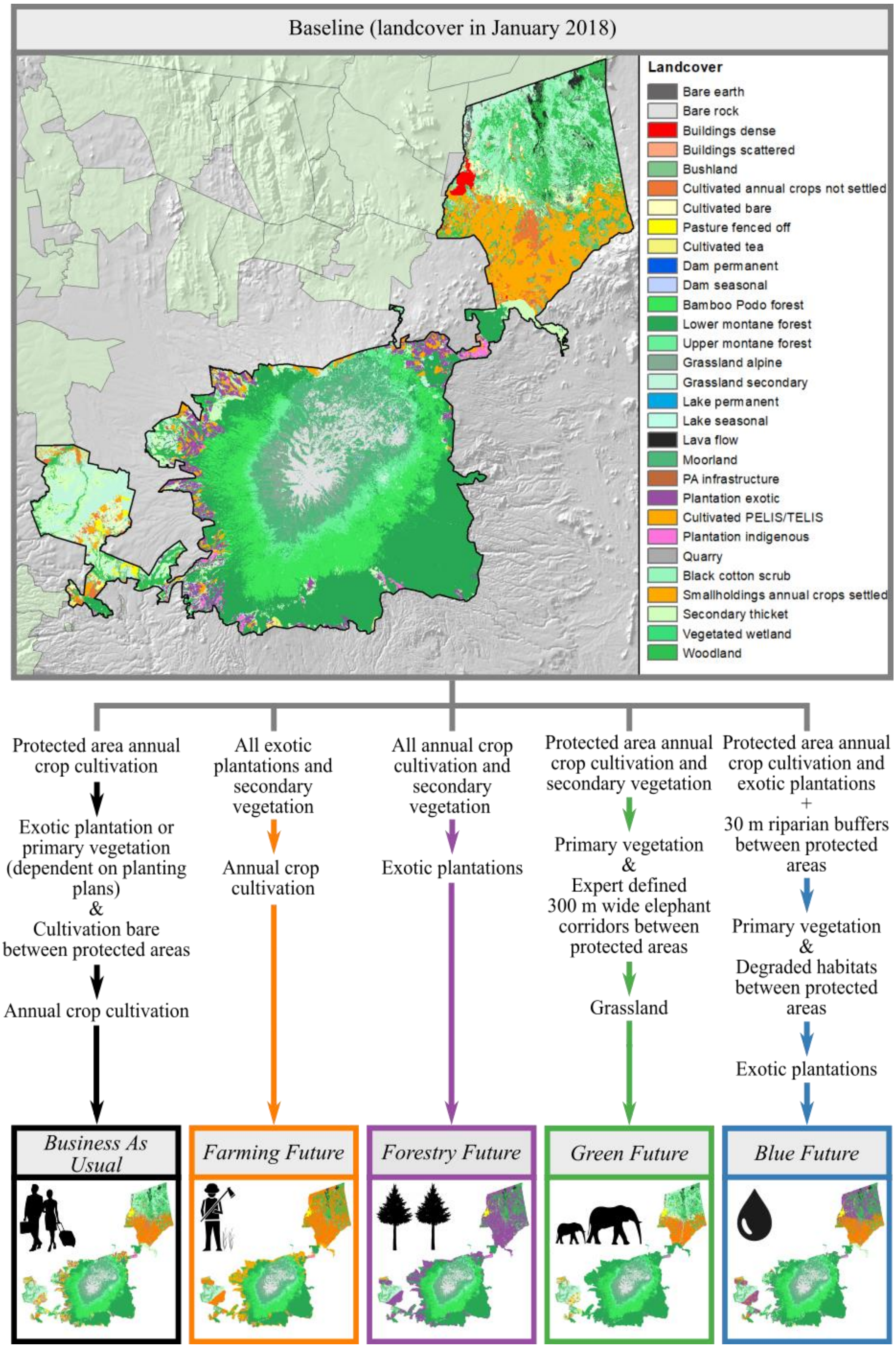


Figure 2.2: A representation of how current landcover, correct as of January 2018 (grey box), was used as the *Baseline* and was transformed into our five future land-use options: (i) *Business As Usual* (black); (ii) *Farming Future* (orange); (iii) *Forestry Future* (purple); (iv) *Green Future* (green); and, (v) *Blue Future* (blue). Landcover changes only occurred in mixed-used areas of Mount Kenya's protected areas and in areas that are not formally protected.

2.4.2 Expert elicitation

We elicited the opinion of 14 experts comprising protected area managers, conservation practitioners and researchers with experience of elephant movements within our study area. The Delphi Technique showed expert agreement through convergence in opinion, with the overall standard deviation from the mean reducing. This provided resistance values for the permeability of landcovers to elephant movement within and between PAs (Table 2.1; SM 2.2).

Table 2.1: Expert elicited opinion on the permeability of landcovers to elephant movement within and between protected areas (PAs). These are shown as the mean and standard deviation (SD) for the best estimate and the values which were inverted to represent resistance then normalised along a scale of 1 for low resistance and 100 for full resistance values

Landcover type	Mean best estimate across experts	SD	Normalised resistance value
Tea cultivation within PA	18.31	13.82	100
Annual crop cultivation within PA	28.31	15.88	86
Moorland within PA	36.38	20.81	75
Bare earth within PA	44.38	17.46	63
Exotic plantation within PA	51.92	14.51	52
Upper montane forest within PA	62.31	21.08	38
Bamboo forest within PA	65.77	17.18	33
Indigenous plantation within PA	69.23	12.56	28
Bushland within PA	71.92	13	24
Secondary grassland within PA	71.54	15.19	24
Secondary thicket within PA	73.08	21.65	23
Lower montane forest within PA	85.77	10.14	4
Woodland within PA	87.54	8.54	1
Dense buildings between PA	6.77	5.78	100
Tea cultivation between PA	22.38	17.46	80
Scattered buildings between PA	26.92	18.74	74
Fenced pasture between PA	38.23	21.27	60
Smallholding between PA	39.62	20.56	58
Cultivation bare between PA	40.77	20.98	57
Bare earth between PA	47.31	20.17	48
Annual crop cultivation between PA	56.54	21.15	37
Exotic plantation between PA	59.62	16.26	33
Scrubland between PA	63.46	18.53	28
Bushland between PA	79.62	12.33	7
Woodland between PA	84.62	10.89	2
Grassland between PA	85.38	11.63	1

2.4.3 Connectivity analyses

We validated our *Baseline* resistance surface with empirical elephant movement data, finding significantly lower resistance values around elephant location points than around the random locations within ($t = -14.91$, $p = <0.001$) and between PAs ($t = -3.86$, $p = <0.001$). We then modelled connectivity among our ten core areas (Table 2.2), identifying nine linkages, three occurring within PAs and the remaining six between PAs (Table 2.3; Figure 2.3).

Table 2.2: The ten core areas (Figure 2.3) included in our connectivity analyses, the larger protected area they were a part of and their centrality measures. Core area centrality represents the importance of individual core areas in keeping the whole network connected. In contrast, area-corrected centrality is the importance of a core area in keeping the network connected in relation to their land area (km^2), where a higher value indicates the core area is an important 'stepping stone' (Dutta et al., 2016).

Core area name	Larger protected area	Area km^2	Core area centrality	Area-corrected centrality
MK Forest	Mount Kenya	1282	29	0.023
Imenti	Mount Kenya	47	17	0.358
Lewa	-	370	9	0.024
Thego	Mount Kenya	35	29	0.838
Samburu	-	1066	9	0.008
Sangare	-	23	33	1.455
Solio	-	164	17	0.104
Nyeri Forest	-	11	17	1.437
Ol Pejeta	-	365	9	0.025
Aberdare	-	1750	9	0.005

Table 2.3: Linkage metrics for the *Baseline* landcover map (Figure 2.2): linkage quality (the ratio of cost weight distance to Euclidean distance), where a lower value is better quality; optimal path resistance (the ratio of cost weight distance to least-cost path length) where a higher value shows greater resistance along the least-cost path; alternative pathways (ratio of cost weight distance to effective resistance), where a higher value shows more alternative routes between core areas; and, linkage centrality, where a higher value shows greater importance of a linkage in keeping the network connected.

Core area 1	Core area 2	Linkage quality	Optimal path resistance	Alternative pathway availability	Linkage centrality
MK Forest	Imenti	7.39	4.64	86.75	16
MK Forest	Lewa	13.9	7.91	23.41	9
MK Forest	Thego	5.79	3.09	72.82	24
Imenti	Samburu	7.14	4.89	49.95	9
Thego	Sangare	3.05	2.21	147.2	25
Sangare	Solio	2.18	1.78	215.53	16
Sangare	Nyeri Forest	10.53	6.99	91.18	16
Solio	Ol Pejeta	1.6	1.37	404.23	9
Nyeri Forest	Aberdare	3.49	2.81	75.23	9

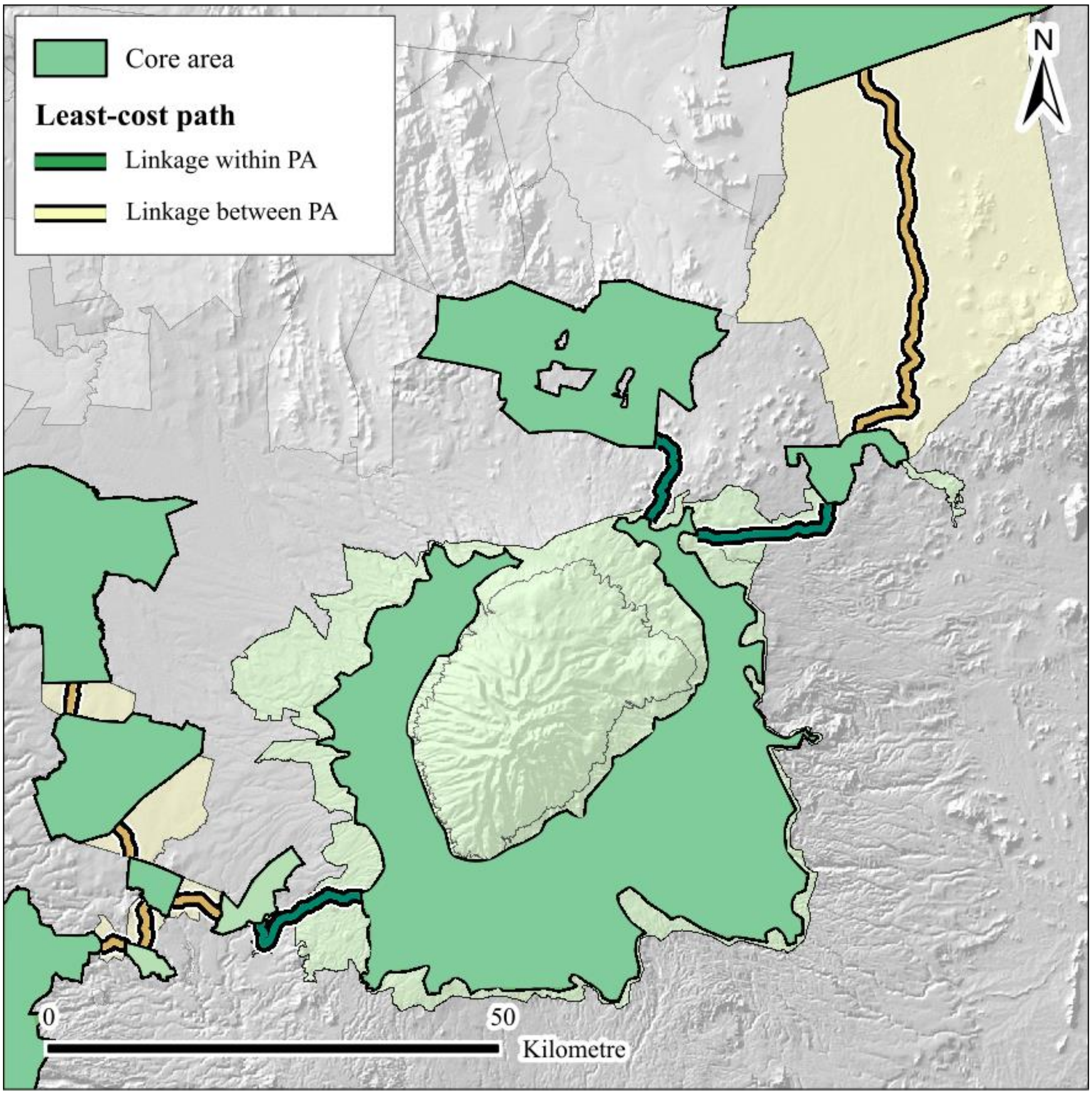


Figure 2.3: The core areas (mid-green) between which connectivity was modelled through least-cost paths within PAs (dark green) and between PAs (brown). Least-cost paths are the route across the resistance surface between two core areas with the lowest effective distance.

Baseline values represent current connectivity. Linkage quality, the difficulty of movement between the core areas given how close they are, had a lower range within PAs than between PAs, and optimal path resistance had a higher range within PAs than between them (Table 2.3; Figure 2.4). Alternative pathways had a higher range between PAs (Table 2.3; Figure 2.4 and 2.5). Centrality measures did not change under the different land-use options because changes in landcover did not affect the spatial position of the linkages or core areas within the overall network. The most important linkages for maintaining connectivity across the study area measured by linkage centrality (Table 2.3) were between Thego and Sangare, then MK

Forest and Thego. The most important core areas, as defined by area-corrected centrality, which represents the most important 'steppingstones' (Dutta et al., 2016), were Sangare, followed by Nyeri Forest, Thego and Imenti (Table 2.2).

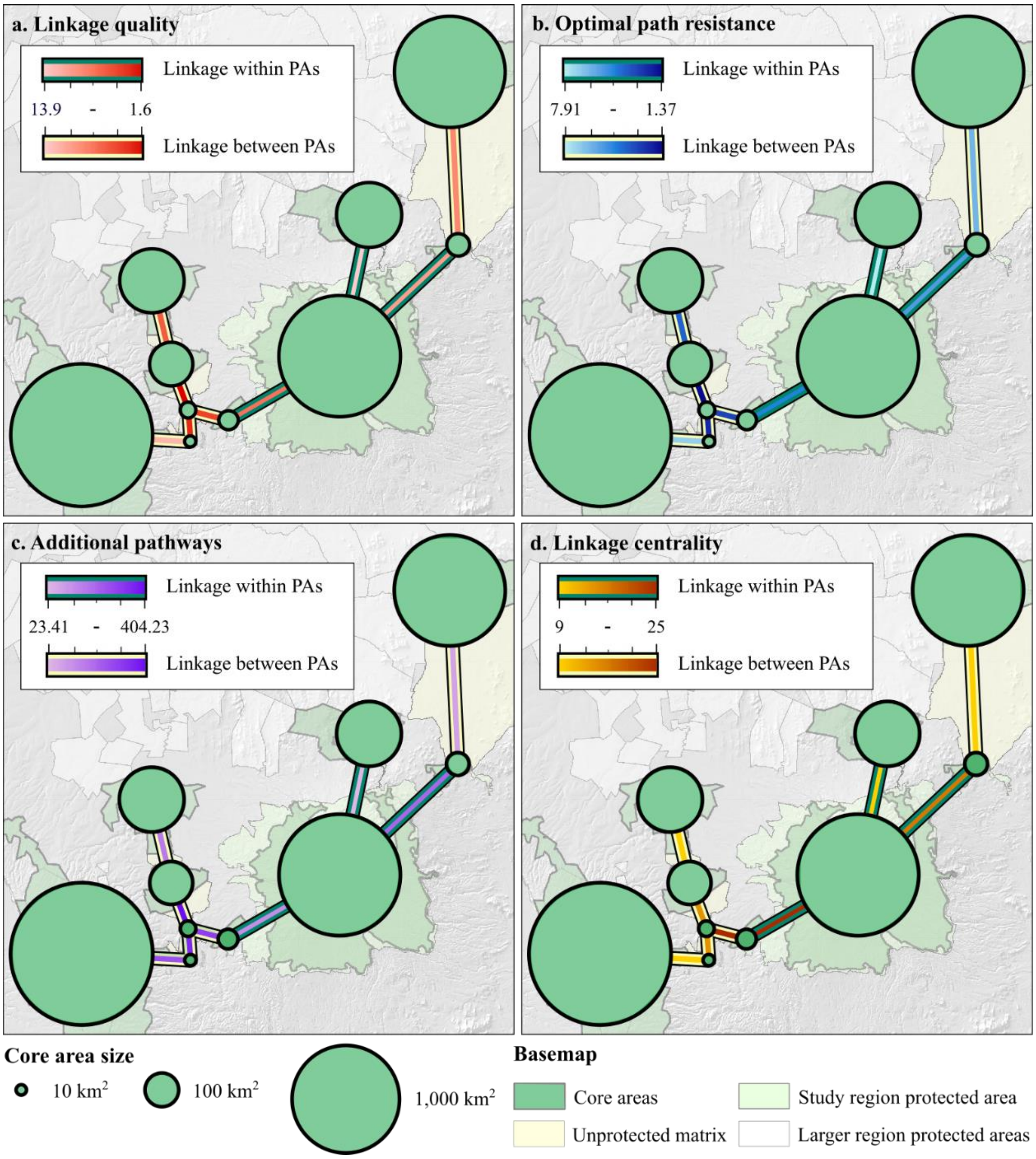
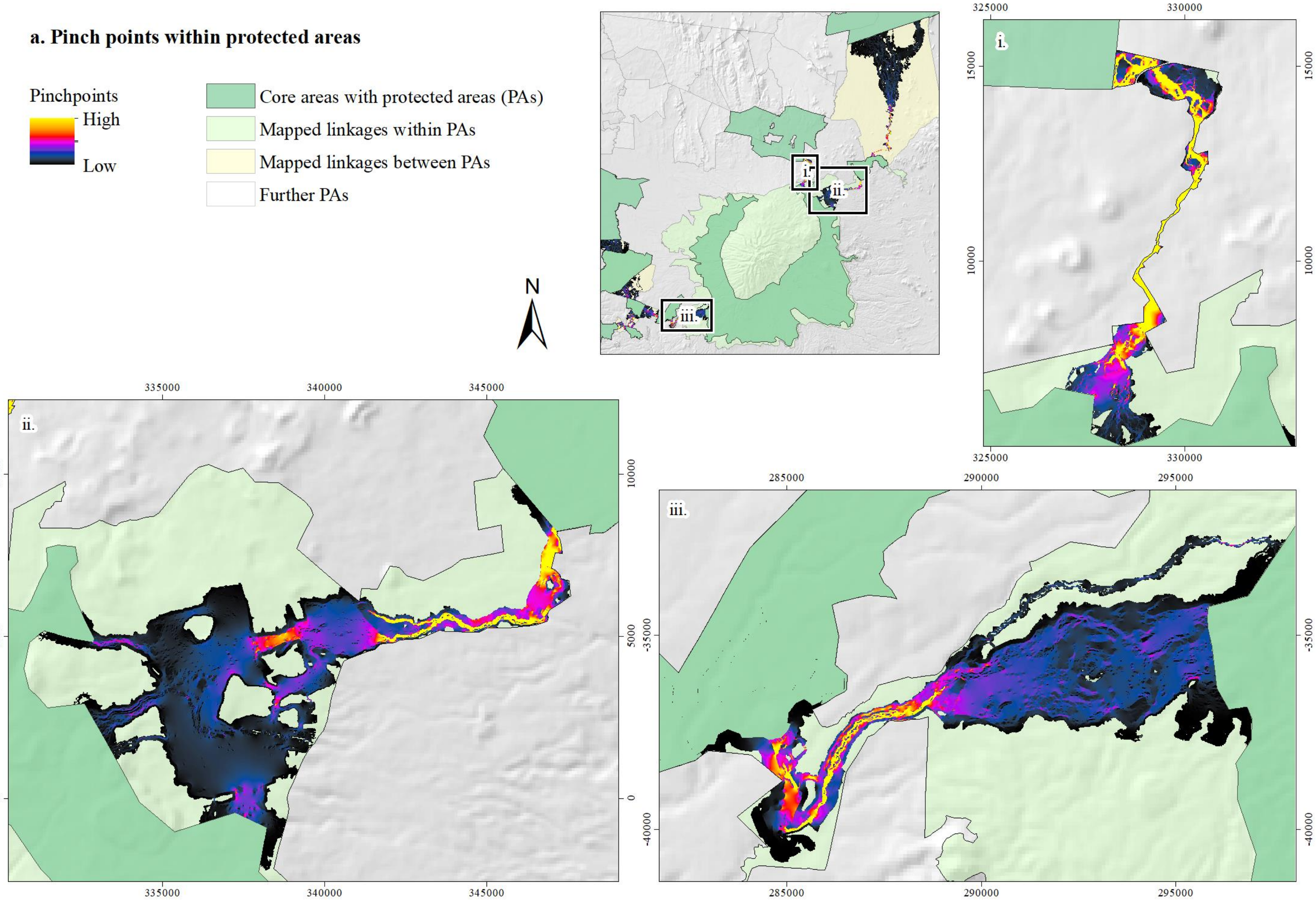


Figure 2.4: A ‘tube map’ style data visualisation showing linkage connectivity metrics between core areas (Figure 2.3) for the Baseline landcover map (Figure 2.2) for (a) Linkage quality, the ratio of cost weight distance to Euclidean distance, where a value of one is the best quality; (b) Optimal path resistance, the ratio of cost weight distance to least-cost path length, where a higher value indicates more difficult movement; (c) Alternative pathways (purple) shows the ratio of cost weight distance to effective resistance, where a higher value shows more alternative routes; and, (d) linkage centrality (orange), where a higher value shows greater importance in keeping the broader network connected.


a. Pinch points within protected areas
Pinchpoints
High
Low
Core areas with protected areas (PAs)
Mapped linkages within PAs
Mapped linkages between PAs
Further PAs
N
i.
ii.
iii.
325000
330000
15000
10000
335000
340000
345000
5000
0
285000
290000
295000
-35000
-40000

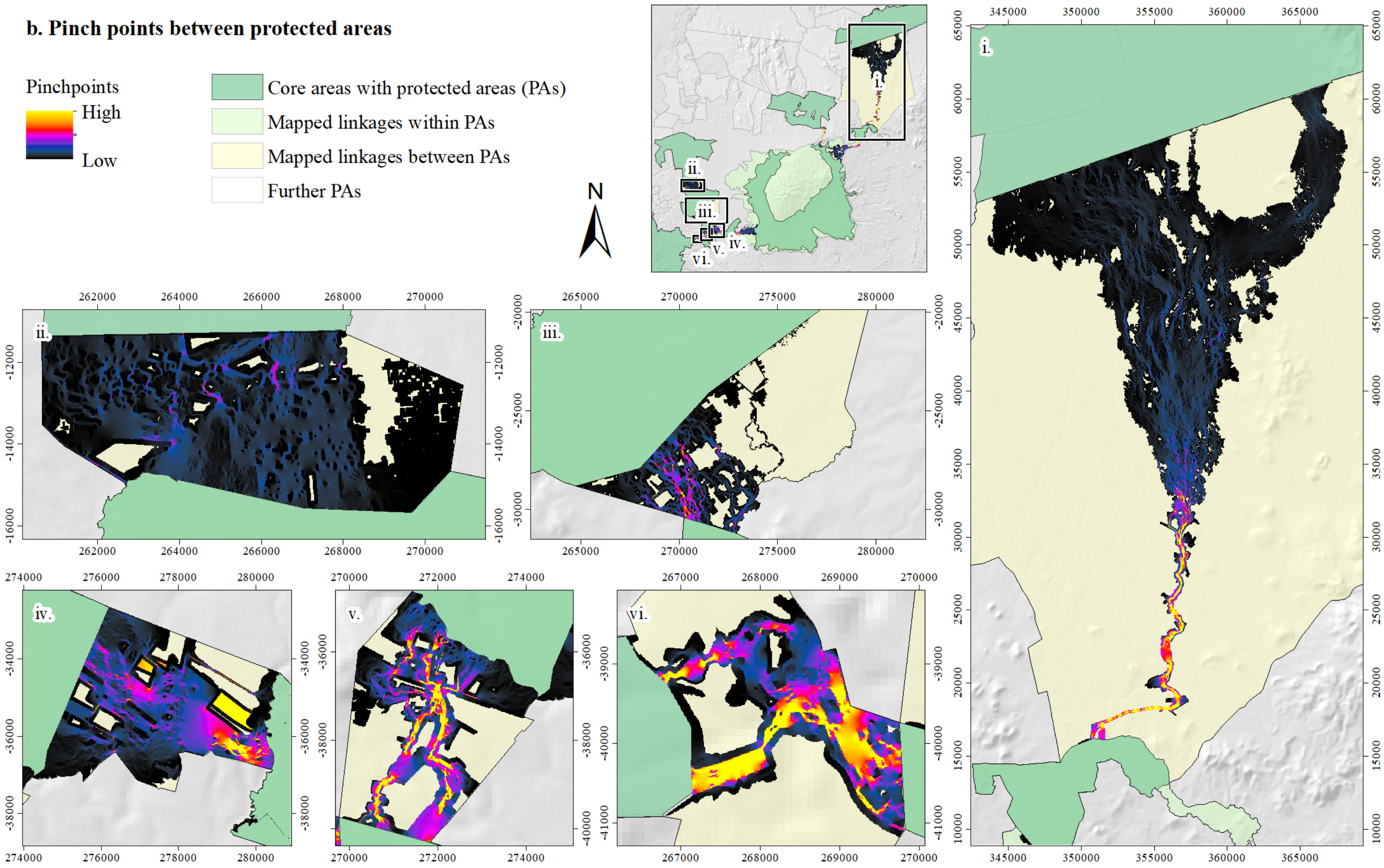


Figure 2.5: The 'pinch points' (McRae *et al.*, 2016) representing areas of constrained movement where fewer alternative routes are available (a) within and (b) between protected areas for *Baseline*. Grid values on insets allow location NB using UTM Zone 37N coordinates.

Changes in the other connectivity metrics for the five future land-use options are represented by comparing the mean values across linkages to those from *Baseline* (Table 2.4; Table S2.2). Under *Business As Usual* linkage quality deteriorated and higher optimal path resistance occurred both within and between PAs, whereas more additional pathways were available inside PAs. *Farming Future* saw the most pronounced loss of linkage quality, increase in optimal path resistance and fewer alternative pathways between PAs. However, there was a marginal increase in alternative pathways within PAs (1%). *Forestry Future* also saw connectivity deteriorate across all metrics, with losses around half those of *Farming Future*, except for alternative pathways, which showed the only deterioration within PAs. *Green Future* saw the most pronounced improvements across all metrics. In comparison, *Blue Future* showed mixed results with generally improved within PA connectivity and deteriorated between PA connectivity.

Table 2.4: Percentage change between the Baseline landcover map (Figure 2.2) and the future land-use options grouped by linkage location for: linkage quality (the ratio of cost weight distance to Euclidean distance), where a lower value is better quality; optimal path resistance (the ratio of cost weight distance to least-cost path length), where a higher value indicates more difficult movement and, alternative pathways (ratio of cost weight distance to effective resistance), where a higher value shows more alternative routes.

Land-use option	**Linkage location**	**Linkage quality**	**% change**	**Optimal path resistance**	**% change**	**Alternative pathways**	**% change**
Baseline	within PA	9.03	-	5.21	-	60.99	-
Business As Usual	within PA	10.07	-12%	5.95	-14%	73.83	21%
Farming Future	within PA	13.17	-46%	7.49	-44%	61.34	1%
Forestry Future	within PA	11.02	-22%	6.29	-21%	57.4	-6%
Green Future	within PA	5.98	34%	3.94	24%	231.2	279%
Blue Future	within PA	6.57	27%	4.06	22%	139.36	128%
Baseline	between PAs	4.66	-	3.34	-	163.89	-
Business As Usual	between PAs	4.87	-5%	3.42	-2%	164.12	0%
Farming Future	between PAs	16.37	-251%	13.45	-303%	84.6	-48%
Forestry Future	between PAs	10.96	-135%	8.84	-165%	130.84	-20%
Green Future	between PAs	1.22	74%	1.04	69%	195.99	20%
Blue Future	between PAs	6.2	-33%	3.25	3%	67.79	-59%

2.5 Discussion

Kenya is a nation with a rapidly growing economy and human population. It also has a wealth of biodiversity both inside and outside of PAs (Tyrrell *et al.*, 2019), with megafaunal populations that need to move through varied landscapes often heavily populated by diverse human communities (Government of Kenya, 2017). This means that landscape connectivity both within and between PAs is under threat and the government has identified connectivity planning as a crucial part of national policy under the Vision 2030 Development Plan (Government of Kenya, 2007, 2017), while also declaring a national moratorium in forest resource extraction within state-run PAs (Government of Kenya, 2018a). We used elephants as a surrogate for broader species movements, recognising that this will not accurately represent all species. Our analysis used a mixed methods approach to understand connectivity in one of Kenya's most important conservation landscapes and in this section, we discuss the modelling process and how it was used to identify linkages and pinch points under current and potential future landcover.

2.5.1 Modelling current connectivity

The savannah elephant is one of the most well-studied large mammals in Africa and research on the Mount Kenya populations has been ongoing for decades (Vanleeuwe, 2010; Vanleeuwe & Gitau, 2020), including a number of studies that have recorded the movement of this species throughout the landscape. However, the cost of collecting these spatial data means that it is limited to a relatively small number of elephants within portions of the study area (Graham et al., 2009; Ihwagi et al., 2019). We were, therefore, unable to use this information to develop the resistance surfaces needed for this study, especially for landcover types outside the PAs that elephants are known to "streak" through and so are rarely recorded (Jachowski, *et al.*, 2013). Instead, we developed an approach based on expert elicitation so that we could consult with a wide range of elephant researchers and produce resistance scores for every landcover type, both within and outside PAs. This approach was time consuming because it used the Delphi Technique, an iterative approach for creating consensus, but produced robust results that were supported by the available movement data collected from collared elephants (Graham et al., 2009; Ihwagi et al., 2019). This suggests the same methodology could be used elsewhere when modelling landscape connectivity for other well-studied species in data-poor regions.

The analysis of our *Baseline* or current connectivity, both within and between PAs in the study area, highlight where the current risks are to landscape connectivity. These identify immediate concerns following the national moratorium on forest resource extraction, which was due in part to concerns about illegal felling within Mount Kenya's PAs. The *Baseline* pinch point maps (Figure 2.5) show critical areas where elephant movement is restricted. These primarily run through plantation forestry areas within the PAs and secondary vegetation between PAs. Here the linkage between MK Forest and Lewa had the lowest quality, highest optimal path resistance and lowest alternative pathways of all PA linkages. This is not surprising, as elephants regularly move through suboptimal habitat (Graham et al., 2010; Ihwagi et al., 2019), but it is notable because the linkage was created in 2012 to divert elephants and stop them from breaking fences and moving through farmland (Nyaligu & Weeks, 2013). After 'seeding' the trial with dung, elephants began to use this pathway overnight and today the project is recognised as a success, with over 1,000 elephant journeys a year (Lewa, unpublished data), and is used by elephants and other species as both a transit corridor and an extension of habitat (Green et al., 2018). Yet, the fact that this linkage performs relatively poorly compared to the two remaining linkages within PAs supports findings that interventions to restore connectivity tend to be less successful than retaining existing connectivity (Rey Benayas et al., 2009).

Area-corrected centrality identified the most important 'stepping stones' for connecting the system, with Sangare, Nyeri Forest and Thego as the most critical for connecting the PAs of Mount Kenya to the contiguous PAs of Aberdare and Laikipia-Samburu. The linkage between Thego and Sangare had the highest centrality value, reinforcing its importance in maintaining overall landscape connectivity. This provides support for plans to establish safe passage for elephants in this location (Rhino Ark Conservation Trust, 2019), with the pinch point maps (Figure 2.5) showing where to best focus efforts spatially. However, it also scored poorly across the first three measures when compared to the linkages between Sangare and Solio and Solio and Ol Pejeta, suggesting that these are better connected at present and maintaining existing connectivity here should also be a priority. These findings provide valuable insights for those planning connectivity conservation, but fostering support from local farmers living between PAs will be essential as they have been opposed to such interventions between Thego and Sangare in the past (Kamweya *et al.*, 2012b).

2.5.2 Modelling future connectivity

Our land-use options for the year 2030 detail divergent land-use futures, showing the drivers of land-use change and capturing how they could further affect landscape connectivity. Premoratorium plans (*Business As Usual*) were predicted to degrade connectivity, except for marginal increases in alternative pathways in two linkages within Mount Kenya's PAs currently undergoing reforestation (Mount Kenya Trust, 2018). Our modelling demonstrated that connectivity across the study area would weaken the most with increased agriculture (*Farming Future*) and, to a lesser degree, silviculture (*Forestry Future*). This is to be expected as agriculture and silviculture are two major drivers of biodiversity and ecosystem service degradation (Bond et al., 2019; Chapin et al., 2000; Curtis et al., 2018).

In comparison, connectivity would improve universally across the study area with reforestation and grassland corridor establishment (*Green Future*). The pronounced increase in alternative pathways within the PAs shows that reforestation would produce bigger improvements than the grassland corridors. The specific spatial zoning recommendations for forest conservation (*Blue Future*) showed similar but less marked improvements, with poorer connectivity between PAs due to expansion of silviculture and the reforested riparian reserves running perpendicular to the least-cost paths. It is important that Kenya's commitment to reforest 5.1 million hectares under the Bonn challenge (IUCN, 2011) should not be met by exotic plantations, which can have negative effects on biodiversity (Bond et al., 2019). Community-led restoration is gaining momentum in the PAs of Mount Kenya (Mount Kenya Trust, 2018), and the *Green Future* and *Blue Future* could contribute to Kenya meeting 0.7% or 0.4% of its reforestation target, respectively. Additionally, tree planting here would capture atmospheric carbon in the area with the highest aboveground carbon capture potential across all of Kenya (Busch et al., 2019). Given forest ecosystem integrity is already globally weakened (Grantham et al., 2020) and there is a long trend of the isolation of forest PAs (DeFries et al., 2005), conservation should capitalise on the ecosystem restoration movement, helping ensure landscape connectivity, as well as broader benefits for biodiversity and local livelihoods (Chazdon & Brancalion, 2019; Wiens & Hobbs, 2015). Ecosystem restoration efforts therefore need to promote multifunctionality to support biodiversity and people (Mansourian et al., 2020; Sayer et al., 2013), ensuing a mix of land sparing and sharing to facilitate species persistence and provide the food and other materials people need (Tilman et al., 2017).

2.5.3 Conclusion

Achieving landscape connectivity will continue to be a key part of the post-2020 targets of the CBD (Convention on Biological Diversity, 2020), especially in light of climate change, as people and species move and have to adapt (Leal Filho *et al.*, 2020; Lottering *et al.*, 2020). At present, most fine-scale studies of connectivity focus on maintaining connectivity, especially in the tropics, where the loss of natural vegetation has been relatively low (Jacobson et al., 2019). However, landcover change in countries like Kenya has accelerated in recent decades, so re-establishing landscape connectivity is becoming increasingly important. Such restoration is generally much slower and more expensive than habitat conservation, so this approach should be secondary to maintaining and protecting existing connectivity (Rey Benayas et al., 2009). However, our results show that restoring small parcels of land can make a large difference, making this approach affordable and feasible.

2.6 Acknowledgements

We thank H. Vanleeuwe for piloting the expert elicitation survey, L. Osipova for guidance on the validation process and T. Dutta for guidance on Linkage Mapper. Ethical approval was given by the University of Kent for the semi-structured interviews (ARW/DN 14/06.17) and expert elicitation process (SAC Ref 8-PGR-19/20). This work was also made possible by research affiliations with the Kenya Forest Service (RESEA/1/KFS/VOL.III(97)) and Kenya Wildlife Service (KWS/BRM/5001).

2.7 Supplementary information

S2.1 Landcover mapping

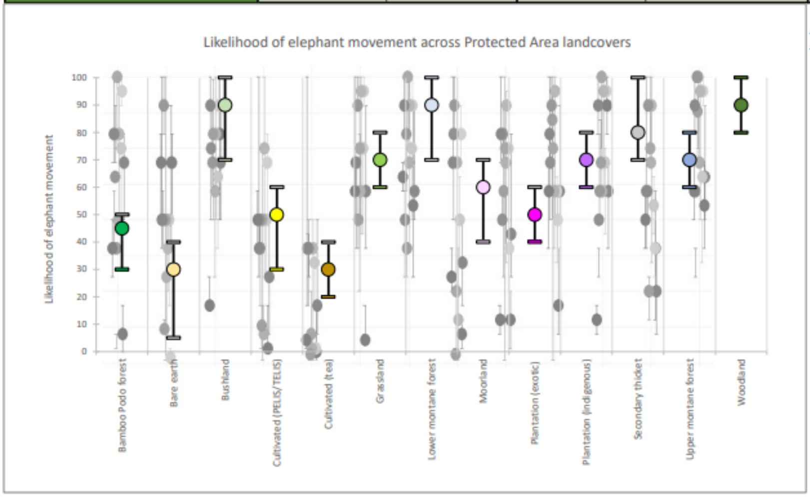
We mapped current landcover within our study area using a combination of remote sensing and digitising. We first developed a typology of different landcovers based on the African Union's Africover project and the regional SLEEK standard (African Union, 2016). We then obtained Sentinel 2a satellite imagery from 29th of January 2018 (ESA, 2018), which consisted of four sets of 10 m resolution raster layers. This date was selected because it had low cloud cover and was taken at the beginning of a moratorium of forest resource extraction (Government of Kenya, 2018a), which represented the starting time point from which our land-use options diverged. The imagery was then clipped to the study area and converted from radiance to reflectance values using a DOS-1 conversion. This increases accuracy of vegetation classification as it represents the physical reflective property of the target material rather than radiance, which is subject to atmospheric effects (Ray, 1994).

We then mapped the natural vegetation using the Semi-Automatic Classification Plugin (Congedo, 2018) for QGIS (QGIS Development Team, 2018). This first involved specifying the location of known patches for each natural vegetation landcover type, based on training sites that had been identified through previous vegetation mapping studies (Bussmann & Beck, 1995; Niemelä & Pellikka, 2004). The software then produced a set of spectral signatures for each landcover type. These were used to classify the remaining pixels using a maximum likelihood algorithm. To improve classification accuracy and remove errors, we put the output through a pixel sieve to remove landcover patches <1 ha as these were likely to be misclassifications. Natural vegetation types are stratified by elevation (Bussmann & Beck, 1995), therefore we next refined classifications by incorporating elevation data by overlaying the output raster onto a digital elevation model (DEM) accessed from the German Aerospace Centre (DLR, 2018). We subsequently used on-screen digitising to map the anthropogenic landcover types. This was guided by existing GIS layers on plantation blocks from the Kenya Forest Service and GG's prior field experience. For this, we used DigitalGlobe's QuickBird satellite imagery (Google, 2018). We used a fixed scale of 1:3,000 and 10 m snapping to already digitised polygons to avoid overlaps. The results were ground-truthed in September 2018 using the African Union's SLEEK project random sampling methodology and identifying 25 sites along roads (African Union, 2016).

The map of landcover cover current as of January 2018 was then used as the Baseline and transformed into five maps of potential future landcover, one for each future land-use option (Figure 2.2). This was based on a set of landcover change rules defined within the future land-use option narratives based on the grounded theory analysis and then validated by the interviewees. ‘*Business as Usual*’ was produced to capture the continuation of planned harvesting and replanting of exotic timber plantations, reforestation and agricultural expansion outside of protected areas in previously cleared areas. '*Farming Future'* involved all secondary habitats and cultivation within mixed-use areas of the PAs and outside PAs becoming annual crops. *'Forestry Future'* involved all secondary habitats and cultivation within mixed-use areas of the PAs and outside PAs becoming exotic timber plantations. *'Green Future'* was produced to capture all secondary habitats and cultivation within Mount Kenya's PAs being reforested and 300 metre wide grassland elephant corridors established between protected areas. '*Blue Future’* captured specific policy recommendations, including the reforestation of exotic timber plantations more than 500 metres inside the PA boundary, the reforestation of riparian reserves and the relocation of exotic timber plantations outside of Mount Kenya's PAs (Government of Kenya, 2018a).

S2.2 Expert opinion on landcover resistance values

	1 = no movement - 100 = free movement				
	Lower bound estimate score for the likelihood of elephant movement	Upper bound estimate score for the likelihood of elephant movement	Best estimate score for the likelihood of elephant movement	What is the probability between 50% (even chance) and 100% (certainty) that the best estimate will fall within the bounds?	
PA landcover (hover for definitions)	***Lower bound***	***Upper bound***	***Best estimate***	***Confidence***	***Comments (hover for Round 1 comments)***
Bamboo *Podo* forest	30	50	45	60%	
Bare earth	5	40	30	70%	
Bushland	70	100	90	70%	
Cultivated (PELIS/TELIS)	30	60	50	50%	
Cultivated (tea)	20	40	30	30%	
Grassland	60	80	70	60%	
Lower montane forest	70	100	90	80%	
Moorland	40	70	60	70%	
Plantation (exotic)	40	60	50	40%	
Plantation (indigenous)	60	80	70	40%	
Secondary thicket	70	100	80	70%	
Upper montane forest	60	80	70	70%	
Woodland	80	100	90	70%	



Back to PA landcover definitions list

Click here to proceed to DA landcover definitions

Figure S2.1: an example of the data collection process used for expert elicitation. This was from the second round of the survey where the experts were asked to review the scores they gave using a four-point structured question (Speirs-Bridge et al., 2010) against those given by other experts in an application of the Delphi Technique (Linstone & Turoff, 1975).

Experts scored the permeability of each landcover type to elephant movements using a four-point structured question (see main text Section 2.3.3). We used the Delphi Technique to increase the accuracy of expert opinion (Linstone & Turoff, 1975; Mukherjee et al., 2015) (Figure S2.1). The overall standard deviation (SD) from the mean reduced for both within PA scores (first round SD = 21.46 and second round SD = 15.73) and between PA scores (first round SD = 21.26 and second round SD = 16.60) (Table 2.1; Figure S2.2)

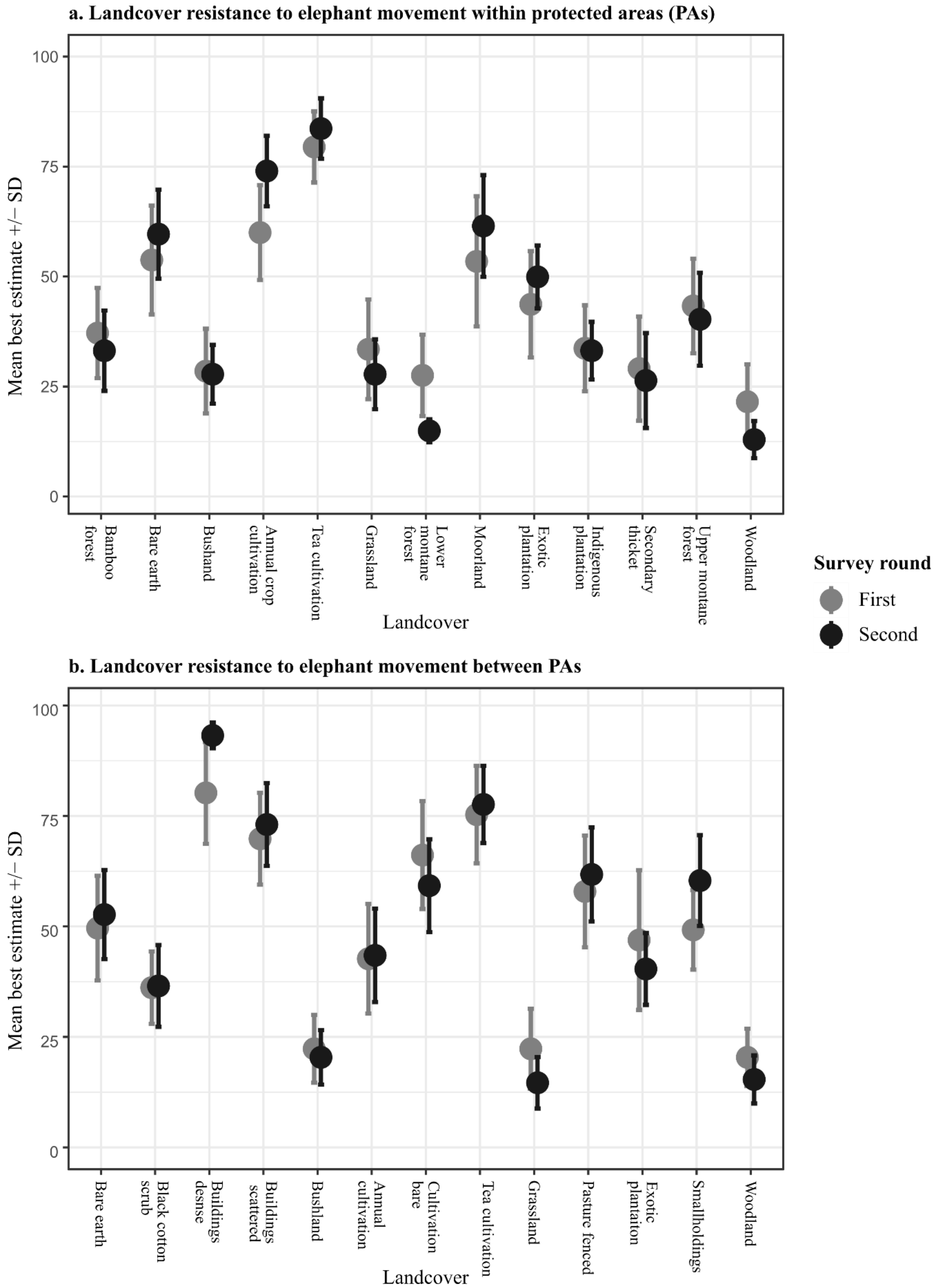


Figure S2.2: Mean best estimate landcover connectivity scores +/- standard deviation elicited as the likelihood of elephant movement across a 10 metre pixel for (a) movements within protected areas (PAs) and (b) between PAs. The two rounds show the difference in opinion between the first round and second round of our expert elicitation survey where we used a four-point structured question (Speirs-Bridge et al., 2010) and the Delphi Technique (Linstone & Turoff, 1975).

S2.3 Resistance surface

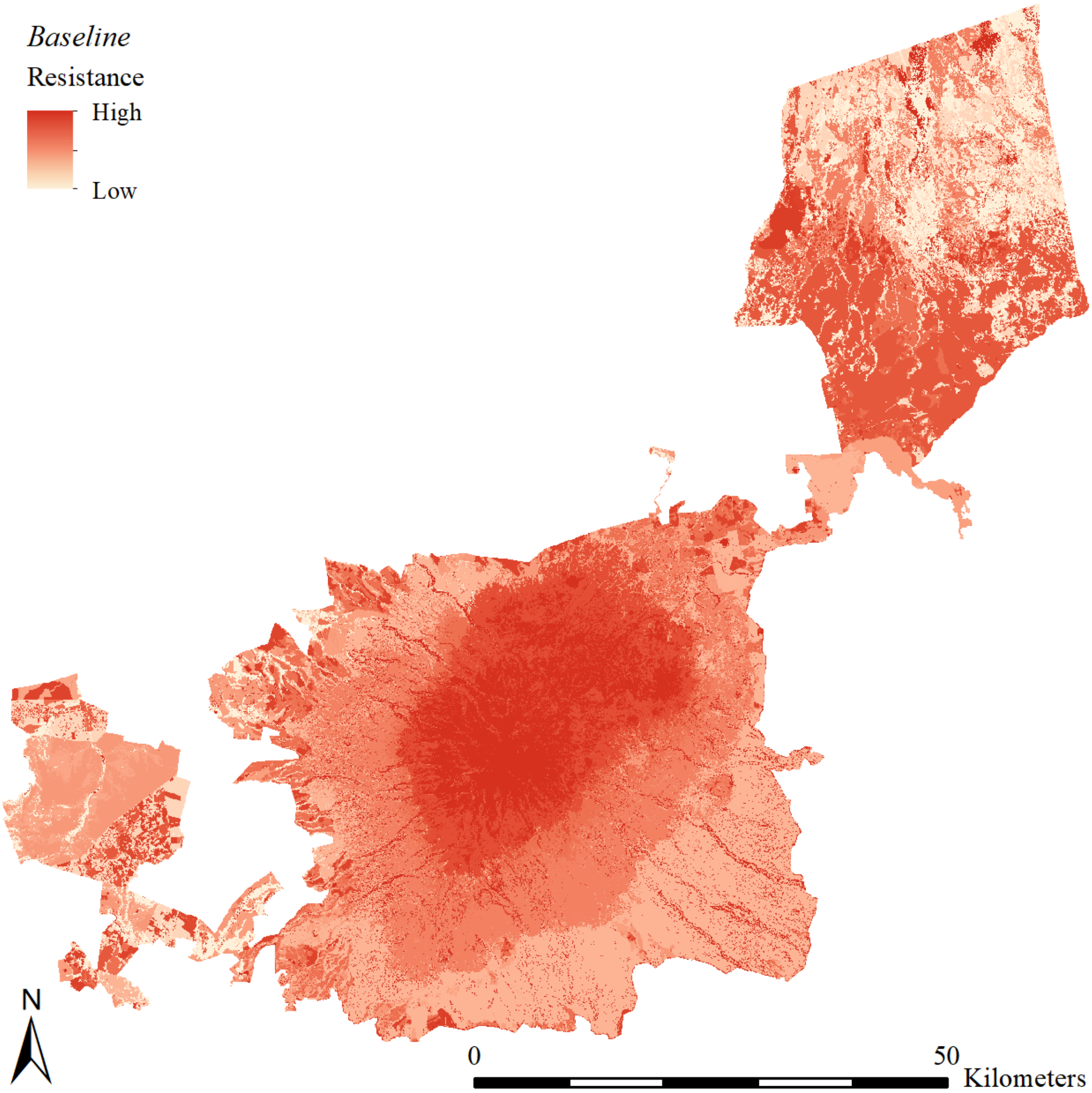


Figure S2.2: The combined resistance surface for with and between protected area connectivity for the *Baseline* scenario, which represented landcover in January 2019

S2.4 Linkage values

Table S2.1: Linkage values for the nine linkages and percentage change between the *Baseline* and the five future land-use options for: linkage quality (the ratio of cost weight distance to Euclidean distance), where a lower value is better quality; optimal path resistance (the ratio of cost weight distance to least-cost path length), where a higher value shows more resistance; alternative pathways (ratio of cost weight distance to effective resistance), where a higher value shows more additional pathways; and Linkage centrality the value of the linkage in keeping the network connected.

Land-use option	Linkage quality	% change	Optimal path resistance	% change	Alternative pathway availability	% change	Linkage centrality
MK Forest to Imenti:							
Baseline	7.39	-	4.64	-	86.75	-	16
Business As Usual	7.62	3%	4.74	2%	74.13	-15%	-
Farming Future	9.07	23%	5.66	22%	104.63	21%	-
Forestry Future	8.34	13%	5.20	12%	85.36	-2%	-
Green Future	3.91	-47%	3.07	-34%	272.45	214%	-
Blue Future	4.24	-43%	3.07	-34%	181.42	109%	-
MK Forest to Lewa:							
Baseline	13.90	-	7.91	-	23.41	-	9
Business As Usual	14.67	6%	8.66	9%	25.94	11%	-
Farming Future	19.63	41%	11.02	39%	31.40	34%	-
Forestry Future	16.63	20%	9.34	18%	27.74	19%	-
Green Future	10.89	-22%	6.26	-21%	19.26	-18%	-
Blue Future	11.69	-16%	6.26	-21%	20.41	-13%	-
MK Forest to Thego:							
Baseline	5.79	-	3.09	-	72.82	-	24
BAU	7.93	37%	4.46	44%	121.41	67%	-
Farming Future	10.80	86%	5.78	87%	48.00	-34%	-
Forestry Future	8.11	40%	4.34	41%	59.09	-19%	-
Green Future	3.14	-46%	2.49	-19%	401.88	452%	-
Blue Future	3.80	-34%	2.85	-8%	216.23	197%	-
Imenti to Samburu:							
Baseline	7.14	-	4.89	-	49.95	-	9
Business As Usual	7.58	6%	5.21	7%	59.26	19%	-
Farming Future	7.87	10%	5.48	12%	42.86	-14%	-
Forestry Future	6.34	-11%	4.40	-10%	95.57	91%	-
Green Future	1.55	-78%	1.23	-75%	171.56	243%	-
Blue Future	3.44	-52%	2.43	-50%	51.66	3%	-
Thego to Sangare:							
Baseline	3.05	-	2.21	-	147.20	-	25
Business As Usual	3.23	6%	2.36	7%	146.52	0%	-
Farming Future	11.04	262%	7.72	249%	37.89	-74%	-
Forestry Future	9.56	214%	6.78	207%	67.49	-54%	-
Green Future	1.12	-63%	1.00	-55%	165.53	12%	-

Blue Future	8.57	181%	4.32	96%	60.47	-59%	-
Sangare to Solio:							
Baseline	2.18	-	1.78	-	215.53	-	16
Business As Usual	2.04	-6%	1.65	-7%	194.96	-10%	-
Farming Future	49.97	2190%	46.45	2509%	232.78	8%	-
Forestry Future	28.85	1222%	26.82	1407%	346.58	61%	-
Green Future	1.08	-51%	1.00	-44%	178.33	-17%	-
Blue Future	10.53	382%	2.64	49%	64.88	-70%	-
Sangare to Nyeri Forest:							
Baseline	10.53	-	6.99	-	91.18	-	16
Business As Usual	10.86	3%	7.04	1%	96.63	6%	-
Farming Future	20.61	96%	15.42	121%	104.66	15%	-
Forestry Future	13.05	24%	9.92	42%	133.61	47%	-
Green Future	1.18	-89%	1.00	-86%	96.26	6%	-
Blue Future	6.93	-34%	4.99	-29%	80.74	-11%	-
Solio to Ol Pejeta:							
Baseline	1.60	-	1.37	-	404.23	-	9
Business As Usual	1.76	10%	1.32	-3%	416.20	3%	-
Farming Future	3.51	119%	2.14	56%	50.18	-88%	-
Forestry Future	3.44	115%	2.09	53%	75.59	-81%	-
Green Future	1.29	-20%	1.00	-27%	455.13	13%	-
Blue Future	3.22	101%	2.03	49%	78.86	-80%	-
Nyeri Forest to Aberdare:							
Baseline	3.49	-	2.81	-	75.23	-	9
Business As Usual	3.73	7%	2.94	5%	71.17	-5%	-
Farming Future	5.24	50%	3.52	25%	39.21	-48%	-
Forestry Future	4.50	29%	3.02	7%	66.18	-12%	-
Green Future	1.13	-68%	1.00	-64%	109.15	45%	-
Blue Future	4.51	29%	3.11	11%	70.12	-7%	-

Chapter 3 Structured decision-making shows broad support from diverse stakeholders for habitat conservation and restoration

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3.1 Abstract

There is growing recognition of the need to integrate biodiversity conservation with ecosystem restoration to benefit the planet and people. However, decision-making processes should first acknowledge the perspectives of the local communities and other stakeholders living with priority landscapes. This can recognise the trade-offs inherent within different choices to best foster local support for and stewardship of conservation and restoration projects. We developed a structured decision-making process to understand stakeholder values and preferences for future land-use options following a moratorium on forest resource extraction in Kenya's Central Highlands. We used semi-structured interviews with stakeholders to define the decision-making context including the study area limits. We posed the question: "*what future land-use options are feasible for the study region, which is most preferable, how does this vary between different stakeholder groups, and what values drive these preferences?*" We then mapped the options and assessed their performance using a geographical information system. We next engaged stakeholders (N = 51) to value the importance of each ecosystem service benefit using two decision models, initially from their individual perspectives and then through consensus building representing their shared values as a stakeholder group (N = 6). Stakeholders held significantly different values for provisioning, cultural and regulation and maintenance services; particularly those relating to traditional uses, biodiversity and water. However, when aggregating ecosystem service performance with stakeholder importance within and across stakeholder groups, they predominantly preferred the land-use option of habitat conservation and restoration. One stakeholder group and some individuals instead preferred the option of increased plantation forestry. We discuss how this shows structured decision-making as a valuable process for engaging local communities and building stewardship of habitat conservation and restoration projects and examine its implications for environmental planning in the broader region.

Keywords: Multi criteria decision analysis, ecosystem services, local communities, habitat restoration, landscape connectivity, Kenya.

3.2 Introduction

Human impacts on the natural environment are profound (Ruckelshaus et al., 2020; Steffen et al., 2015) and increasing worldwide (Ellis *et al.*, 2021; Williams *et al.*, 2020), with anthropogenic activities driving contemporary biodiversity loss (Green et al., 2020; Pimm et al., 2014) and eroding ecosystem functioning (Cardinale et al., 2012). While maintaining and protecting intact habitats is vital for biodiversity conservation and ecosystem service delivery (Allan et al., 2020; Dinerstein et al., 2020), the need for targeted restoration in areas where ecosystem integrity has become compromised is also becoming more widely recognised (Chazdon & Brancalion, 2019; Leclère et al., 2020). Indeed, the Convention on Biological Diversity's 'Post 2020 Zero Draft' outlines global agreement on future strategies (Convention on Biological Diversity, 2020), emphasising the need "to put biodiversity on a path to recovery" before 2030. This imperative is further augmented by the 'United Nations Decade on Ecosystem Restoration' (UNEP, 2021).

Integrated conservation and restoration planning can only be achieved successfully by engaging with local communities living within target landscapes (Chazdon, 2019; Erbaugh & Oldekop, 2018; Wiens & Hobbs, 2015). However, these local communities are often politically and geographically marginalised (Brockington et al., 2006; Neumann, 2004), and restoration must not come at the expense of their livelihoods (Bond et al., 2019; Pritchard & Brockington, 2019). Local community requirements and views should be accounted for in decision-making, as well as the standpoints of corporate, public and other actors (Chazdon & Brancalion, 2019; Smith et al., 2009). This type of transparent and equitable land-use decision-making should benefit those who will be most affected by the outcome (Guerrero et al., 2018). By ensuring local community support for, and stewardship of, conservation and restoration activities, such projects are more likely to be effective in perpetuity (Bennett et al., 2018; Maxwell et al., 2020; Pretty & Smith, 2004).

Encouraging stewardship of conservation and restoration projects is especially important in regions such as Sub-Saharan Africa, where a large proportion of extant terrestrial megafauna reside (Barlow et al., 2018; Ripple et al., 2016). These species need large, connected landscapes to support seasonal movements. However, they exist within rapidly transforming human-modified landscapes (Ellis et al., 2021), where structural connectivity between protected areas is particularly constrained (Saura et al., 2018; Ward et al., 2020). Additionally, Sub-Saharan protected areas are often ineffective. This is due, in part, to underfunding (Bruner et al., 2004; Lindsey et al., 2018; Coad et al., 2019; Lindsey et al., 2020;). However, in some locations, protectionist, exclusionary and militarised practices against local communities have fostered resentment and antipathy (Duffy et al., 2019; Oldekop et al., 2016), increasing the likelihood of conflict and failed implementation (Waylen et al., 2010; Zafra-Calvo & Geldmann, 2020). Structured decision-making allows for careful and organised analysis of problems, providing an audit trial of how a decision was made (Clemen & Reilly, 2001). It is appropriate for engaging diverse stakeholders in conservation and restoration planning because it does so through a transparent and defensible process of identifying and evaluating the values that underlie the decisions (Esmail & Geneletti, 2018). The recognition of distinct perspectives also highlights the trade-offs inherent within choices, facilitating decision-making that balances conservation and restoration with other social and economic goals (Adams et al., 2016). Moreover, it often helps to garner support from local communities by being explicit about the benefits they should receive from conservation and restoration plans (Brooks et al., 2012; Mustajoki et al., 2020).

Here, we use structured decision-making to understand the complex trade-offs between future environmental outcomes and how these vary between different stakeholder groups within Kenya's Central Highlands, where ecosystem integrity is under threat (Government of Kenya, 2017). The study region is a complex socioecological system driven by diverse land-use, ethnicity, industry and habitats (Kiteme et al., 2008). An estimated 500,000 people live within the study region, with 5.2 million inhabiting the wider seven counties (Rose et al., 2019). We pose the question: "*what future land-use options are feasible for the study region, which is most preferable, how does this vary between different stakeholder groups, and what values drive these preferences?*" We evaluate future land-use options, following a national moratorium on timber harvesting within state-run protected areas that was enacted after illegal clearing occurred within Mount Kenya's protected areas (Government of Kenya, 2018a). This policy follows on from two previous moratoria (Emerton, 1999; Vanleeuwe, 2004), highlighting that the underlying land-use governance issues were not properly addressed in the past, and that equitable and transparent decision-making would be constructive moving forwards. All stakeholder groups were represented at each step in the structured decision-making process, assessing and quantifying preferences for the ecosystem service benefits the groups are likely to receive from the range of future land-use options. The process recognised a multiplicity of diverse stakeholder perspectives and highlighted which outcomes were grounded in shared values.

3.3 Methods and results

3.3.1 Study Region

We used Kenya's Central Highlands as our study region. The region centres on the protected area complex of Mount Kenya, which is designated as a World Heritage Site due to its high biodiversity, cultural and aesthetic values. The protected areas contain a variety of habitat types, including Afromontane forests and Afro-alpine moorlands, that provide habitat for species of conservation concern, such as an estimated population of over 2,000 forest-dwelling African bush elephants *Loxodonta africana* (Vanleeuwe & Gitau, 2020) and local/Afromontane endemics (Musila et al., 2019; Riggio et al., 2019). Much of the original extent of these forests have been degraded or converted to agriculture or silviculture (Emerton, 1999; Vanleeuwe, 2004), as has the surrounding forest-savannah mosaic (Kiteme et al., 2008). Nonetheless, the matrix contains fragmented pockets of higher quality natural habitats within additional state-run protected areas and 'wildlife conservancies'. Wildlife conservancies are land parcels that are owned and managed by private landowners or a community for wildlife conservation and other compatible land-uses that improve livelihoods, such as ecotourism (Government of Kenya, 2018b).

The region supports local livelihoods through domestic and international tourism (Steinicke & Neuburger, 2012), which provides 9% of the national gross domestic product and brought over one million tourists to Kenya annually before the COVID-19 pandemic (Turner, 2017). It also provides other ecosystem services, notably as a water catchment, regulating water flows for domestic, agricultural and industrial uses for a third of Kenya's people across a half of the country's land surface (Gathaara et al., 1999). Additional livelihood benefits are provided from a diverse agricultural and silvicultural industry, which employs over half the labour force nationally and contributes to over a fifth of gross domestic product (Ulrich et al., 2012; Zaehringer et al., 2018).

3.3.2 Structured decision-making framework

We engaged with stakeholders, including local communities, at all stages of the structured decision-making process (Figure 3.1), whether they had power to shape the problem we were addressing or were otherwise affected by it. We first carried out stakeholder interviews to set the decision-making context. Subsequently, we completed a multi criteria decision analysis (MCDA), which is a structured decision-making tool that can systematically and transparently examine stakeholder values and preferences for alternative decision options (Esmail & Geneletti, 2018). MCDAs are widely-applied in medicine (Adunlin, Diaby and Xiao, 2015), forest management (Uhde et al., 2015) and wider environmental decision science (Cegan et al., 2017; Favretto et al., 2016; Langemeyer et al., 2016).

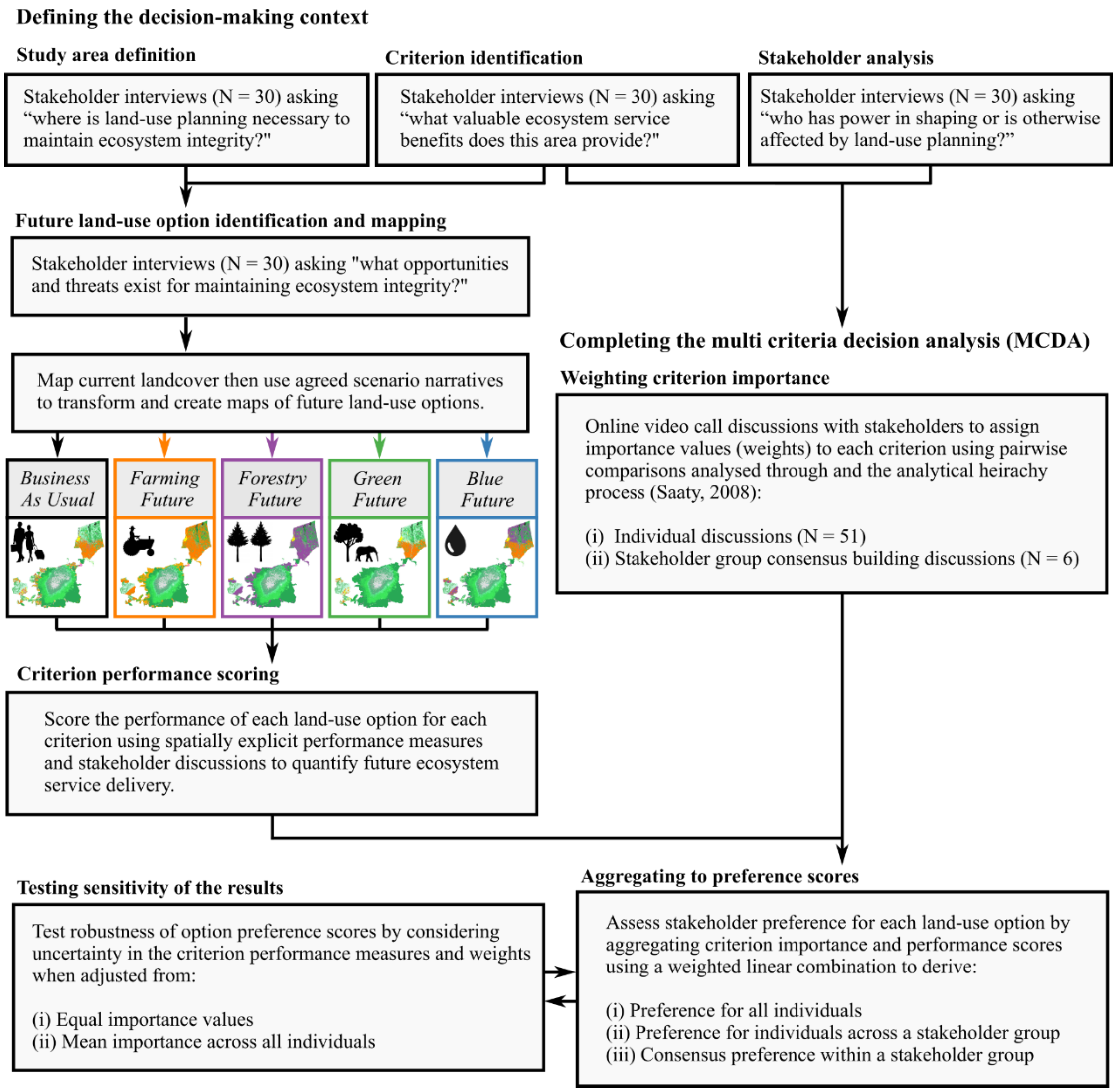

Figure 3.1: The methodological steps used in the structured decision-making process, from defining the decision-making context to completing the multi-criteria decision analysis (MCDA). The overarching question posed to the stakeholders was: "*what future land-use options are feasible for the study region, which is most preferable, how does this vary between different stakeholder groups, and what values drive these preferences?*"

3.3.2.1 Defining the decision-making context

We used semi-structured interviews with stakeholders to determine the decision-making context for the MCDA (Figure 3.1). We used a mixed approach to recruiting stakeholders to reduce the biases arising from the use of a single method. First, we used snowball sampling, where individuals interviewed suggest further potential interviewees. Initially, we targeted five local community-based conservation NGO staff and two protected area managers from the government wildlife agency. Snowball sampling is most effective at accessing hard-to-reach groups, but might over represent similar individuals and perspectives (Sadler et al., 2010). Therefore, in addition to snowball sampling, we iteratively used the outputs of the stakeholder analysis to identify new stakeholders, including local community representatives. Consequently, we were also able to reach farmers, foresters, water resource managers, researchers as well as national and county government officials from ministries/agencies responsible for land-use and environmental planning. We stopped conducting interviews (N = 30) when responses reached saturation and no new information was reported across all respondents.

During the interviews, we asked each individual a series of open questions, followed by verbal prompts, to: (i) verify that they understood and agreed with the question being posed; (ii) define the extent of the study area and its boundaries to be considered in the MCDA; (iii) identify the ecosystem service benefits relevant as ‘criteria’, which are the objectives of the decision-making process (Esmail & Geneletti, 2018); (iv) ascertain the stakeholders within the study area; and, (v) identify and map distinct future land-use options. We coded the interview transcripts using NVivo 12 software (QSR International, 2018) and analysed them using grounded theory (Charmaz & Belgrave, 2015). After all the interview transcripts had been analysed, we shared initial findings with interviewees for feedback and validation. This was done to ensure that they unanimously agreed on the decision-making context within which the MCDA would occur.

Kenya’s Central Highlands are a fragmented landscape (Didier et al., 2011; Kiteme et al., 2008). Retaining and restoring landscape connectivity, using elephants as a proxy for broader biodiversity, is a major focus of land-use planning (Bastille-Rousseau & Wittemyer, 2020; Evans & Adams, 2016; Green et al., 2018; Ihwagi et al., 2019). We therefore asked the interviewees to decide on the extent of the study area and its boundaries within the region, based on where they felt it was necessary to maintain ecosystem integrity using elephants movements as a proxy for wider species movements. They felt that the study area should cover where elephants still move within and between protected areas and/or where interventions to preserve or restore habitats are being considered. They delineated the area as the protected area complex of Mount Kenya, plus sections of the surrounding matrix of communally- and privately-owned lands that connect with the contiguous protected areas of Aberdare, Laikipia and Samburu (Figure 3.2).

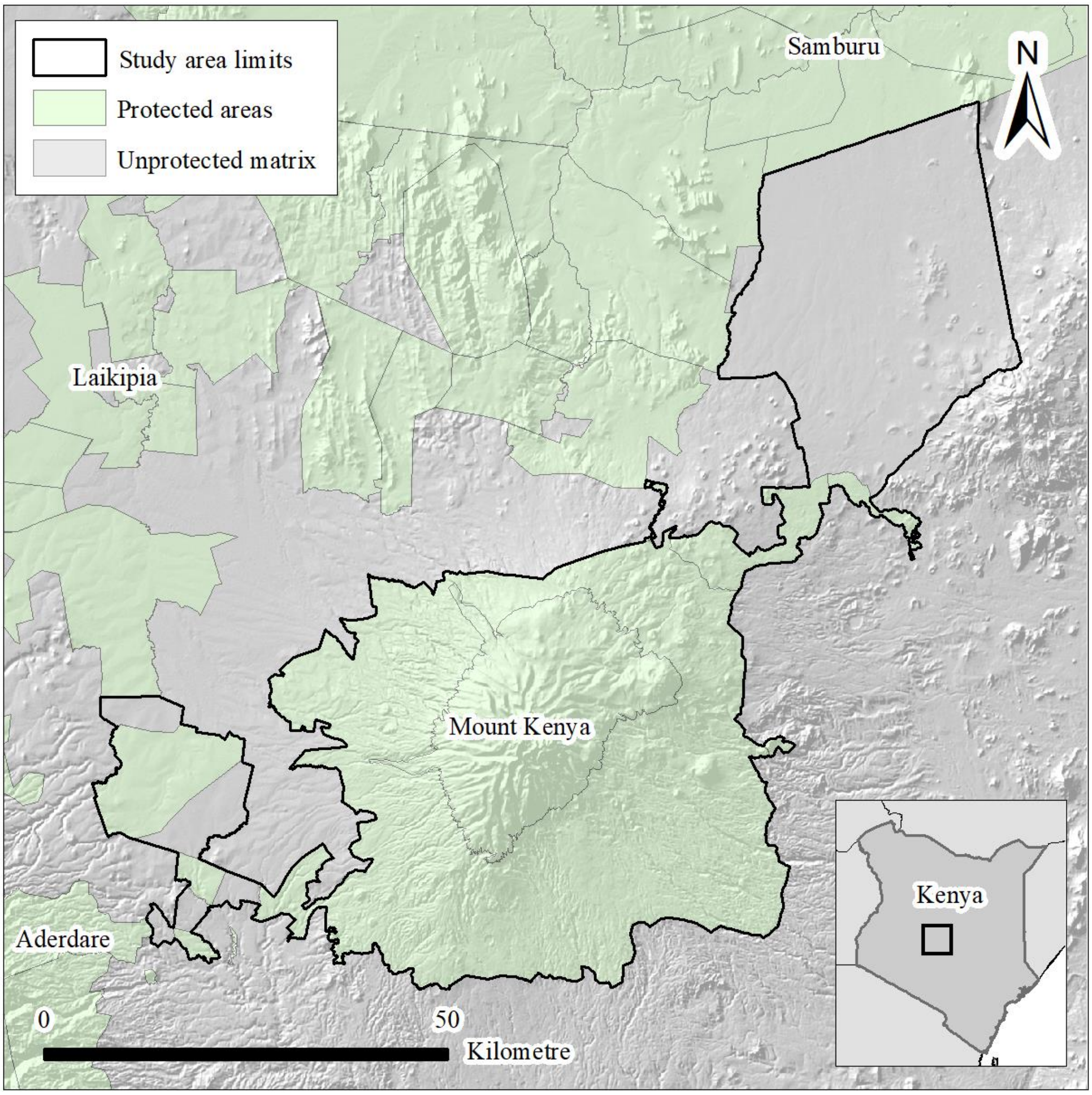

Figure 3.2: The stakeholder-defined study area (outlined in black) for the multi-criteria decision analysis (MCDA). The area comprised the Mount Kenya protected areas and the sections of the surrounding matrix that connect with the contiguous protected areas of Aberdare, Laikipia and Samburu. The inset map shows the location of the study area within Kenya.

We engaged the interviewees in a discussion about the ecosystem service benefits and values they derive from within the study area. They were prompted to talk through what was important to them as individuals and the communities within which they work and live. The responses were used to identify the criteria for our MCDA. These were honed through stakeholder feedback to ensure they comprehensively captured the diverse values that stakeholders carry, with a particular focus on ensuring that the values that local communities hold were fully represented (Mustajoki et al., 2020). The criteria were named using terminology that was understandable across our interviewees, but were grouped according to the Common International Classification of Ecosystem Services (CICES) (Haines-Young & Potschin-Young, 2018). In total, the interviewees identified 13 criteria, consisting of six provisioning, three cultural and four regulation and maintenance ecosystem services (Table 3.1).

Table 3.1: A list of our stakeholder-derived multi-criteria decision analysis (MDCA) 'criteria', which were the ecosystem service benefits that formed the objectives of our structured decision-making process (Esmail & Geneletti, 2018). The Common International Classification of Ecosystem Services (CICES) code (Haines-Young & Potschin-Young, 2018) for each criterion is provided with the 15 performance measures agreed to score the criterion under each land-use option.

Criterion (CICES code)	Performance measure scoring approach
Cash crop production (1.1.1.2)	The hectares of mapped area under (i) perennial crops and (ii) exotic plantations during their cash crop cultivation phase
Livestock grazing (1.1.3.1)	The hectares of (iii) exotic plantation and (iv) secondary grassland
Subsistence crop production (1.1.1.1)	The hectares of (v) annual crops
Traditional medicines (1.1.5.2)	The hectares (vi) primary vegetation
Water provision (1.3.X.X)	The Co$ting Nature measure of (vii) realised water provision indexed globally (Mulligan et al., 2010)
Wood for fuel or construction (1.1.5.3)	The hectares of (viii) exotic plantations
Cultural heritage (3.1.2.3)	The hectares of (ix) primary vegetation
Outdoor recreation (3.1.1.1)	The hectares of vegetation given an ordinal multiplier by participants of 1 for (x) primary vegetation and 0.2 for (xi) secondary vegetation
Benefits from tourism (3.1.1.2)	The hectares of vegetation given an ordinal multiplier by participants of 1 for (xvi) primary vegetation and 0.2 for (xvii) secondary vegetation
Biodiversity conservation (2.2.2.1)	The hectares of (xviii) primary habitats and (xix) Linkage Mapper's additional pathway availability (McRae & Kavanagh, 2019), a measure of landscape connectivity
Climate change mitigation (2.2.6.1)	The sum of the Co$tingNature measures of (xii) forest carbon storage and (xiii) forest carbon sequestration (Mulligan et al., 2010)
Soil erosion prevention (2.2.1.1)	The inverse of the WaterWorld measure of (xiv) runoff (Mulligan, 2013)
Water flow regulation (2.2.1.3)	The inverse of the WaterWorld measure of (xv) hillslope net erosion (Mulligan, 2013)

We asked interviewees to identify all pertinent stakeholders within the study area. They identified 36 stakeholder organisations (see S3.1 in Supplementary Materials), which were subsequently placed into 13 categories according to their operational remit. They were then classified into six distinct stakeholder groups: (i) large scale agricultural and horticulture operators (hereafter '*Big Farms*'); (ii) conservation practitioners and ecotourism operators ('*Conservationists*'); (iii) county government officials ('*Counties*'); (iv) forest resource users ('*Forest Users*'); (v) downstream pastoral communities ('*Pastoralists*'); and, (vi) proximate smallholder farmers ('*Smallholders*'). This guided participant recruitment for the remainder of the MCDA, with each stakeholder group requiring at least five members to ensure they were adequately represented.

We asked interviewees questions about current land-uses in the study area, future land-uses, and the opportunities and threats for maintaining ecosystem integrity. In doing so, we could then identify and map future land-use options as the alternative choices to be evaluated through our MCDA (Esmail & Geneletti, 2018). The discussions were informed and bounded by policy constraints to ensure they captured possible futures for the year 2030 (Government of Kenya, 2007, 2018a). Interviewees responded with 40% identifying agricultural land-uses (n =12), 37% silvicultural land-uses (n =11), 43% biodiversity conservation and landscape connectivity land-uses (n =13), and 40% water resource conservation land-uses (n =12). After all the interviews were completed and analysed, we used these themes to represent five land-use options in narrative form and presented them to interviewees to ensure they captured distinct, divergent and plausible future land-use scenarios (Mckenzie et al., 2012b; Peterson et al., 2003). The five land-use options were:

(i) *'Business As Usual',* which involved the planned harvesting and replanting of exotic timber plantations, plus agricultural expansion outside of protected areas in areas where clearing had already occurred.

(ii) '*Farming Future',* where all cultivation and secondary habitats within mixed-use areas of Mount Kenya's protected areas and outside protected areas were converted to annual crops.

(iii) *'Forestry Future',* where all cultivation and secondary habitats within mixed-use areas of Mount Kenya's protected areas and outside protected areas were converted to exotic timber plantations.

(iv) *'Green Future',* where all cultivation and secondary habitats within the protected areas of Mount Kenya were reforested and 300 metre wide grassland elephant corridors were established between protected areas.

(v) ‘*Blue Future’,* which captured specific policy recommendations including reforestation of protected area exotic timber plantations more than 500 metres from the boundary, the reforestation of riparian reserves and the relocation of exotic timber plantations outside of Mount Kenya's protected areas (Government of Kenya, 2018a).

We next mapped these future land-use options to quantify how they would affect ecosystem service delivery. We first mapped current landcover within the study area using a combination of remote sensing and digitising (see S3.2), the results from which were ground-truthed using the African Union’s SLEEK project random sampling methodology (African Union, 2016). We then used the landcover changes detailed in the narratives to transform current landcover into maps for each of the five land-use options (Figure 3.3; Section S3.2), using R (R Core Team, 2019) and the package ‘raster’ (Hijmans, 2019).

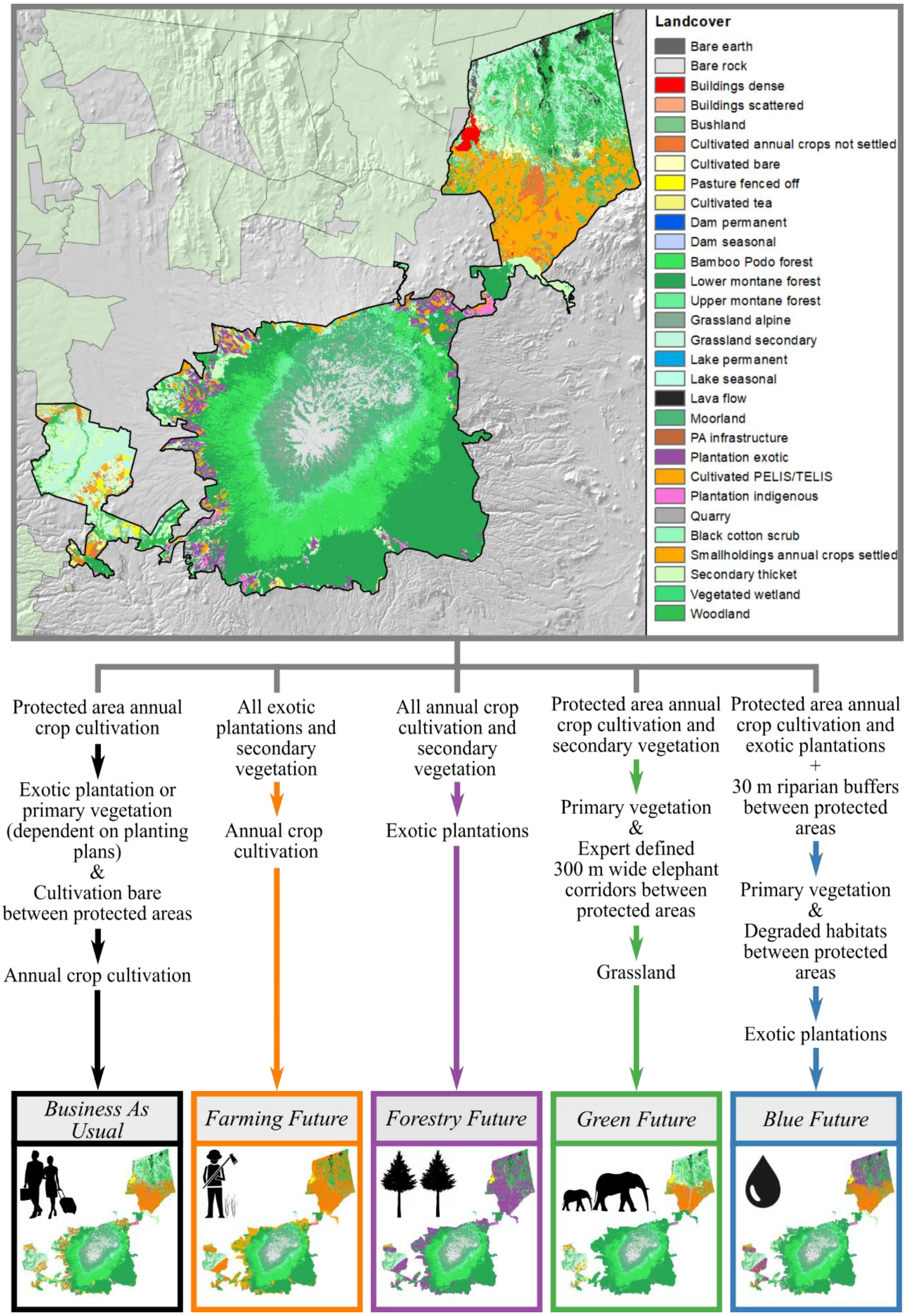

Figure 3.3: A representation of how current landcover, correct as of January 2018 (grey box), was transformed into our five future land-use options to be used in the multi-criteria decision analysis (MCDA): (i) *Business As Usual* (black); (ii) *Farming Future* (orange); (iii) *Forestry Future* (purple); (iv) *Green Future* (green); and, (v) *Blue Future* (blue). Landcover changes only occurred in mixed-used areas of Mount Kenya's protected areas and in areas that are not formally protected.

3.3.2.2 Completing the multi criteria decision analysis (MCDA)

Having used the semi-structured interviews to define the decision-making context, we completed the MCDA process through individual stakeholder online video call discussions (N = 51) and, subsequently, stakeholder group online video call discussions (N = 6). All of these were facilitated by GG and began with a presentation to familiarise the participants with the study purpose, process, and the decision-making context agreed through the interviews and follow up interactions. During the individual stakeholder discussions, participants assigned relative importance weights to each criterion (hereafter 'importance'). They were also asked to place themselves in the stakeholder groups they identified with: *Big Farms* (n = 7); *Conservationists* (n = 14); *Counties* (n = 5); *Forest Users* (n = 6); *Pastoralists* (n = 7); and, *Smallholders* (n = 12). The stakeholders were then invited to join the consensus building stakeholder group discussions to reweight criterion importance from a shared perspective, with a quorum of four participants. These individual and group discussions also allowed the stakeholders to provide feedback on the methodology we proposed to score the 'performance' of each future land-use option in terms of ecosystem service delivery. We subsequently undertook the performance scoring process using a geographic information system (GIS) and the maps for each land-use option. We then aggregated the importance weights of each criterion with their relevant performance scores for each future land-use option creating preference scores, hereafter 'preference'.

3.3.2.1.1 Weighting criterion importance

The relative importance of each criterion was weighted by stakeholders during the individual and group consensus building discussions. Two decision models were used to do this. In the first, the stakeholders used a ranking and relative weighting technique (Roszkowska, 2013). For this, criteria were placed in order of descending importance and then assigned a numerical importance value as a percentage. In the second, the stakeholders weighted each criterion using pairwise comparisons, where they signified the importance of each criterion relative to the others, attributing values from equal to extreme preference along a nine-point scale. This was analysed using the analytical hierarchy process (Saaty, 2008) and consistency ratios, a measure of the consistency of participant judgements when compared to random choices, tested using the R package 'ahpsurvey' (Cho, 2019).

During the weighting process, participants were regularly reminded that the importance values they gave should relate to future ecosystem service delivery in 2030. In the individual stakeholder discussions it was stressed that the 'importance' was from their personal perspective. Outputs from all individuals are termed 'importance across all individuals', whereas the importance individuals assigned from within particular stakeholder groups is termed 'importance for individuals across a stakeholder group'. Geometric mean values were calculated for both importance across all individuals and importance for individual values across a stakeholder group. We next carried out the consensus building stakeholder group discussions, where the emphasis was on capturing importance from a shared stakeholder group perspective, with outputs termed 'consensus importance within a stakeholder group'.

We checked for significant differences in importance values between the two decision models (see S3.3), using a multivariate analysis of variance (MANOVA) and post-hoc discriminant analysis using the R packages 'psych' (Revelle, 2020) and 'candisc' (Friendly & Fox, 2020). No significant differences in weights were apparent between the two decision models. Given the analytical hierarchy process has been found to be more robust and less prone to biases than the ranking and relative weighting approach (Németh et al., 2019), we only present the analyses based on the pairwise comparisons in the main text (a comparison of the weights can be found in S3.3). We also used MANOVA and post-hoc discriminant analyses to test for significant differences in criterion importance between stakeholder groups.

Mean importance across all individuals (N = 51) (Table 3.2; Figure 3.4) was highest for biodiversity conservation, followed by water flow regulation, climate change mitigation and water provision. These criteria also had the most variable importance across all individuals.

Table 3.2: Criterion importance (weights) from pairwise comparisons, analysed through the analytical hierarchy process (Saaty, 2008), emerging from the multi-criteria decision analysis (MCDA). Sample size (N and n), mean value (M) and standard error (SE) are shown for all participants (importance across all individuals), as well as individual stakeholders (importance for individuals across a stakeholder group) and consensus (consensus importance within a stakeholder group) for our six stakeholder groups. Asterisks * indicates a significant difference in criterion importance where p <0.05 and double asterisks ** where p <0.001 between stakeholder groups based on individual importance across a stakeholder group (MANOVA post-hoc discriminant analysis).

	All participants		***Big Farms***			***Conservationists***			***Counties***			***Forest Users***			***Pastoralists***			***Smallholders***		
	Individual N = 51		**Individual** n = 7		**Consensus**	**Individual** n = 14		**Consensus**	**Individual** n = 5		**Consensus**	**Individual** n = 6		**Consensus**	**Individual** n = 7		**Consensus**	**Individual** n = 12		**Consensus**
Criteria	**M**	**SE**	**M**	**SE**		**M**	**SE**		**M**	**SE**		**M**	**SE**		**M**	**SE**		**M**	**SE**	
Cash crop production	4.41	0.68	3.03	0.94	8.28	2.92	0.47	0.69	4.35	2.68	11.53	3.55	1.47	0.79	4.00	1.75	1.25	7.64	2.13	1.94
Livestock grazing	4.85	0.81	2.68**	0.85	6.19	2.80**	0.56	0.97	3.10**	0.84	6.56	2.63**	0.75	2.27	14.93**	3.97	10.56	4.47**	0.64	1.00
Subsistence crop production	4.61	0.52	3.13	1.13	2.84	3.36	0.75	2.67	4.10	1.23	5.83	4.22	0.98	1.51	4.57	1.29	2.51	7.36	1.41	2.52
Traditional medicines	2.23	0.30	1.47*	0.32	1.49	0.97*	0.15	0.32	1.42*	0.22	0.65	2.44	0.79	1.05	4.64*	1.32	2.13	2.99	0.64	0.54
Water provision	10.13	0.91	7.93	1.98	21.37	10.61	1.25	7.61	16.38	3.85	6.97	6.86	3.07	13.93	10.07	2.24	9.78	9.90	2.06	4.33
Wood	3.08	0.36	1.65	0.27	5.44	2.44	0.83	1.75	3.30	1.38	6.19	4.31	1.40	4.18	2.86	0.51	3.56	4.07	0.61	0.36
Cultural heritage	4.80	0.84	2.72	0.75	0.98	3.24	0.62	3.04	1.81	0.47	0.81	6.32	2.65	0.19	10.38	4.60	5.28	5.04	1.44	1.51
Outdoor recreation	5.49	0.81	5.67	1.70	2.94	4.23	1.11	15.22	3.11	0.89	4.05	1.92	0.50	1.71	5.20	0.84	2.64	9.80	2.66	1.51
Benefits from tourism	8.81	0.92	11.38	2.23	5.78	7.68	1.27	7.50	8.08	2.45	8.22	5.33	1.19	3.77	12.13	5.23	25.15	8.73	1.00	2.67
Biodiversity conservation	22.25	1.96	26.36	6.69	16.51	24.33	3.57	55.00	18.45	3.96	13.15	35.53	5.50	35.06	14.98	3.57	10.45	16.62	3.63	22.75
Climate change mitigation	10.33	1.43	6.38	2.37	3.94	13.29	3.07	0.45	12.41	7.06	26.66	12.62	6.29	27.65	5.52	1.39	6.67	9.96	1.97	43.74
Soil erosion prevention	7.54	0.77	12.42**	1.85	9.37	9.74	1.49	3.23	11.7**	2.44	3.38	3.57**	1.44	1.48	3.48**	1.19	6.67	4.77**	0.84	7.01
Water flow regulation	11.48	1.10	15.17	3.22	14.87	14.39	2.03	1.55	11.79	2.43	6.01	10.71	3.72	6.39	7.24	2.22	13.35	8.65	2.32	10.11

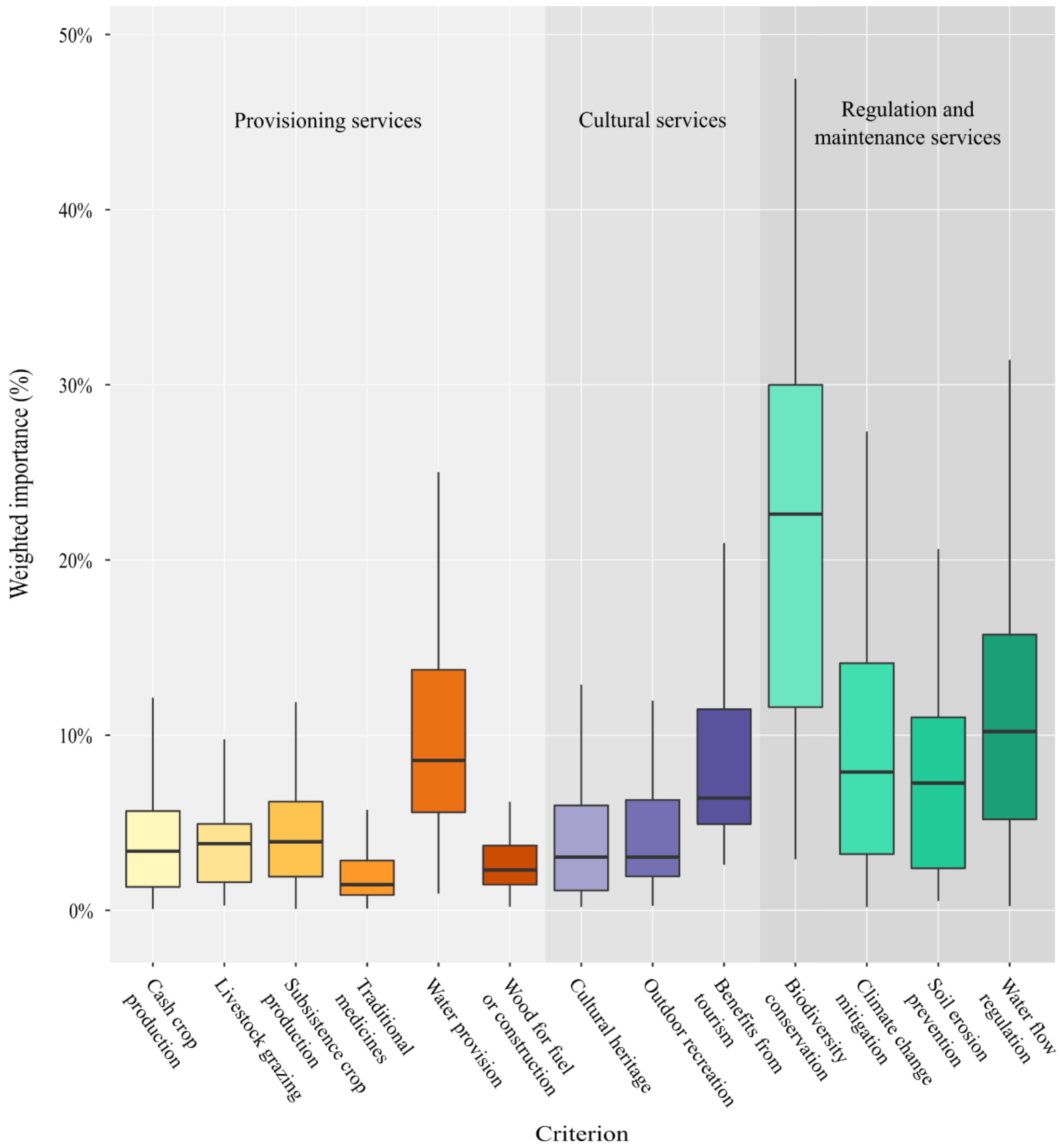

Figure 3.4: Box plots showing criterion importance (weights) across all individuals from pairwise comparisons, analysed through the analytical hierarchy process (Saaty, 2008), emerging from the multi-criteria decision analysis (MCDA). Coloured boxes show the standard deviation, with the central vertical line indicating the geometric mean, and the horizontal black line either side of the coloured boxes represents the range.

Importance for individuals across a stakeholder group highlighted differences between the stakeholder groups (Table 3.2; Figure 3.5). Livestock grazing was weighted as significantly more important by *Pastoralists* than other stakeholder groups ($p < 0.001$). Similarly, traditional medicines were weighted as significantly more important by *Pastoralists* than *Big Farms*, *Conservationists* and *Counties* ($p = 0.016$). Soil erosion prevention was weighted as significantly more important by *Big Farms* and *Counties* than by *Forest Users*, *Pastoralists* and *Smallholders* ($p < 0.001$).

Compared to mean importance for individuals across a stakeholder group, consensus importance within a stakeholder group (Table 3.2; Figure 3.5) saw *Big Farms* weight water provision as more than twofold more important and biodiversity conservation as less important. *Conservationists* weighted recreation threefold more important and biodiversity twofold. *Counties* weighted climate change mitigation as twofold more important. *Forest Users* weighted water provision as more important and gave over twofold the importance to climate change mitigation. *Pastoralists* weighted tourism as more important and provisioning services, cultural heritage, outdoor recreation and biodiversity conservation as less important. *Smallholders* gave a fourfold higher importance to climate change mitigation.

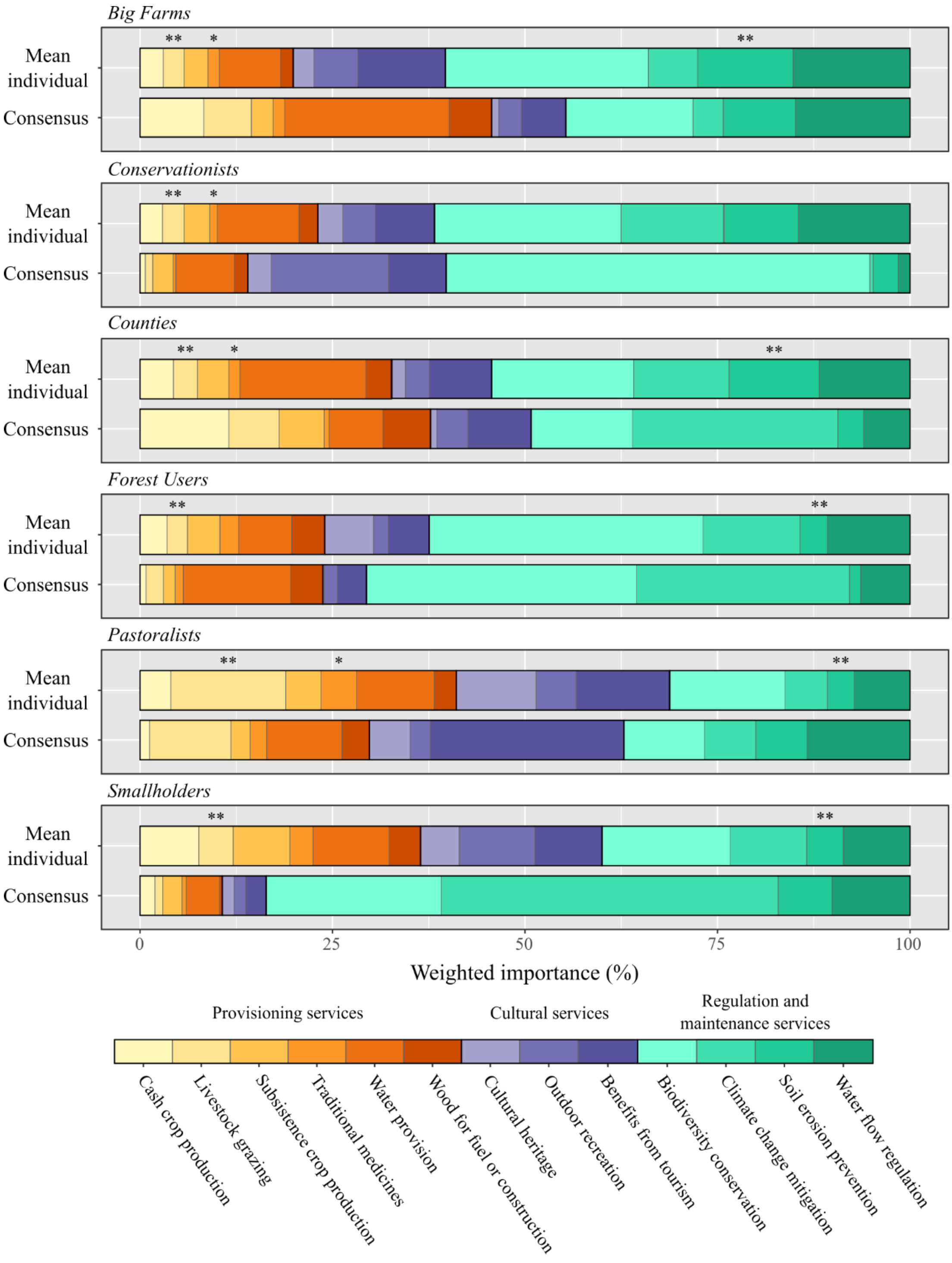

Figure 3.5: Horizontal mosaic bar plots showing the importance (weights) for mean individual (importance for individuals across a stakeholder group) above and the consensus (consensus importance within a stakeholder group) below from pairwise comparisons, analysed with the analytic hierarchy process (Saaty, 2008). Colours show the 13 criteria in the three overarching ecosystem service groups (outlined with black boxes). Asterisks indicates a significant difference in criterion importance where $p < 0.05$ and ** where $p < 0.001$ between stakeholder groups based on individual importance across a stakeholder group (MANOVA post-hoc discriminant analysis).

Once the weighting was complete, we analysed consistency ratios, a measure of the consistency of participant's judgements when compared to random choices, finding 19 out of 51 participants were below the standard 0.1 threshold for inclusion (Saaty, 2008). Excluding participants above the threshold did not change the preference rankings (please see Section 3.3.2.2.3 below), so we proceeded with all data. The consistency ratios for the stakeholder groups *Big Farms*, *Conservationists*, *Counties* and *Pastoralists* were below the same threshold.

3.3.2.1.2 Criterion performance scoring

We scored the performance of each land-use option for each criterion, using spatially explicit performance measures (Table 3.1) to quantify how each option affected ecosystem service delivery. The interviewees and stakeholders felt that the area of different landcovers was a suitable proxy for nine of the criteria. Landcovers for two criteria, outdoor recreation and tourism, were assigned multipliers agreed through stakeholder discussions to capture the relative value of primary vegetation compared to secondary vegetation, the latter being one fifth of the value of the former. Biodiversity conservation was scored using hectares of primary habitats and a measure of functional connectivity derived from the Linkage Mapper software (McRae & Kavanagh, 2019). The functional connectivity measure was informed by expert opinion on elephant movements across different landcover types and validated by empirical data (Space for Giants, unpublished data; Save the Elephants, unpublished data). For carbon and water provisioning services, we inputted the land-use option maps into Policy Support System's Co$tingNature (Mulligan et al., 2010), which generated measures of forest carbon storage and sequestration and realised water provision. Stakeholder discussions highlighted that exotic timber plantations over multi-decade time series may be a net producer of atmospheric carbon, as supported by the literature (Waller et al., 2020). To account for this, we classified exotic timber plantations as grasslands when modelling climate change mitigation (carbon services). For soil erosion and water flow control, we used Policy Support System's Waterworld (Mulligan, 2013) to produce measures of hillslope net erosion and runoff. Finally, raw criterion performance scores (see S3.4) were summed from their performance measures and normalised along a scale of 0-100, with 0 representing the worst level of performance and 100 the best, transforming input data to aid in comparability (GoUK, 2009).

Our land-use options showed pronounced differences in criterion performance (Figure 3.6). *Business As Usual*, by its nature as a counterfactual portraying what would have happened without the intervention of the moratorium (Government of Kenya, 2018a), had mid- to low-range performance. *Farming Future* performed highest for cash crop production and subsistence crop production, and lowest for all other criteria except climate change mitigation. *Forestry Future* performed highest for wood products, livestock grazing, water provision, water flow regulation, and soil erosion and low for other criteria, some of which had equal performance to *Farming Future*. *Green Future* performed highest for traditional medicines, cultural heritage, outdoor recreation, climate change mitigation, benefits from tourism and biodiversity conservation with mid- to low-range performance for other criteria. *Blue Future*, by result of it capturing specific spatial zoning recommendations (Government of Kenya, 2018a), performed with high- to mid-range scores across all criteria.

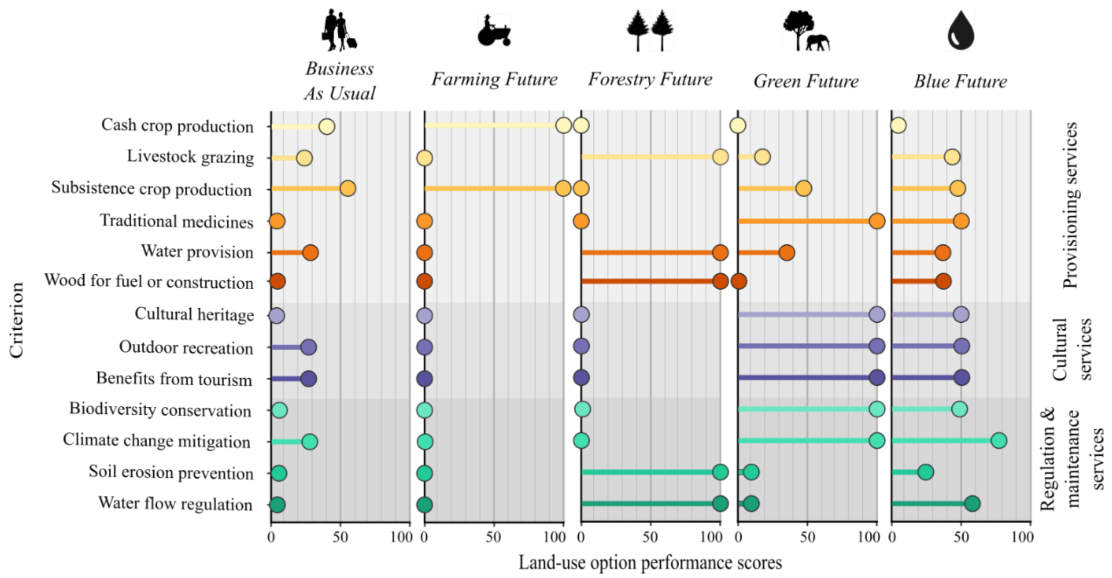

Figure 3.6: Horizontal lollipop chart showing performance scores, relative measures of future ecosystem service delivery, for each criterion for the five land-use options normalised on scale of 0 - 100 from worst to best performance (see S3.4 for raw scores).

3.3.2.1.3 Aggregating to preference scores

We assessed stakeholder preference for each land-use option, the decision outcome of the MCDA (Esmail & Geneletti, 2018), by aggregating importance weights with performance scores for each criterion: (i) 'preference for all individuals'; (ii) 'preference for individuals across a stakeholder group'; and, (iii) 'consensus preference within a stakeholder group'. To calculate these, we used a weighted linear combination approach:

$$S_i = \sum_{j=1}^{n} w_j s_{ij}$$

Here, the overall preference score for each land-use option (S_i) was calculated by summing the normalised criterion performance score ($_j$) as an option performance score (S_{ij}), which was then multiplied by the criterion weight (W_j) for all criteria (n).

Mean preference across all individuals (n = 51) was highest for *Green Future,* followed by *Blue Future* (Figure 3.7; Table 3.4). Preference was highest for *Green Future* for 42 individual stakeholders, with the remaining nine preferring *Forestry Future*. *Blue Future* was the second preference for 32 stakeholders.

Mean preference for individuals across a stakeholder group was also highest for *Green Future*, with the strongest preference amongst *Forest Users*, followed by *Smallholders, Pastoralists, Conservationists*, *Big Farms* and finally *Counties*. Here *Blue Future* was the second preference for all stakeholder groups, except for *Counties*. Mean preference for individuals across a stakeholder group was consistently lowest for *Farming Future*.

When compared to mean individual preference across a stakeholder group, consensus preference within a stakeholder group was stronger for *Green Future* across all stakeholder groups, with the exception of *Big Farms,* who had the highest preference for *Forestry Future.* Once again, *Business As Usual* and *Farming Future* ranked lowest for all stakeholder groups. Across all three levels of preference score comparison (mean preference across all individuals, mean preference for individuals across a stakeholder group and consensus preference within a stakeholder group) the greatest variability in scores was for *Forestry Future* followed by *Green Future*.

We carried out a sensitivity analysis to test the robustness of our option preference scores, by considering uncertainty in the criterion performance measures and weights and identify reversal points in land-use option preference rankings. This form of model evaluation is an important step in an MCDA as it provides an understanding of the values that have the biggest effect on the outcome (Delgado & Sendra, 2004; Esmail & Geneletti, 2018). We found the model was mildly sensitive to agricultural, silvicultural and hydrological ecosystem services, which caused switches to occur between *Green Future* and *Forestry Future* being of highest preference (see S3.5).

Table 3.3: Stakeholder preference scores for the five land-use options from pairwise comparisons, analysed with the analytic hierarchy process (Saaty, 2008). Values show sample size (N and n), mean value (M) and standard error (SE) for all participants (preference across all individuals), then individual (individual preference across a stakeholder group) and consensus (consensus preference within a stakeholder group) for our six stakeholder groups.

	All participants		***Big Farms***			***Conservationists***			***Counties***			***Forest Users***			***Pastoralists***			***Smallholders***		
	Individual N = 51		**Individual n = 7**		**Consensus**	**Individual n = 14**		**Consensus**	**Individual n = 5**		**Consensus**	**Individual n = 6**		**Consensus**	**Individual n = 7**		**Consensus**	**Individual n = 12**		**Consensus**
Land-use option	**M**	**SE**	**M**	**SE**		**M**	**SE**		**M**	**SE**		**M**	**SE**		**M**	**SE**		**M**	**SE**	
Business As Usual	17.03	0.57	14.62	1.12	17.66	15.77	0.952	13.15	17.68	1.84	20.92	14.32	1.61	16.67	18.34	1.09	17.99	20.20	1.26	18.49
Farming Future	9.06	1.07	6.19	1.79	11.13	6.34	1.12	3.36	8.50	2.96	12.13	7.82	2.21	2.42	8.59	2.97	3.79	15.04	3.18	4.65
Forestry Future	37.23	1.94	40.03	5.94	57.36	40.15	2.96	15.48	46.40	3.38	29.19	28.32	4.63	28.49	38.68	6.61	43.99	31.97	4.48	22.97
Green Future	62.39	1.75	61.52	5.37	43.98	61.98	2.61	86.11	55.89	4.32	60.78	70.45	4.50	76.19	62.04	5.95	60.67	62.22	4.15	77.36
Blue Future	47.01	0.66	46.22	1.42	41.36	48.34	1.16	46.87	45.80	3.01	48.62	49.41	2.38	54.10	46.00	1.20	47.52	45.77	1.51	58.88

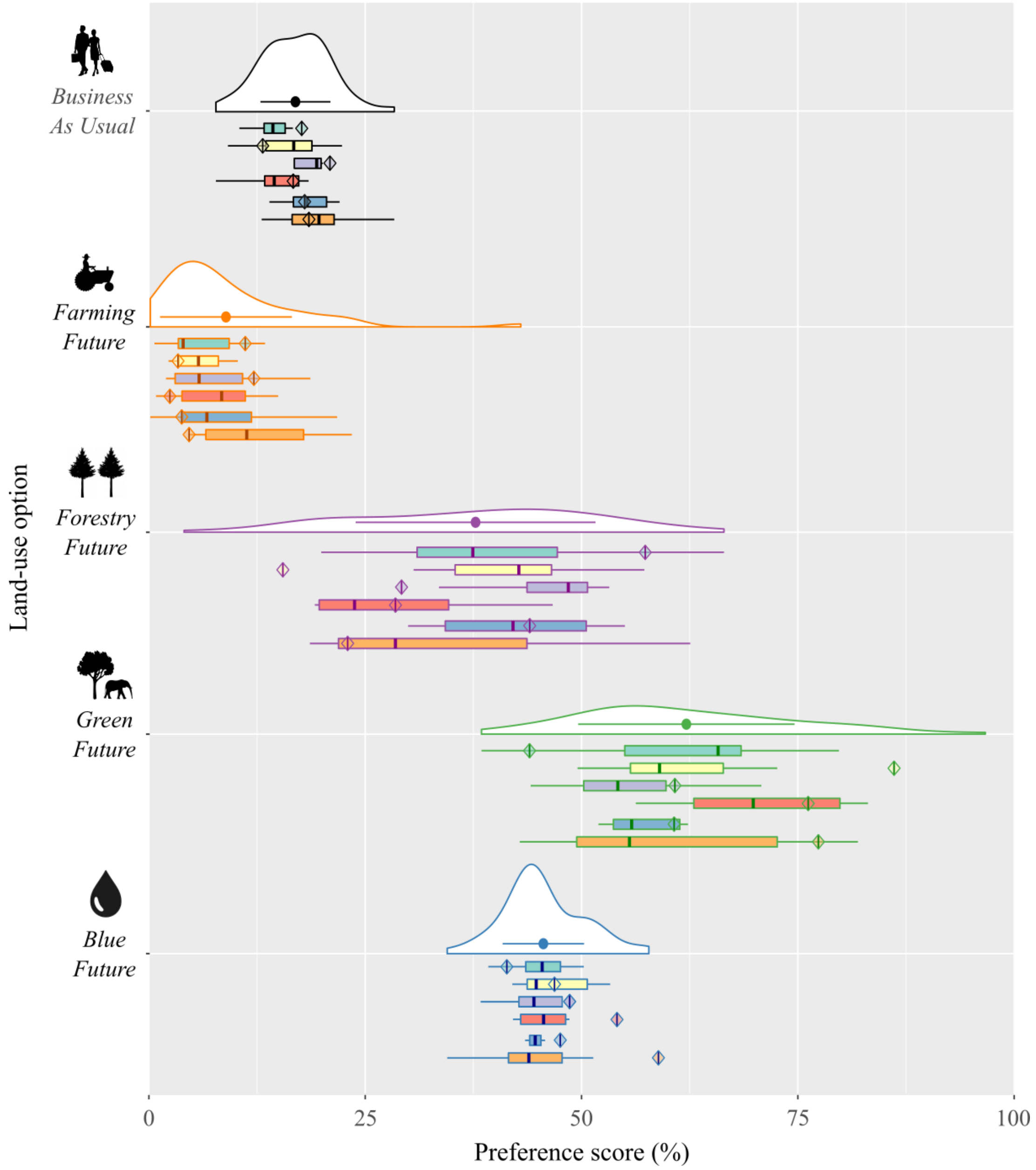

Figure 3.7: Stakeholder preference scores for the five land-use options: *Business As Usual* = black; *Farming Future* = orange; *Forestry Future* = purple; *Green Future* = green; and, *Blue Future* = blue. Horizontal half violin plots show preference value distribution across all individuals (personal values), with the point and line within the white of half violins showing mean and standard deviation. Horizontal boxplots show preferences for individuals (personal values) across a stakeholder group. the diamonds indicate consensus preference (shared values) within a stakeholder group: *Big Farms* = green; *Conservationists* = yellow; *Counties* = light purple; *Forest Users* = red; *Pastoralists* = blue; and, *Smallholders* = orange. See Section 3.3.2.1.1 for further details on the different stakeholder values.

3.4 Discussion

Our structured decision-making process uncovered broad stakeholder preference for habitat restoration and wildlife corridor establishment (*Green Future*), as well as the rezoning of forestry activities (*Forestry Future* and *Blue Future*) within Kenya's Central Highlands. These preferences were based on an agreement of the importance of biodiversity conservation, climate change mitigation and water-based ecosystem services, which were most valued by *Forest Users*, *Smallholders* and *Conservationists*. Indeed, *Green Future* consistently ranked top. Wildlife corridor establishment is a major focus of national planning (Government of Kenya, 2017) and the preference for this was surprising because local support from local community land-owners has been mixed in the past (Kamweya et al., 2012b), with successful projects established mainly by large agribusiness (*Big Farms*) (Nyaligu & Weeks, 2013). *Forestry Future* was the only land-use option that outranked *Green Future* beyond the preference of individuals. This happened when *Big Farms* gave higher importance to provisioning services and water flow control during the consensus building stage of the MCDA. This highlights the importance of maintaining water-based ecosystem services, as large agribusinesses are a dominant actor in the region, and these services are vital to the livelihoods of all stakeholders (Notter et al., 2007; Sungi, 2018). Forestry is also an important land-use in Kenya's Central Highlands (Emerton, 1999; Kehlenbeck et al., 2011), providing timber and cash crops during plantation establishment, as well as leaving land available for livestock grazing. Nonetheless, caution is needed as exotic plantation establishment within the savannahs and grasslands between protected areas would have negative impacts on biodiversity (Bond et al., 2019) and potentially release the carbon stored in soil (Waller et al., 2020).

While *Green Future* and *Forestry Future* were top preferences, they were also characterised by large variation, suggesting they capture unresolved differences in opinion. If either is implemented, challenges could therefore arise when landowners/users are affected. For instance, *Forest Users* as individuals across a stakeholder group had the strongest mean preference for *Green Future* as they would benefit from the expansion of community-led reforestation in the short term. This type of reforestation is currently done in partnership with the Kenya Forest Service and the Mount Kenya Trust (a local NGO) (Mount Kenya Trust, 2018). However, individuals would eventually lose the right to cultivate their land after the trees are established, as well as suffer losses due to reduced plantation forestry with the protected areas. Elsewhere such relocation has caused increased environmental destruction and social tensions between aggrieved communities (Agrawal and Gibson, 1999; Witcomb and Dorward, 2009). Implementing the reforestation indicative of *Green Future*, or the forestry expansion in *Forestry Future*, must thus be undertaken with care and a long-term outlook that supports the needs of all stakeholders.

Business As Usual, representing what happened after the two previous moratoria on forest resource extraction (Emerton, 1999; Vanleeuwe, 2004), had a low preference and variation was low. This indicates that ending the current moratorium without rezoning forestry activities within the protected areas would not be an outcome welcomed by stakeholders. It also highlights the risk of history simply repeating itself, as shown by the repeated moratoria. *Blue Future* consistently ranked as second preference and variation was low, demonstrating consistent support for the rezoning of plantation forestry to the periphery of the protected area complex and the expansion of reforestation activities. *Blue Future,* therefore likely represents the most balanced option, emphasising the ecosystem service benefits associated with improved environmental management of montane habitats (Notter et al., 2007; Viviroli et al., 2007). However, the successful implementation of rezoning plans will require management authorities to galvanise the local support we found to appropriately ensure effectiveness and stewardship. A key consideration will be avoiding human-human and human-wildlife conflict, as this could undermine restoration and conservation goals by appropriately relocating plantation forestry and reforesting former plantations. Reforesting degraded areas of low connectivity value within the protected areas was also discussed positively by the stakeholders, with the areas of Imenti, Gathiuru, and Thego mentioned specifically. It was proposed that this should be done via agroforestry using predominantly native flora, which is likely to support local livelihoods (Chazdon & Brancalion, 2019; Orsi et al., 2011). The spatial planning required to implement any further rezoning could also be conducted at the catchment level, targeting catchments that would deliver the greatest benefits for the largest human populations first (Garcia et al., 2018). A spatial MCDA (e.g. Vogdrup-Schmidt *et al.*, 2017; Musakwa, 2018) would be a useful tool in this regard, but would require continued stakeholder consultation.

The stakeholder discussions raised the underlying challenges associated with implementation. It was felt that the needs of downstream communities (*Pastoralists*) needed greater consideration, as they would be affected by upstream water resource management and particular concern was expressed about the hydrological and socioeconomic impacts of damming projects (Mukiri & Mundia, 2016; Sungi, 2018). Stakeholders also stressed the economies of scale that influence a landowner’s ability to tolerate and mitigate human wildlife conflicts. *Big Farms* recognised that they could afford to experience limited crop damage when wildlife moves across their land (e.g. Nyaligu and Weeks, 2013), whereas *Smallholders* cannot (e.g. Kamweya et al, 2012a; 2012b). Payment for ecosystem service initiatives could therefore be a valuable approach to promoting landscape connectivity, being more appropriate than land purchase or easement approaches in Kenya (Curran et al., 2016). Interesting discussions also occurred centred around where responsibilities for climate change mitigation through reforestation lie. *Smallholders*, *Forest-users* and *Counties* weighted climate change mitigation highly but, during the consensus building stage, *Big Farms* gave this a lower weight, stating that this was the responsibility of the high-income countries where the industrial revolution began. This has been argued in the literature (e.g. Agarwal *et al.*, 2019) and reinforces the findings of other studies that have shown how local communities are aware of the severity of the ecological and climate crises and are on the frontlines of responding to it (Bluwstein et al., 2021).

The potential opportunities and challenges associated with maintaining ecosystem integrity in this study region align closely with the ecosystem service benefits and concerns that stakeholders have articulated in similar Afromontane systems (e.g. Fisher et al., 2011, van Soesbergen et al., 2017, Mengist, Soromessa and Legese, 2020). Ensuring a sustainable future for the ecosystems within Kenya’s Central Highlands requires meaningful engagement with the local communities living within the target landscapes, particularly because delivering restoration successfully will be difficult as Kenya transitions through economic development. Moreover, the study region is experiencing more variable and intense rainfall as a consequence of climate change (Schmocker et al., 2016), so environmental interventions to promote water infiltration and slow river discharge (Notter et al., 2007) should be viewed favourably. Nevertheless, in contrast, Kenya’s lowlands are experiencing more pronounced and frequent droughts (Collier et al., 2008), meaning national priorities for agriculture, forestry and other land-uses will inevitably shift. This reinforces the need to integrate conservation and restoration within wider land-use planning that scales from catchment-level through to national-level. Such actions will be central to meeting the global drive for restoration (UNEP, 2021) as part of the UN Decade on Restoration and to achieving the 2050 goal of living in harmony with nature (Convention on Biological Diversity, 2020).

3.5 Acknowledgements

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3.6 Supplementary information

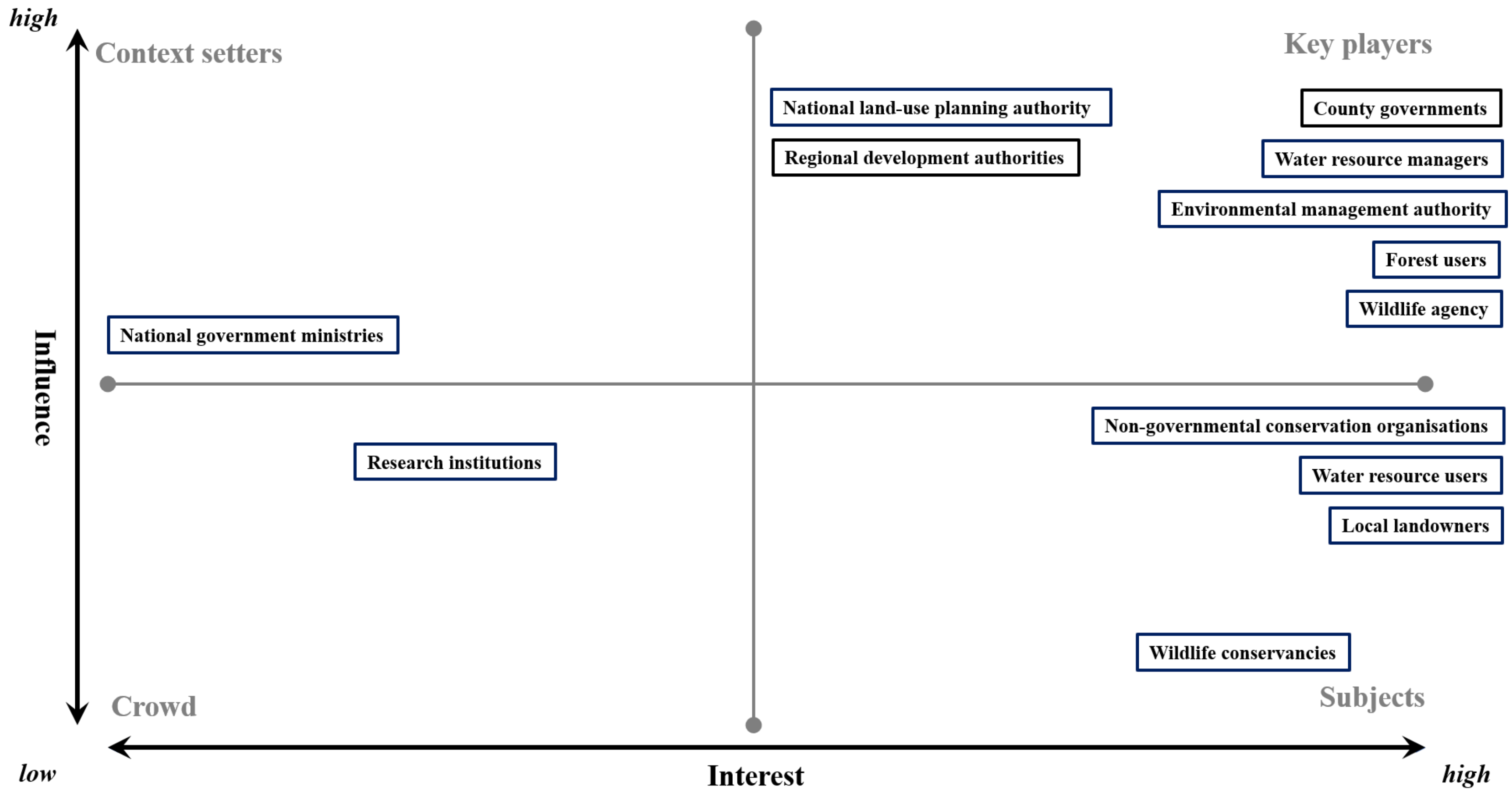
S3.1 Stakeholder analysis

Table S3.1: The six final stakeholder groups used for the multi criteria decision analysis (MCDA). These were compiled from 13 types of stakeholder categorised by operational remit, informed by an interest-influence matrix (Reed et al., 2009), from the initial 36 stakeholder organisations identified through interviews.

Final stakeholder groups	Stakeholder categorised by remit	Stakeholder organisations identified through interviews
(i) *Big Farms* – large scale agricultural and horticulture operators	Local landowners	Mount Kenya Elephant Corridor Committee
	Water resource users	Mount Kenya Flower Growers Association
(ii) *Conservationists* – practitioners and ecotourism operators	Non-governmental conservation organisations	Conservation Alliance of Kenya
		Kenya Wildlife Conservancies Association
		Laikipia Wildlife Forum
		Lewa Wildlife Conservancy
		Mount Kenya Trust
		Mpala Research Centre
		Northern Rangelands Trust
		Rhino Ark Charitable Trust
		Save The Elephants
		Space For Giants
	Research institutions	Centre for Research in ASAL Development
		Kenya Forest Research Institute
		Mpala Research Centre
	Wildlife agency	County Governments and Ministries
(iii) *Counties* – government officials	County governments	County Natural Resource Committees
		County Wildlife Conservation Committees
		Kenya Wildlife Service
		Members of County Assembly
	Environmental management authority	National Environmental Management Authority
	National government ministries	Ministry of Agriculture and Irrigation
		Ministry of Environment and Forestry
		Ministry of Interior
		Ministry of Lands
		Ministry of Tourism and Wildlife
		Ministry of Water and Sanitation
	National land-use planning authority	National Lands Commission
	Regional development authorities	Ewaso Nyiro North Development Authority
	Water resource managers	Water Resource Management Authority

(iv) *Forest users*	Forest users	Community Forest Associations
		Kenya Forest Service
(v) *Pastoralists* – downstream communities	Wildlife conservancies	Northern Rangelands Trust
		Laikipia Wildlife Forum
		Lewa Wildlife Conservancy
(vi) *Smallholders* – proximate farmers	Water resource users	Mount Kenya Flower Growers Association
	Local landowners	Mount Kenya Trust

Figure S3.1: Stakeholder categories, based on operational remit (Table S3.1), placed on an interest-influence matrix (Reed et al., 2009). Context setters are likely to be “highly influential but have little interest”, crowd are “stakeholders who have little interest in or influence over desired outcomes”, key players are “stakeholders who […] have high interest in and influence over a particular phenomenon”, and subjects have “high interest but low influence and although they are supportive”.



S3.2 Landcover mapping methods

We mapped current landcover within our study area using a combination of remote sensing and digitising. We first developed a typology of different landcovers based on the African Union's Africover project and the regional SLEEK standard (African Union, 2016). We then obtained Sentinel 2a satellite imagery from 29th of January 2018 (ESA, 2018), which consisted of four sets of 10 m resolution raster layers. This date was selected because it had low cloud cover and was taken at the beginning of a moratorium of forest resource extraction (Government of Kenya, 2018a), which represented the starting time point from which our land-use options diverged. The imagery was then clipped to the study area and converted from radiance to reflectance values using a DOS-1 conversion. This increases the accuracy of vegetation classification as it represents the physical reflective property of the target material rather than radiance, which is subject to atmospheric effects (Ray, 1994).

We then mapped the natural vegetation using the Semi-Automatic Classification Plugin (Congedo, 2018) for QGIS (QGIS Development Team, 2018). This first involved specifying the location of known patches for each natural vegetation landcover type, based on training sites that had been identified through previous vegetation mapping studies (Bussmann & Beck, 1995; Niemelä & Pellikka, 2004). The software then produced a set of spectral signatures for each landcover type. These were used to classify the remaining pixels using a maximum likelihood algorithm. To improve classification accuracy and remove errors, we put the output through a pixel sieve to remove landcover patches <1 ha as these were likely to be misclassifications. Natural vegetation types are stratified by elevation (Bussmann & Beck, 1995). Therefore we next refined classifications by incorporating elevation data by overlaying the output raster onto a digital elevation model (DEM) accessed from the German Aerospace Centre (DLR, 2018). We subsequently used on-screen digitising to map the anthropogenic landcover types. This was guided by existing GIS layers on plantation blocks from the Kenya Forest Service and GG's prior field experience. For this, we used DigitalGlobe's QuickBird satellite imagery (Google, 2018). We used a fixed scale of 1:3,000 and 10 m snapping to already digitised polygons to avoid overlaps. The results were ground-truthed in September 2018 using the African Union's SLEEK project random sampling methodology and identifying 25 sites along roads (African Union, 2016).

The map of landcover cover current as of January 2018 was then transformed into five maps of potential future landcover, one for each future land-use option (Figure 3.3). This was based on a set of landcover change rules defined within the future land-use option narratives based on the grounded theory analysis and then validated by the interviewees. '*Business as Usual*' was produced to capture the continuation of planned harvesting and replanting of exotic timber plantations, reforestation and agricultural expansion outside of protected areas in previously cleared areas. '*Farming Future'* involved all secondary habitats and cultivation within mixed-use areas of the PAs and outside PAs becoming annual crops. *'Forestry Future'* involved all secondary habitats and cultivation within mixed-use areas of the PAs and outside PAs becoming exotic timber plantations. *'Green Future'* was produced to capture all secondary habitats and cultivation within Mount Kenya's PAs being reforested and 300 metre wide grassland elephant corridors established between protected areas. '*Blue Future'* captured specific policy recommendations, including the reforestation of exotic timber plantations more than 500 metres inside the PA boundary, the reforestation of riparian reserves and the relocation of exotic timber plantations outside of Mount Kenya's PAs (Government of Kenya, 2018a).

S3.3 A comparison of weighting methods

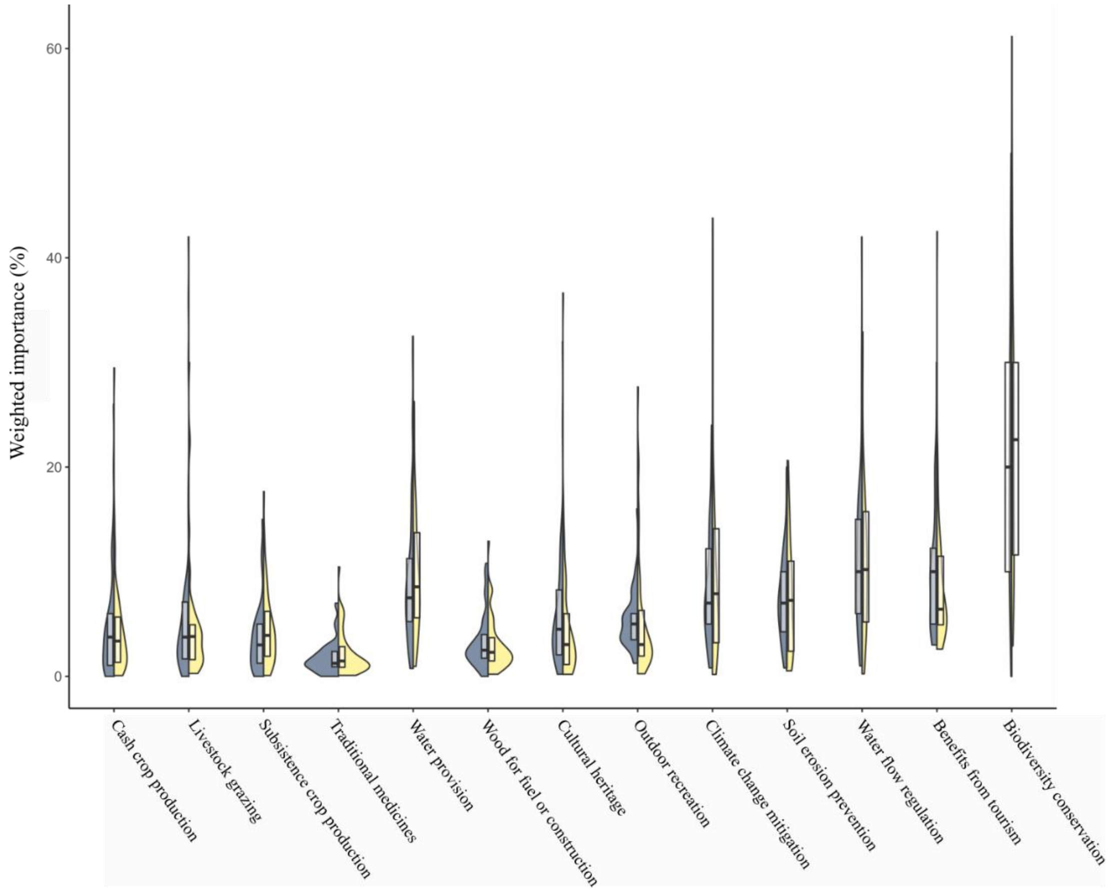


Figure S3.2: Split violin and box plots showing importance across all individuals (weights) for both decision models, the ranking and relative weighting method (blue) and the pairwise comparisons, analysed with the analytic hierarchy process (Saaty, 2008) (yellow). Violin plots show the distribution of importance values, whereas boxplots show the standard deviation with the central line indicating the geometric mean.

S3.4 Performance scores

Table S3.2: Performance scores, measures of ecosystem service delivery, for each criterion and each land-use option.

		Business As Usual		*Farming Future*		*Forestry Future*		*Green Future*		*Blue Future*	
Criterion	**Objective performance measure**	**Raw score**	**Normalised score**	**Raw score**	**Normalised score**	**Raw score**	**Normalised score**	**Raw score**	**Normalised score**	**Raw score**	**Normalised score**
Cash crop production	(i) Hectares of perennial crop cultivation	1372	38.98	0	100	0	0	305	0.8	1363	3.58
	(ii) Hectares of exotic plantations under cultivation	13490		38115		8		8		8	
Livestock grazing	(iii) Hectares of established exotic timber plantations	37818	24.54	16	0	123477	100	128	16.01	46383	42.94
	(iv) Hectares of secondary grasslands	46782		12393		12393		32050		19042	
Subsistence crop production	(v) Hectares of annual crop cultivation	12660	54.76	85394	100	40	0	40264	47.13	33037	46.57
Traditional medicines	(vi) Hectares of primary vegetation	215362	3.54	214021	0	214021	0	251938	100	233038	50.15
Water provision	(vii) Co$ting Nature measure of realised water provision indexed globally	203611	27.4	202407	0	206802	100	203940	34.88	203958	35.3
Wood for fuel or construction	(viii) Hectares of established exotic timber plantations	4886	3.94	16	0	123477	100	128	0.09	46383	37.56
Cultural heritage	(ix) Hectares of primary vegetation	215362	3.54	214021	0	214021	0	251938	100	233038	50.15
Outdoor recreation	(x) Hectares of primary vegetation	215362	26.19	214021	0	214021	0	251938	100	233038	49.26
	(xi) Hectares of secondary vegetation	65245		12757		12757		49191		29013	
Benefits from tourism	(xvi) Hectares of primary vegetation	215362	26.19	214021	0	214021	0	251938	100	233038	49.26
	(xvii) Hectares of secondary vegetation	65245		12757		12757		49191		825	
Biodiversity conservation	(xvii) Hectares of primary vegetation	215362	4.75	214021	0	214021	0.68	251938	100	233038	48.98

	(xix) Linkage mapper's additional pathway availability, a measure of landscape connectivity	1206		692		957		1870		825	
Climate change mitigation	(xii) Co$tingNature measure of forest carbon storage (/1000)	180120	27.22	173751	0.42	173630	0	197421	100	192015	77.25
	(xiii) Co$tingNature measure of forest carbon sequestration	4067360		3868341		4067360		4584685		4403008	
Water flow regulation	(xiv) The inverse of the WaterWorld measure of runoff (/1000)	-47662	3.89	-47764	0	-45137	100	-47499	10.07	-46251	57.6
Soil erosion prevention	(xv) The inverse of the WaterWorld measure of hillslope net erosion	25110	5.16	24273	0	40518	100	25993	10.59	28210	24.24

S3.5 Sensitivity analysis

We carried out a sensitivity analysis by first assigning equal importance weightings to each criterion, representing a null response. When weighted as such, *Green Future* ranked highest. We then adjusted the weighting of each criterion from a tenfold decrease to a tenfold increase to investigate robustness in the model by recording reversal points in land-use option preference rankings. Here the reversal point for *Farming Future* to outrank *Green Future* occurred when a just over sixfold higher weight was given to cash crop production. The reversal point for *Forestry Future* occurred when a just over threefold higher weight was given to wood for fuel or construction, a fourfold higher weighting was given to livestock grazing, a fourfold higher weighting was given to soil erosion prevention, a fourfold higher weighting was given to water flow control, or when a fivefold higher weighting was given to water provision. *Business As Usual* and *Blue Future* did not have reversal points within the tenfold range of equal importance.

We then adjusted importance weightings for each criterion from mean importance across all individuals to test the robustness of our mean response. Here reversal points for *Forestry Future* to outrank preference for *Green Future* occurred when a fourfold higher weighting was given to water flow control, a fivefold higher weighting was given to water provision and soil erosion prevention, an eightfold higher weighting was given to livestock grazing or when a tenfold higher weighting was given to wood for fuel or construction.

We also tested whether the model was sensitive to changes in the ‘multipliers’ (see main text Section 3.3.2.1.1.2) assigned to secondary vegetation for outdoor recreation and benefits from tourism and the connectivity component of biodiversity conservation. There were no reversal points here.

Given our model was sensitive to importance weightings, we concluded that our model was robust enough to capture how differences in the importance weightings stakeholders gave to each criterion affect preference toward the different land-use options.

Chapter 4 Restoring habitat fragments produces smaller, better connected but more expensive conservation networks

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4.1 Abstract

1. Failure to halt habitat destruction and degradation has led to a focus on ecosystem restoration. Spatial planning that integrates this with biodiversity conservation and broader land-use planning can benefit biodiversity, ecological processes, human health, wellbeing and wider society. Systematic conservation planning is a comprehensive, transparent, and repeatable approach for designing ecological networks, finding priority areas to meet conservation targets, whilst maintaining connectivity and minimising cost.
2. We used the systematic conservation planning approach to identify where best to locate zones for conservation and restoration in Kenya to meet area-based targets for 36 vegetation types and 127 important species. We also compared the results from 10 scenarios, based on specifying that increasingly large patches of natural habitat should be excluded from the conservation zone and targets met instead by restoring larger patches elsewhere. We then compared the outputs based on their financial cost, land area, patch characteristics, and overlap with human populations and carbon and water services.
3. Kenya's protected areas cover 19% of its land and meet a third of habitat and species targets. Additional sites were needed to meet representation gaps, requiring conservation attention on a total of 29% and restoration on 6% of Kenya, overlapping with a seventh of total carbon and 80% of total clean water. The scenarios show that the estimated management cost for this ranged from $7.73 to $9.41 billion, with the cost of restoration three orders of magnitude more expensive than conservation. Restoring land to reduce habitat fragmentation, instead of conserving small vegetation patches, increased mean patch size by up to 800% but also increased costs, land area, and the number of affected people.
4. Policy implications: targeted interventions would meet draft CBD policy commitments for area-based conservation and other restoration commitments. This has unprecedented implications for the 12-13 million people living in these areas. If incentivised and implemented equitably, this is a chance to mainstream biodiversity within these communities for the benefit of both the planet and its people.

Keywords: Marxan with Zones, Systematic conservation planning, spatial prioritisation, Kenya, area of habitat, patch characteristics

4.2 Introduction

Humans have transformed more than 80% of terrestrial habitats, with over half of the Earth's land surface under intensive human use (Ellis et al., 2021). This continues to impact the environment negatively, leading to the loss of biodiversity and ecosystem services (Ruckelshaus et al., 2020; Williams et al., 2020). In response, the Convention on Biological Diversity (CBD) aims 'to galvanise urgent and transformative action by Governments and all of society' (Convention on Biological Diversity, 2020). Their post-2020 'zero-draft' commitments seek to mainstream biodiversity conservation alongside society's efforts to achieve the Sustainable Development Goals, and the first two 2030 Action Targets are to have 50% of land under spatial planning and 30% managed through well connected and effective networks of protected areas (PAs) and other effective area-based conservation measures (OECMs). With global terrestrial PA coverage at just over 15% (IUCN and UNEP-WCMC, 2021), the need for further area-based conservation must align biodiversity goals with other socio-economic and climate aims (Bhola et al., 2021; Maxwell et al., 2020).

While humans have converted 51 million km^2 of global land, they have also left 20 million km^2 degraded (Minnemeyer et al., 2011). In combination, this isolates wildlife populations and has other knock-on effects on biodiversity and ecosystem services. To reverse this habitat loss, the UN have named this the 'decade on ecosystem restoration' (UNEP, 2021), echoing the CBD's call for humanity to be on a pathway to recovery by 2030, although the zero-draft currently has an undefined target for restoration. This presents a chance to revolutionise biodiversity conservation practices, creating systems that benefit nature and people. Integrating this drive for restoration into spatial planning of conservation and other land-uses can help mainstream biodiversity in human societies and industries (Milner-Gulland et al., 2021).

Systematic conservation planning is the most widely applied approach for designing ecological networks (Margules & Pressey, 2000). This centres on a structured approach to creating management policies and identifying conservation areas. It provides a comprehensive, transparent, and repeatable way to plan area-based conservation at various scales. Marxan is the most commonly used systematic conservation planning software package (Ball et al., 2009) and it is designed to identify sets of priority areas for meeting conservation targets whilst maintaining connectivity and minimising costs. However, this software can only meet targets for one management action (normally for conservation or protection), whereas decision makers often want to assign land or sea parcels to different management zones. This is why Marxan with Zones was developed, which lets users set targets for different types of management and specify how well different species and ecosystems are represented in different zones (Watts et al., 2009). Marxan with Zones has been used most often in marine spatial planning, as it lets planners identify where different types of marine protected area should be established, based on some species being able to persist in limited-take marine protected areas (Grantham et al., 2013; Klein et al., 2010). This software can also be used to identify priority areas for conservation and restoration simultaneously (e.g. Barbosa et al., 2019), so here we present the first national study to use this approach and investigate the trade-offs involved.

Our analysis focuses on the national scale because this is the level where CBD commitments are made (e.g. Bicknell et al., 2017), and the level where data on the number of people potentially affected and the implications for ecosystem services conservation are best considered. We used the Republic of Kenya as a case study because this is a mega-diverse nation that is also a priority for ecosystem restoration (Strassburg et al., 2020) and has committed to restoring 51,000 km^2 through the Bonn Challenge (Government of Kenya, 2016). Our analysis consists of the following components: (i) we measure the contribution of Kenya's different PA types for meeting representation targets for vegetation types and important species and maintaining important ecosystem services; (ii) we use Marxan with Zones to identify where best to conserve and restore land to meet any target shortfalls; (iii) we investigate the results of scenarios where small fragments of natural habitat are excluded from the conservation priority areas, forcing the software to meet the targets by restoring habitat instead. Excluding these small fragments automatically produces more connected networks of land, so we then investigate the impacts of this on zone area, management costs, number of people living within the priority areas and ecosystem service provision.

4.3 Methods

4.3.1 Study system

Kenya contains a range of habitat types, including Afromontane, coastal and lowland forests, varied savannas and deserts. Kenya's conservation model was initially based on state-mandated PAs but underwent a policy shift from the 1970s onwards, recognising that 65% of Kenya's megafauna utilise land beyond them (Western et al., 2009). This recognition led to the formalisation of Wildlife Conservancies, defined as 'private or community-owned protected areas managed […] for wildlife conservation and other compatible land-uses that improve livelihoods' (Government of Kenya, 2018b). Rapid human population growth and economic development have driven land-use changes, including agricultural expansion and urbanisation. Today much of Kenya's natural habitats are converted or otherwise degraded, adding to habitat destruction from the colonial era. Kenya's Conservancies are growing in number and the country has also committed to restoring 51,000 km^2 of land by 2030 (Government of Kenya, 2016). Land-use planning relies on national and devolved county policy frameworks. This presents a clear opportunity for Kenya to integrate habitat restoration and conservation within their county and national spatial planning frameworks, as recognised in Kenya's Sixth National Report to the Convention on Biological (Government of Kenya, 2020). So, in our analysis, we assigned land into one of three zones named 'Conserve', 'Recover', and 'Wider landscape' for land that was not needed to meet our conservation and restoration objectives.

4.3.2 Mapping the prioritisation features and setting targets

We produced a list of prioritisation features for inclusion in the spatial prioritisation based on representing Kenya's different vegetation types and threatened vertebrates. We then mapped the intact and restorable extent of the 36 natural vegetation types by combining data on potential natural vegetation types (van Breugel et al., 2015) with global maps of terrestrial habitat types (Jung et al., 2020) and plantations (Harris et al., 2021). We mapped the distributions of the 127 Near Threatened and Threatened terrestrial vertebrate species found in Kenya based on the IUCN range maps for mammals, amphibians, and reptiles (IUCN, 2020). These were processed based on the standard approach for mapping species ranges (Butchart et al., 2015), which included refining the maps to account for the elevation and habitat associations of each species (Brooks et al., 2019). We mapped the current distribution of each species based on the current extent of their associated habitats and their restorable distribution based on the restorable extent of these habitats.

We set overall representation targets for each prioritisation feature based on their total potential extent. Where an overall target was unachievable through selecting their available intact extent, we set their conservation target as the total amount of available extent and the restoration target as the shortfall between the overall target and the conservation target. We set the overall targets for each of the 36 natural vegetation types as 20% of their total potential area, following the approach used in the South African National Spatial Planning process (SANBI, 2018). We set the overall targets for the species using the minimum proportion of each species' distribution required to ensure its persistence (Butchart et al., 2015; Rodrigues, Andelman, et al., 2004). This is calculated as a species-specific function of its global range, based on 100% for a range under 1,000 km^2, 10% for a range over 250,000 km^{2,} and a log-linear scale for the values between. Full details of data processing for vegetation and biodiversity prioritisation features are provided in Supplementary Information S4.1.

4.3.3 Producing the planning system

As part of the spatial prioritisation process, we divided Kenya's land surface into a number of planning units. These were based on a series of 10 km^2 hexagons combined with the different PAs, so that we could accurately represent the boundaries of these conservation areas in the analysis. We grouped these PAs as i) National Parks, ii) Other state-run PAs including National Reserves, National Sanctuaries and RAMSAR sites, iii) Forest Reserves, and iv) Wildlife Conservancies. These were then unioned with the hexagons to create the final layer, which divided Kenya into 64,445 planning units (Figure 4.1). Full details of data processing for producing the planning system are provided in Supplementary Information S4.2.

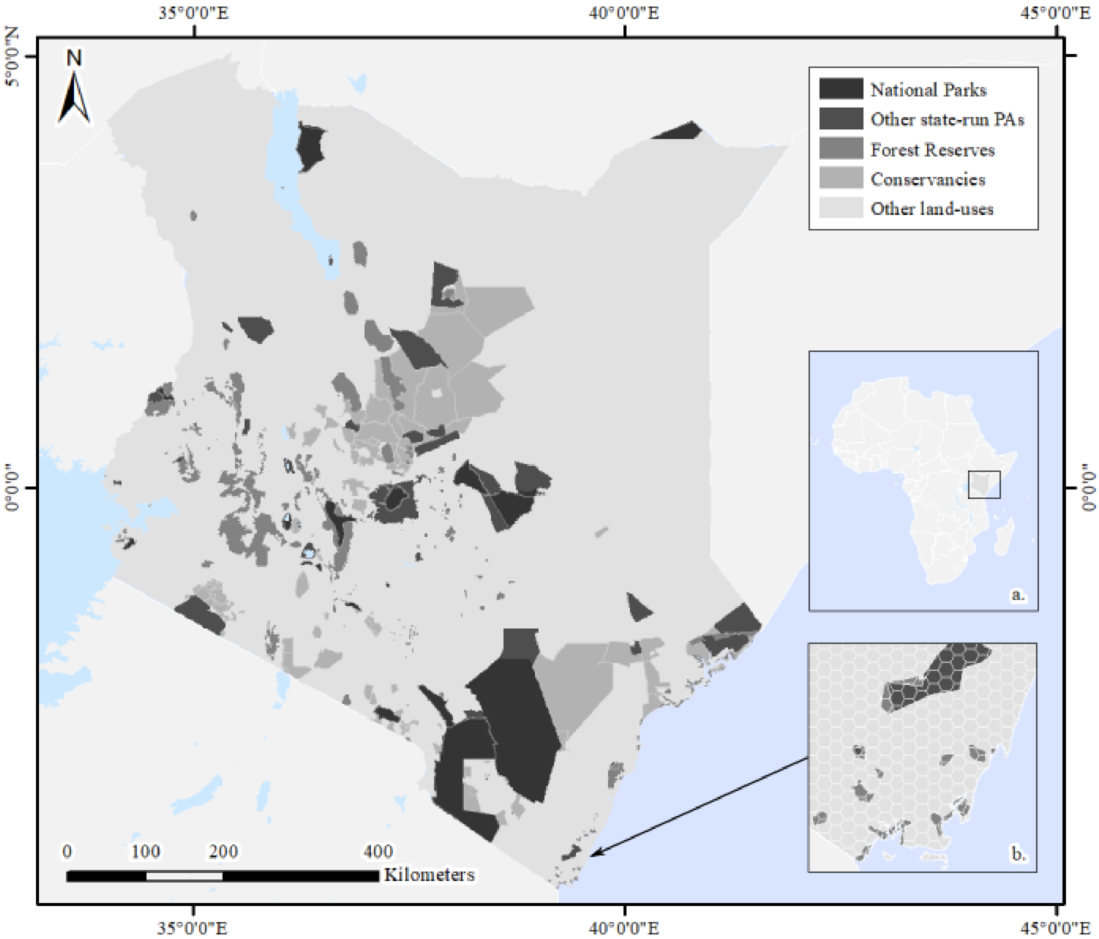

Figure 4.1: The planning region consisting of planning units created by unioning 10 km^2 hexagonal polygons with Kenya's protected areas (PAs): i) National Parks, ii) Other state-run PAs including National Reserves, National Sanctuaries and RAMSAR sites, iii) Forest Reserves, iv) Wildlife Conservancies and also other land-uses. Inset 'a.' shows the location of Kenya within Africa. Inset 'b.' shows the planning units across the south of Kwale County

The spatial prioritisation process involves giving each planning unit a cost for being included in the different zones. Restoration costs were based on the most comprehensive database available (de Groot et al., 2012) and scaled to a quoted price for active tropical forest restoration from the Kenya Forest Service, based on restoration taking place over a three year period. We calculated the cost for including a planning unit in the Conserve zone based on standard measures of annual conservation management costs per area (Moore et al., 2004), where the area was calculated as the amount of intact vegetation types and secondary habitats, multiplied by three to make it comparable to the three year period for restoration. The cost for selecting a planning unit for the Recover zone was the restoration cost for all restorable vegetation types and the conservation management costs of all vegetation types either intact or restored. Full details on cost setting are provided in Supplementary Information S4.2.

4.3.4 Spatial planning approach

We carried out the spatial prioritisation using Marxan with Zones (Watts et al., 2009) through the CLUZ plug-in (Smith et al., 2019a) to identify the most cost-effective solutions to meet the conservation and restoration targets. Marxan with Zones optimises the allocation of planning units to different management zones using a process called simulated annealing, solving a minimum-set problem by applying a complementarity-based algorithm to identify portfolios of planning units that meet the targets whilst maintaining connectivity and minimising planning unit costs. Marxan with Zones accounts for connectivity by measuring the amount of external edge for planning units in each portfolio and letting the user set a Boundary Length Modifier value for each zone which is multiplied by the edge to give a boundary cost. The higher the boundary zone cost value, the higher the cost for having portfolios containing isolated planning units, making it more likely that Marxan with Zones will select larger patches of planning units. Analyses are based on multiple runs, where each run produces a near-optimal solution by selecting portfolios of planning units that meet the different objectives. Marxan then identifies the 'best' output, which is the results of the run with the lowest combined cost (based on the combined planning unit costs for each zone, penalties for not meeting any targets and an optional boundary length penalty based on the edge of the selected planning units). A second output is maps of selection frequency for each zone, which is the number of times each planning unit was selected for a particular zone across the multiple runs. We built a planning system to allocate planning units to one of three management zones:

i) 'Conserve': areas where conservation attention is required to meet representation targets for intact prioritisation features.

ii) 'Recover': areas where conservation attention to meet representation targets for intact prioritisation features should be integrated with ecological restoration to meet targets for restorable prioritisation features. Thus, planning units in the Recover zone met both conservation and restoration targets.

iii) 'Wider landscape': areas where other land-uses could occur without compromising conservation or restoration targets. We emphasise that these areas must be managed with sound environmental governance and practice, to safeguard biodiversity and ecosystem services not captured in our analyses. We 'locked-in' planning units that contained ≥25% urban landcover into this zone, deciding they would be unsuitable for conservation or restoration.

4.3.5 Sensitivity analysis and calibration

We undertook sensitivity analysis and calibration of the input parameters for Marxan with Zones. This involved calibrating the number of iterations for each run, the feature penalty factor, and the zone boundary cost. We identified an optimal trade-off between run time and the median score at one billion iterations and 100 runs. The feature penalty factor was calibrated to ensure all targets for each run were met above 99.9%. We calibrated the zone boundary cost, also known as the boundary length modifier, to select viable patch sizes that were not too fragmented (Ball et al., 2009). This resulted in a zone boundary cost of 0.45 between the Conserve zone and the Wider landscape zone and a zone boundary cost of 165 between the Recover zone and the Conserve zone and Wider landscape zone. Full details on sensitivity analysis and calibration can be found in Supplementary Information S4.3.

4.3.6 Prioritisation scenarios

As a first step, we calculated the conservation and restoration cost, land area and amount of intact habitats, carbon and water in each of Kenya's PA types. We then used CLUZ to automatically calculate the contribution of each PA type to meeting the targets. This identified that restoration within PAs was essential to meet targets for several species. To identify where within the PAs to best restore the habitat for these species we ran an initial Marxan with Zones analysis. We then 'locked in' the selected planning units into the Recover zone, locking in the remaining planning units within the PAs into the Conserve zone. We were then ready to run analyses to identify which land outside of the PAs should be selected to meet the overall targets.

Our first main spatial prioritisation identified the best set of planning units for meeting the conservation and restoration targets, whilst minimising the planning unit and boundary costs. This was based on selecting planning units to be included in the conserve zone, even if the planning unit only contained a small fragment of natural habitat. We then carried out a subsequent nine prioritisation scenarios to investigate the results of specifying that planning units containing different amounts of anthropogenic landcover types should not be included in the Conserve zone. This meant we named our first analysis the >0% Scenario, as Marxan with Zones could select any planning unit outside PAs for the Conserve zone, i.e. planning units where the natural vegetation was >0% intact. The other scenarios were based on increasing this natural vegetation threshold in 10% increments, from the >10% Scenario to the >90% Scenario. We calculated the percentage of intact vegetation by summing the area of land that was not classified as arable land, exotic plantations, rural gardens and/or urban areas in each planning unit and dividing it by the area of the planning unit (Figure 4.2). This meant we considered secondary habitats other than those listed above as intact, as these had conservation value for some species.

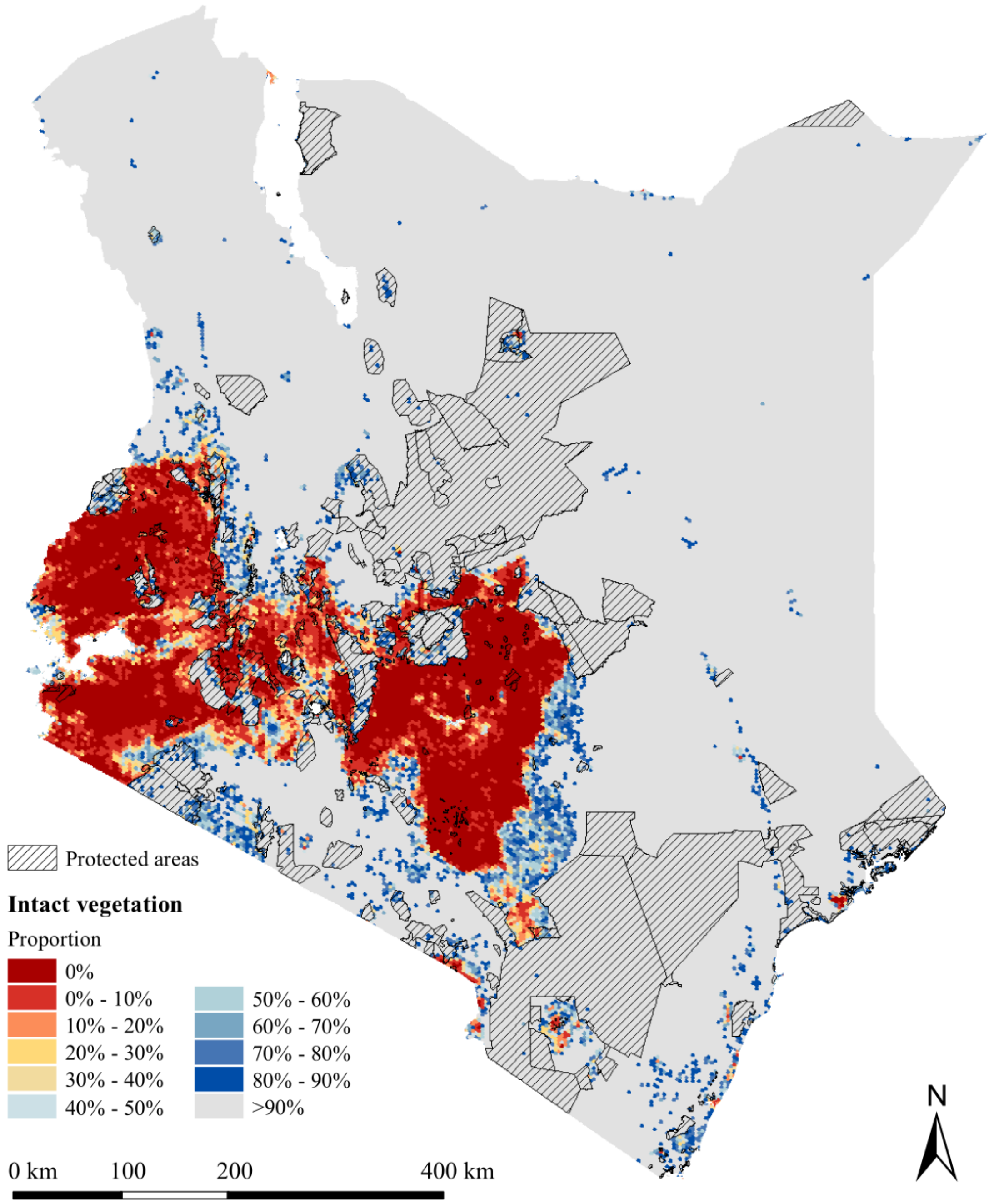

Figure 4.2: The proportion of intact vegetation in each of the 64,445 planning units. We calculated intactness as the proportion of each planning unit that was not arable land, rural gardens, plantations or urban.

We also modified the targets for the >10% Scenario to >90% Scenario scenarios to account for the fact that excluding the less intact planning units from the Conserve zone had implications for meeting the conservation targets. We did not change a feature's conservation target if this made it impossible to meet its overall target, which was the case for some species where all of their intact and restorable habitat was needed. But for the other relevant features we reduced the intact target by the amount in the excluded planning units and added this to the restoration target, so that their overall target could be met by selecting more of their habitat in the Recover zone.

Finally, for each output produced in each scenario we calculated the combined planning unit cost, the mass of carbon, the volume of water and number of people living in the Conserve zone, Recover zone and 'Total portfolio' (which we defined as the sum of these two zones). We estimated the number of people using LandScan's global population distribution model for 2019 (Rose et al., 2019) (Figure S4.8). We estimated the amount of carbon using maps of above- and belowground biomass and soil organic carbon density up to 1m depth (Soto-Navarro et al., 2020) (Figure S4.9; Figure S4.10, Figure S4.11). Water provision implications were estimated using the amount of potential clean water provision calculated through Co$ting Nature (Van Soesbergen & Mulligan, 2014) (Figure S4.12). All data except the area were processed in raster format at a 1-hectare resolution and assigned to the planning units using the Zonal Statistics function in ArcMap and then summed for each zone. Further details on populations carbon and water calculation can be found in Supplementary Information S4.4. We also calculated the number of patches of intact/restorable habitat and the mean area of these patches in the Total portfolio using Python and QGIS to select the relevant areas in the Conserve zone and the Recover zone, merge them together, dissolve the planning units into single part polygons and calculate their area for each output produced in each scenario.

4.4 Results

The planning region has a total land area of 575,373 km^2 and contains 491 PAs covering 19.2% of this land (Figure 4.1; Table S4.5). When combined, these PAs meet targets for 12 out of 36 vegetation types and 45 out of 127 species (Figure 4.3). Restoration within PAs is essential for meeting targets for 40 out of the 60 species requiring restoration, with 6 species having all their restorable extent and 14 having more than half of their restorable extent within PAs. These PAs have an estimated conservation management cost of $106 million, with National Parks containing the most intact habitats (98.4%) and Forest Reserves the least (71.4%) (Table S4.5).

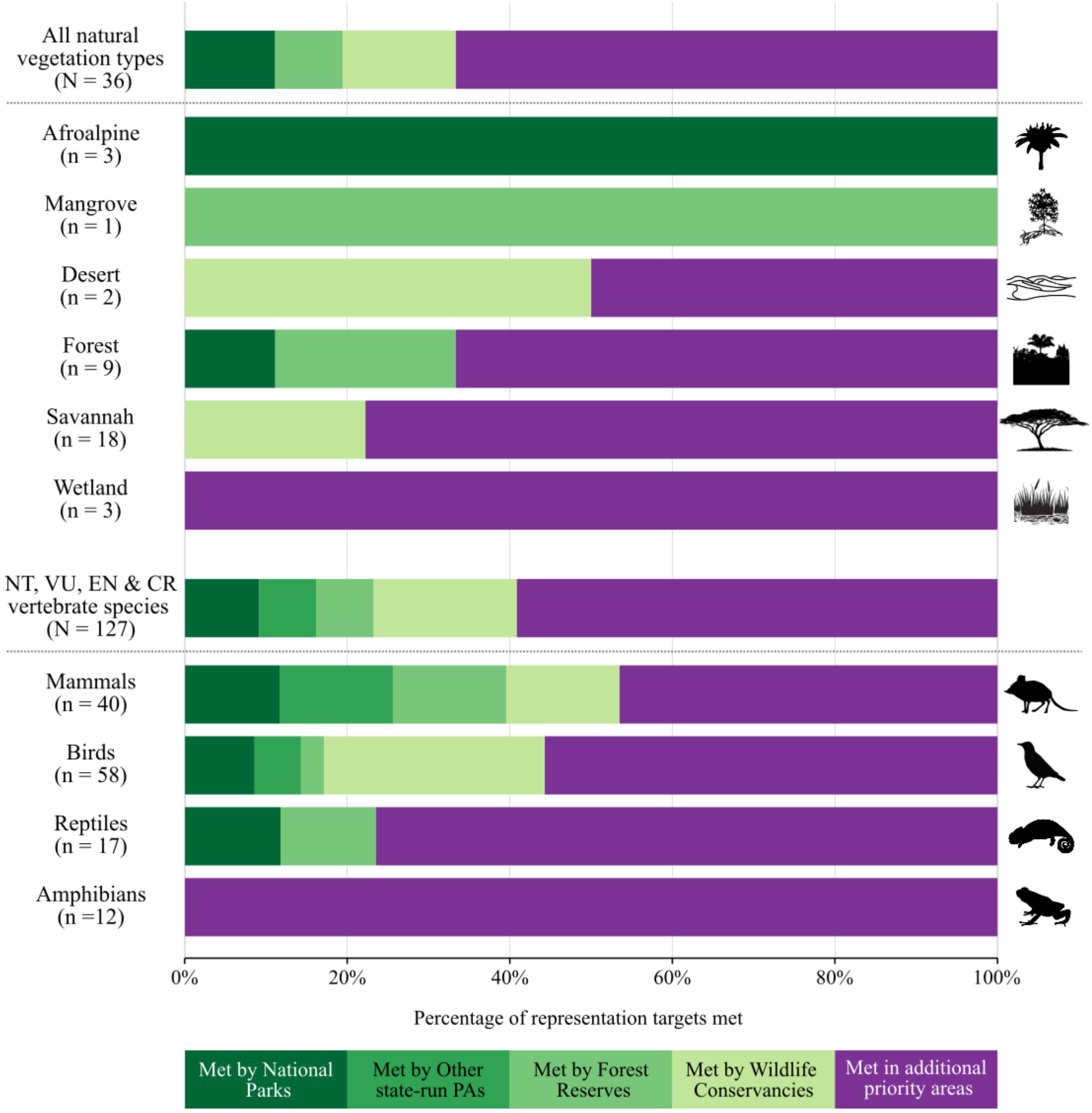

Figure 4.3: The percentage of prioritisation feature representation targets (both intact and restorable) met by sequentially including Kenya's different protected area (PA) types (greens) and the additional priority areas (purple). The number of natural vegetation type features and IUCN listed Near Threatened (NT), Vulnerable (VU), Endangered (EN) and Critical Endangered (CR) in each group are shown in parentheses. Representation targets were 20% of the potential extent of natural vegetation types and 100% of the potential extent for species with a range of <1,000 km^2 and 10% for a species with a range of >250,000 km^2 with a log-linear interpolation used for values between. Further details can be found in Supplementary Materials S4.1.

The results from the ten conservation and restoration scenarios show a fairly consistent spatial pattern, with large areas needed to meet the conservation targets in north-western, central, south-western and south-eastern Kenya and large areas needed to meet the restoration targets in central Kenya and along the coast (Figure 4.4a). Changes across the scenarios are best seen by comparing the >0% Scenario and the >90% Scenario (Figure 4.4b), which shows a decrease in the Conserve zone in western, northern, central and coastal Kenya, with small increases in areas receiving conservation attention mainly in northern Kenya and large increases in the Recover zone in the southern and centre. The selection frequency scores show whether planning units could be replaced in the portfolio by similar sites. This occurs for the Conserve zone in northern Kenya and the Recover zone in western Kenya (Figure 4.5), with a large number of planning units across the country being selected in every run. The >0% Scenario shows areas selectable for both zones in western and coastal Kenya. This was no longer the case in the >90% Scenario.

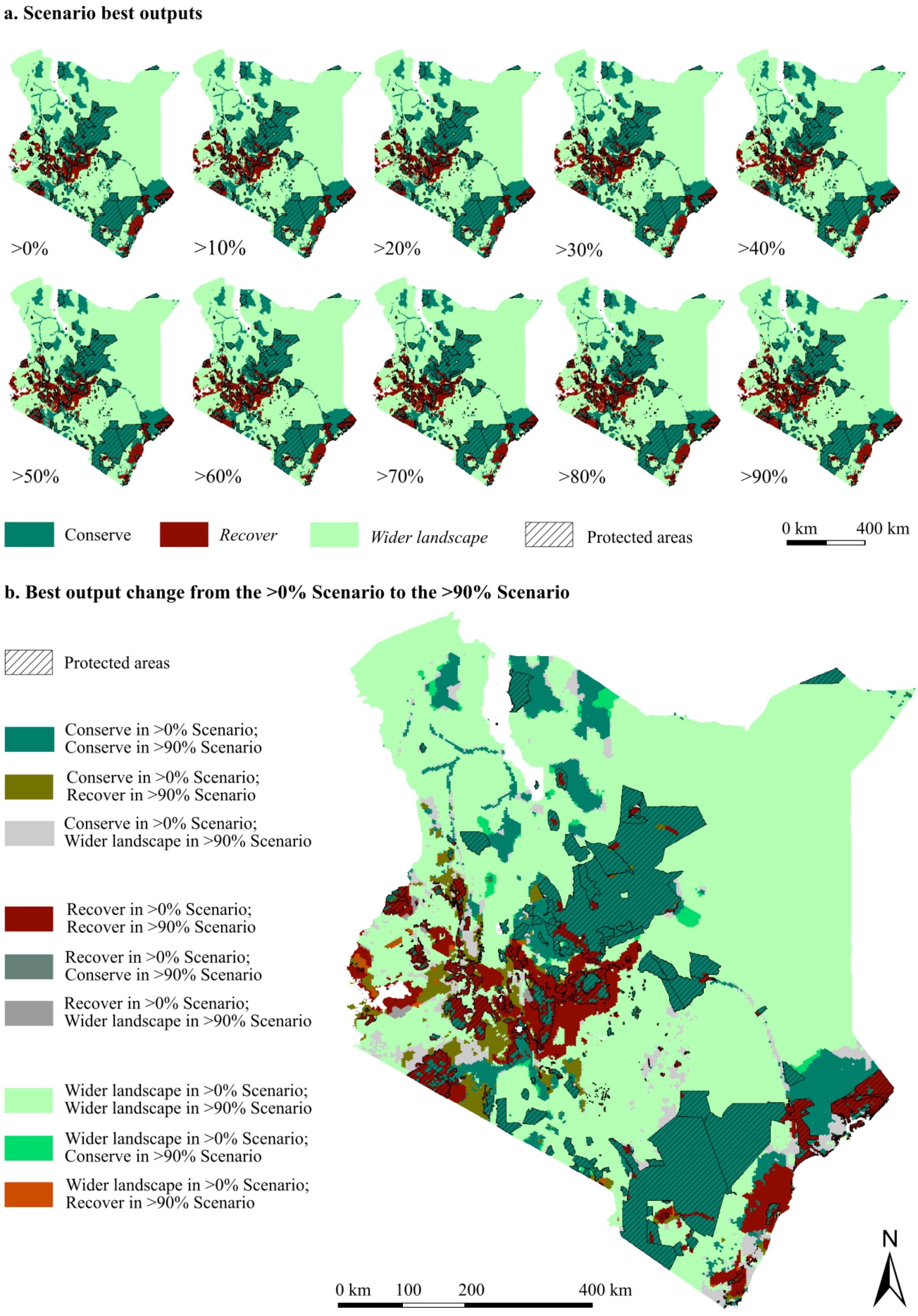

Figure 4.4: (a) The best outputs for the ten prioritisation scenarios and (b) the changes in planning unit allocation to the different zone types between the best output from >0% Scenario and >90% Scenario.

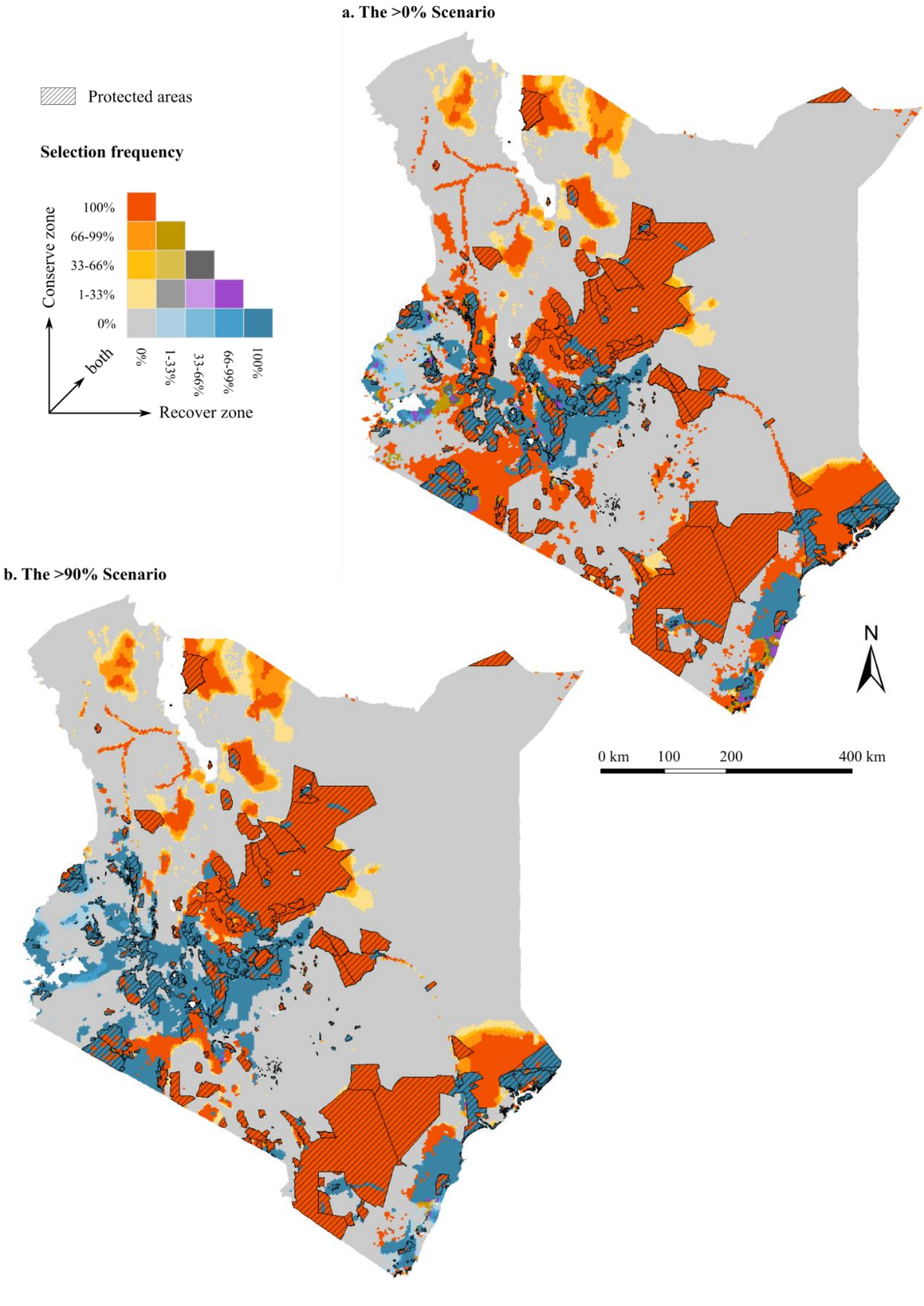

Figure 4.5: Selection frequencies for (a) the >0% Scenario and (b) the >90% Scenario. The selection frequency scores are the number of times each planning unit were chosen for the Conserve or Restore zones in each of the 100 runs. Lower value planning units can be replaced in the portfolio by others containing similar features.

The total financial cost of the best output for the different scenarios ranged between \$7.73 billion for the >0% Scenario and \$9.41 billion for the >90% Scenario, with the largest changes occurring between the >0% and >10%, and >80% and >90% Scenarios (Figure 4.6a). The >0% Scenario best output had a portfolio area of 200,282 km^2, covering 34.8% of the planning system. The total portfolio size rose until the >30% Scenario, after which it fell, and the >90% Scenario was 24,499 km^2 or 1.3% smaller, though the Recover zone component was a third larger (Figure 4.6b). In the >0% Scenario, a third of restoration was in PAs, which fell to a quarter in the >90% Scenario. The number of people living within the best portfolio ranged from 12.3 million for the >0% Scenario, with just over two-thirds occupying land identified for restoration, and 12.92 million in the >90% Scenario, with fourth fifths now found in restoration areas (Figure 4.6c). The patch characteristics changed dramatically as the total portfolios aggregated (Figure 4.7). The best output number of patches went from 4,381 in the >0% Scenario outputs to 527 in >90% Scenario. Mean patch size increased from 45.72 km^2 in the >0% Scenario to 375.5 km^2 in the >90% Scenario.

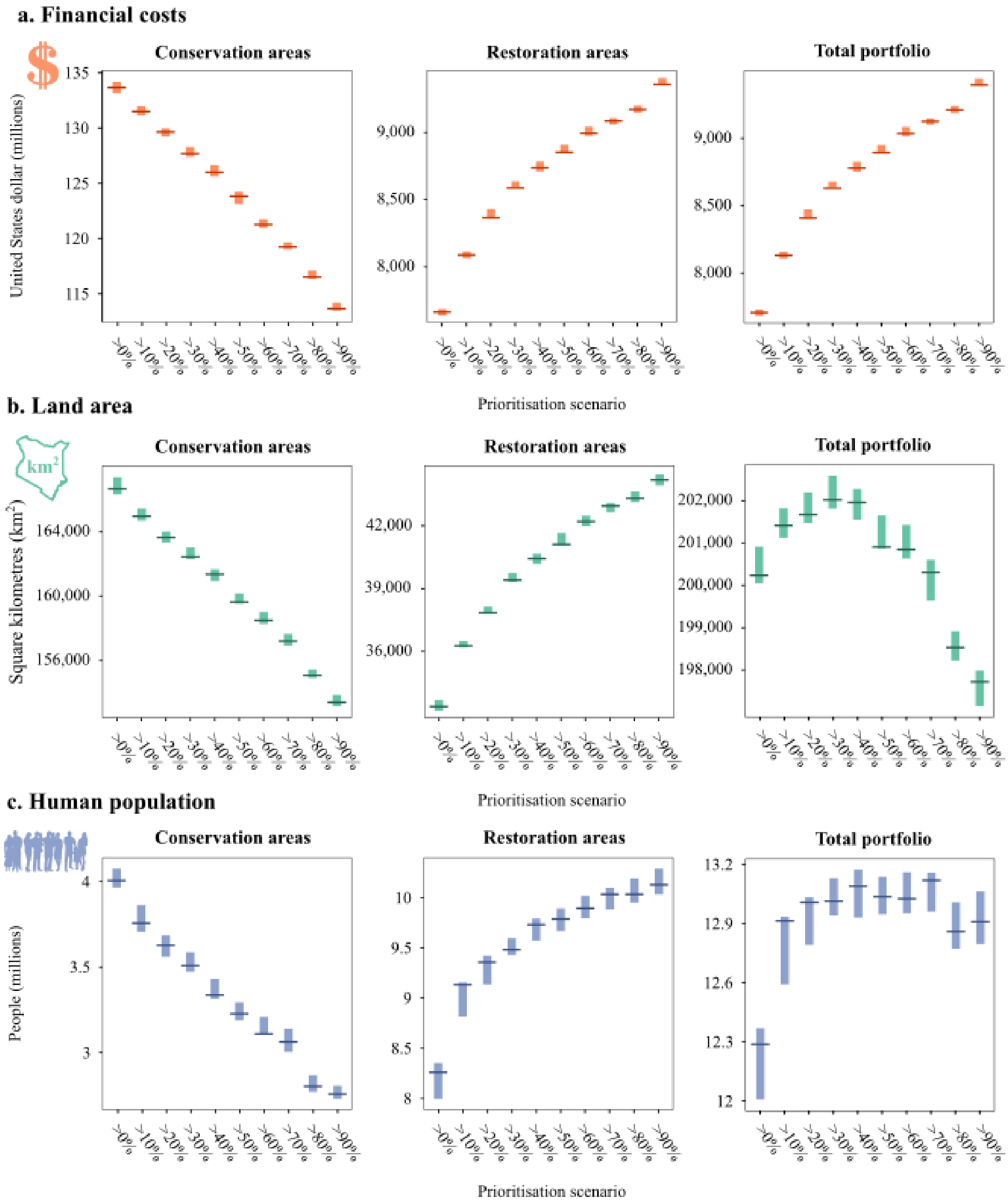

Figure 4.6: (a) The financial costs, (b) the land area and (c) number of people living per unit area for the areas requiring conservation attention (Conservation areas) and those requiring ecological restoration (Restoration areas), and both together (Total portfolio). These are shown as the range across 100 runs (vertical bar) and the best output value (horizontal line) for the ten prioritisation scenarios from the >0% Scenario to the >90% Scenario.

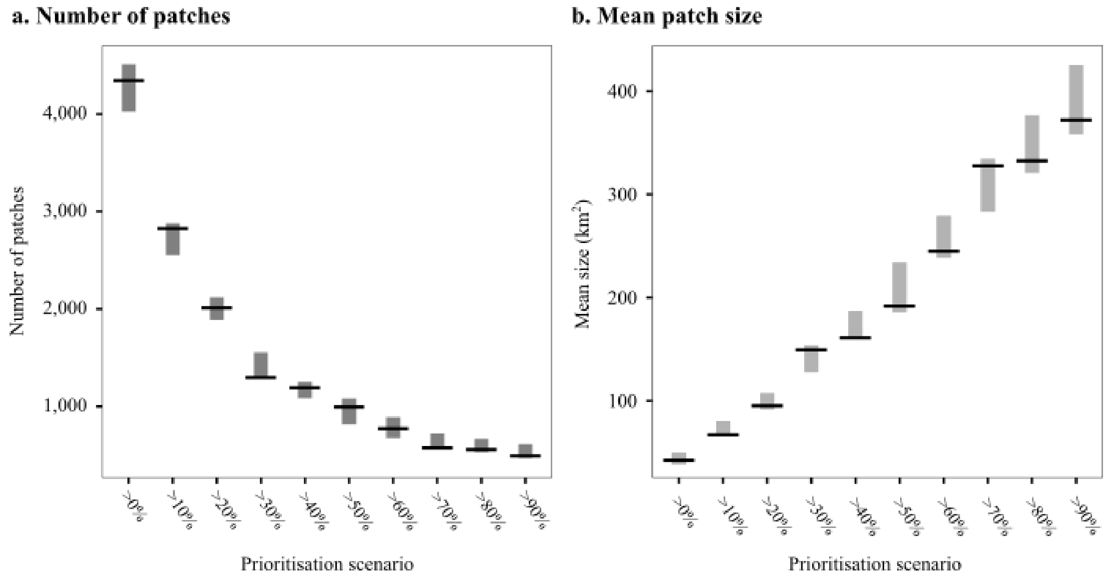

Figure 4.7: Patch characteristics for the total portfolio (the total conservation and restoration areas selected for the Conserve and the Recover zones) described as (a) the number of contiguous habitat patches and (b) the mean patch size. These are shown as the range across 100 runs (vertical bar) and the best output value (horizontal line) for the ten prioritisation scenarios from the >0% Scenario to the >90% Scenario

Kenya's PAs contain 1,047 megatonnes (Mt) of current carbon and could capture 12.6 Mt in above and belowground biomass. They also contain 13,367 million cubic metres of potential clean water, over half the country's total. Including the additional priority areas, the >0% Scenario best output contains 1,791 megatonnes (Mt) of carbon in the conservation areas, which decreases to 1,489 Mt in the >90% Scenario (Table 4.1). The >0% Scenario also has a further 223 Mt of carbon in the Recover zone, which increases to 564 Mt, in the >90% Scenario. The amount of intact carbon in above and belowground biomass was always lower than potential within restoration areas, but the amount in soil was threefold higher than the potential in above and belowground biomass, which was 61 Mt in the >0% Scenario and fourfold higher than the 149 Mt in the >90% Scenario. The total portfolio for the best output of the >0% Scenario contains 23,090 million M^3 or 88% of Kenya's total potential clean water, decreasing to 20,621 million M^3 in the >90% Scenario.

Table 4.1: Estimates of carbon storage and potential, as well as clean water provision in Kenya's protected areas and the amount in the prioritisation scenarios. Values are taken from the 'best' solution, the output out of 100 that has the lowest cost and penalties for not meeting targets and boundary length. Carbon is estimated in megatonnes and clean water provision is in million cubic metres (M^3). These were summed for areas requiring conservation attention (Conservation areas) and those requiring ecological restoration (Restoration areas) for carbon and the areas requiring conservation attention and those requiring ecological restoration (Total portfolio) for water.

	Conservation areas		**Restoration areas**			**Total portfolio**
Scenario	**Intact above and belowground carbon in biomass (megatonne)**	**Intact soil organic carbon (megatonne)**	**Intact above and belowground carbon in biomass (megatonne)**	**Intact soil organic carbon (megatonne)**	**Potential above and belowground carbon in biomass (megatonne)**	**Total potential clean water (million M^3)**
Protected areas	189.4	815.2	6.2	35.9	12.6	13,367
>0% Scenario	329.0	1,462.7	33.5	189.7	60.7	23,090
>10% Scenario	322.7	1,435.9	85.0	426.0	134.7	22,387
>20% Scenario	316.7	1,415.1	86.4	434.9	137.7	22,053
>30% Scenario	313.0	1,394.1	87.2	441.2	139.6	21,905
>40% Scenario	308.6	1,375.0	88.6	448.6	142.3	21,695
>50% Scenario	302.5	1,349.1	88.5	449.0	142.4	21,412
>60% Scenario	294.6	1,317.1	90.8	461.4	146.2	21,294
>70% Scenario	289.2	1,288.6	91.7	466.7	148.0	21,054
>80% Scenario	281.7	1,254.8	91.6	465.9	147.7	20,655
>90% Scenario	273.2	1,216.0	92.5	471.1	149.5	20,621

4.5 Discussion

Reversing biodiversity loss and achieving Kenya's commitments to the Convention on Biological Diversity, the Bonn Challenge and the Sustainable Development Goals will depend on wholesale changes to how we manage land, paired with parallel programmes to conserve and restore ecosystems and ecological processes. Achieving these changes involves understanding and mitigating a whole range of trade-offs between biodiversity conservation, socio-economic and climate goals (Milner-Gulland et al., 2021). Systematic conservation planning was explicitly designed to investigate these issues, as it provides a transparent framework for understanding the impacts of different decisions (Margules & Pressey, 2000). As many of these decisions will be taken by nation states as part of developing national policies and plans, in this analysis, we carried out the first country-level prioritisation to understand the implications of different scenarios for conservation and restoration. This section discusses the contribution of existing conservation areas, where additional priorities for conservation and restoration lie, the trade-offs between the conservation of habitat fragments and incorporating them with restoration, and the broader implications for area-based conservation.

4.5.1 Existing conservation areas and further conservation priorities

Kenya's PAs meet targets for over a third of the country's vegetation types and Near-Threatened and Threatened vertebrate species. They also contain a seventh of its intact carbon and more than half its clean water, showing that this network plays a nationally important role for ecosystem service provision (Dudley et al., 2014). National Parks achieved targets for all Afroalpine vegetation types and one montane forest type, as well as five mammals, five birds, and two reptiles. This mirrors the pattern found in a number of other countries (Joppa & Pfaff, 2009; Venter et al., 2018), where National Parks are placed in 'high and far' locations on land with lower agricultural value. Including Other state-run PAs met targets for six mammal species and three birds. Forest Reserves contain nearly double the amount of carbon and water as National Parks and Other state-run PAs, and met targets for two more forest types and mangroves, as well as several of the mammal, bird, and reptile species. Including Wildlife Conservancies has a large impact, meeting targets for four savanna and one desert vegetation types, as well as six savanna mammals and more than doubling the number of birds meeting targets, including most vultures. The role of these conservancies in filling many of the representation gaps shows the benefits of encouraging different types of PA governance, especially when it comes to savanna ecosystems and species. These woodland and grassland vegetation types are highly valued for livestock and so are generally under-represented in state-owned PAs in East Africa (Riggio et al., 2019) and around the world (Brooks et al., 2004), so our results show the benefit of community-based conservation areas for filling these representation gaps (O'Bryan et al., 2020).

Despite this PA network containing a range of different PA governance types and covering nearly 20% of the country, important gaps in the ecosystems and species it represents still remain. In particular, the network fails to meet targets for two thirds of forest types, three quarters of savanna types, all wetlands and more than half of the near-threatened and threatened species, including all the amphibians (Figure 4.3). One reason for this is that these PAs include large areas of land that need restoring, and our analysis showed that by restoring 10,972 km^2 of land within PAs, it was possible to meet the targets for 6 species and to contribute to the targets for another 40 species. Given that this land is already designated for conservation, it is likely that restoring these habitats would be more politically feasible than designating new conservation and restoration areas. However, our first analysis (the >0% Scenario) showed that much more land is needed outside these PAs to meet the targets, with an additional 11% of the country requiring conservation attention and 4% requiring habitat restoration.

The additional conservation areas were either rangeland systems in Kenya's north, east and south, forest fragments in its centre, west, and along the coast or wetland areas along the shores of Lake Victoria. Our analysis also showed that restoration was essential to meet representation targets that were unachievable by conserving intact habitat. This was the case for nearly half of Kenya's near-threatened and threatened vertebrate species, including all amphibians, as well as eight forest vegetation forest types and one savanna type. The resultant network in the >0% Scenario shows the key conservation and restoration priority areas for Kenya, which would increase the area of land under conservation management from 19% in the current PA network to 35%. This is a very large increase, exceeding the ambitious 30 x 30 target that is being widely promoted (Waldron et al., 2020), but reflecting that Kenya has high levels of biodiversity and endemism and so may need to exceed the international targets that are based on conserving a specific percentage of the globe (Butchart et al., 2015). However, it should be noted that the outputs from all target-based spatial prioritisations are dependent on the values and data that underpin them (Smith et al., 2019b), and we used tried and tested approaches from the literature for setting the vegetation type and species targets (Rodrigues et al., 2004; SANBI, 2018), rather than consulting with the relevant stakeholders from the Kenyan government and civil society.

This means that the results of the >0% Scenario should only be seen as indicative, although there are some broad lessons from the literature that still need to be noted. In particular, implementation should fundamentally be about engaging with the people living within these areas to promote human-wildlife coexistence. Creating these additional conservation areas will require policies that recognise the rights, agency, and tenure of the four million people living in the priority areas we identify. The eight million people living within the restoration priority areas will also need to benefit from restoration activities. Restoring within PAs may be done through active restoration or cheaper means such as assisted regeneration (e.g. Chazdon et al., 2020), but this assumes the lands will be devoid of people afterwards. This will not be the case outside of PAs, where displacement would have negative societal consequences (Barr & Sayer, 2012) and may lead to habitat and biodiversity loss elsewhere (Latawiec et al., 2015). However, with Kenya's population urbanising and rural areas often heavily degraded, restoration and conservation will be about promoting stewardship through financial and other incentivisation, as well as monitoring the equity and effectiveness of restoration as well as conservation projects.

Encouragingly, our results show that the species that depend on restoration to meet their targets (small forest mammals, riparian or forest birds, forest and alpine reptiles, and endemic amphibians) are not those that are likely to come into conflict with people through crop-raiding or other negative impacts, suggesting that co-existence is possible. Thus, creating incentives through agri-environment schemes to develop multifunctional landscapes of indigenous agroforestry or silvopasture would benefit communities and the species we identified as needing restoration. Given that payments for ecosystem services are more practical than land easements in Kenya (Curran et al., 2016), both conservation and restoration must incentivise human-wildlife co-existence to produce equitable solutions.

4.5.2 Trade-offs between conservation and restoration

Many of the patches of natural vegetation selected in the >0% Scenario are small and isolated, especially in the agricultural landscapes in the southwest of Kenya. In such places, it could be better ecologically to focus on restoring these vegetation types, as these larger and better-connected patches would be more valuable for biodiversity. They would also incorporate small protected areas, which are important in their own right (Volenec & Dobson, 2020) and improve connectivity in a country with few intact linkages between PAs (Ward et al., 2020). One obvious disadvantage with this approach is that restoration is a slow process, sometimes taking decades before sensitive species colonise restored patches (Jackson & Sax, 2010), although some of these species are also likely to be absent from the existing small habitat patches. There are also potential benefits on top of increasing long-term ecological connectivity, in terms of the number of people impacted and carbon captured, but there are also likely to be negative impacts in terms of financial cost. This is why we ran the different scenarios to investigate for the first time the trade-offs when deciding whether to meet representation targets by conserving small habitat patches or restoring new areas.

One obvious result from the ten scenarios is that excluding small patches of habitat from the Conserve zone has relatively small impacts on the spatial patterns of the two zones identified by Marxan with Zones. Even when comparing the >0% Scenario with the >90% Scenario, where the latter excludes planning units that contain 10% or more of landcover types that cannot be conserved or restored, the maps look similar. The main differences are that land in coastal, central, western and southern Kenya is added to the Restore zone and more land is added to the Wider landscape zone because targets can be met in less space by restoring big patches of habitat, rather than conserving lots of small patches (Figure 4.4b). This trend with the total area of the Conserve and Restore zones is not linear, though, so that the total area initially increases from 200,282 km^2 in the >0% Scenario to 202,058 km^2 in the >30% Scenario, before declining to 197,783 km^2 in the >90% Scenario (Figure 4.6b). Perhaps most significant is the cost of the portfolio, which is driven by the fact that restoring land is three orders of magnitude more expensive than conserving it. Thus, while the estimated management costs of the existing PA network are $106 million, the predicted management costs in >0% Scenario is $134 million for the Conserve zone and $7.7 billion for the Restore zone (Figure 4.6a). This means that for the other scenarios, the cost of increasing the land for restoration is approximately 250 times more than the savings from reducing the land for conservation. This pattern is to be expected, given the well known difference in cost between conserving and restoring (Possingham et al., 2015), and emphasises the importance of conserving existing large patches of habitat (Allan et al., 2020). However, the situation is more complicated in our scenarios, where the existing habitat is fragmented and less ecologically viable.

Our scenarios showed that when comparing the >0% Scenario and >90% Scenario, the number of habitat patches dropped by 831%, and the mean patch size increased by 821% to 376 km^2. Given the increased management costs, it is probably more feasible to consider the >30% Scenario, which had 1,333 patches with a mean area of 153 km^2. Reducing fragmentation to these levels would have obvious ecological benefits and help achieve the connectivity targets that are likely to be part of the post-2020 commitments. This >30% Scenario would also decrease the number of people located in the Conserve zone by 490,000, although it would increase the number of people in the Restore zone by 1.22 million, so the social impacts would only be lower if restoration was done in a way that had fewer negative effects. The impacts on ecosystem services are also mixed. For example, the potential clean water in the portfolio decreased from 23,090 million m^3 in the >0% Scenario to 20,621 million m^3 in the >90% Scenario, although this was due to the scenarios with more intact habitat covering less area and including more land-uses that currently pollute water. As habitats are restored and protected, they will provide more clean water and sequester and store atmospheric carbon. The carbon gains rapidly increased from 61 Mt in the >0% Scenario to 140 Mt in the >30%, then decreased slowly to 150 Mt in the >90% Scenario. This represents more than Kenya's commitment to reduce its emissions by 46 Mt by 2030, with greater gains possible if interventions promote the rapid accumulation of soil carbon.

4.5.3 Implications for area-based conservation

We present integrated conservation and restoration networks that meet the CBD's "30 by 30" target and contributes three- (>0% Scenario) to four-fifths (>90% Scenario) of Kenya's Bonn Challenge commitment. With decentralised restoration already happening in Kenya (Smucker et al., 2020) and the wider push for restoration (Chazdon et al., 2020; Milner-Gulland et al., 2021; Strassburg et al., 2020), governments must integrate restoration and conservation in spatial planning frameworks that align these with other land-uses. These must also incorporate climate change, ensuring PA connectivity and adaptative management promote human and ecosystem resilience (Elsen et al., 2020; Maxwell et al., 2020). Another key challenge will be diversifying funding mechanisms within and beyond PAs and OECMs. For instance, tourism can support local livelihoods (Naidoo et al., 2019). However an over-reliance on it is short-sighted; something made apparent in Africa during the COVID-19 pandemic (Lindsey et al., 2020). This is also exacerbated by the global decline in conservation funding (Convention on Biological Diversity, 2014). Africa exhibits diverse conservation practices, including less tourism-focused, more financially resilient approaches centring on pastoralism or other cultural values (Bauer et al., 2021) and management partnerships with funding agencies (Baghai et al., 2018). Therefore, multiple strategies that promote self-reliance will still require support.

Local communities are at the forefront of restoration, conservation and broader environmental action. Ensuring their rights, agency, and self-determination at both site-level governance and in larger-scale socioeconomic contexts is vital to promoting environmental stewardship (Armitage et al., 2020). This will require incentivisation and context-dependent strategies at these local scales, but also the big picture thinking our scenarios provide. Integrating restoration with equitable and effective conservation areas and wider multifunctional landscapes is a way to produce better conservation outcomes (Sayer et al., 2013; Zafra-Calvo & Geldmann, 2020). However, the challenges of mainstreaming conservation and implementing such plans mean that government agencies should lead these processes, building on our adaptable national-level analyses and benefiting from the insights provided by this systematic conservation planning approach.

4.6 Acknowledgements

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4.7 Supplementary information

S4.1 Mapping the prioritisation features and setting targets

Potential natural vegetation types

We combined data on potential natural vegetation types (van Breugel et al., 2015) with a global map of terrestrial habitat types (Jung et al., 2020) and plantation data from the Global Forest Watch web portal (Harris et al., 2021) to map the intact and restorable extent of each vegetation type. We first identified a crosswalk to relate which vegetation types corresponded to which habitat types (Table S4.1). We then considered potential natural vegetation types intact if they were not arable land, rural gardens, urban areas, or plantations. Wooded or forest natural vegetation types were intact if they were not savannah, shrubland, grassland or pasture. All vegetation types except deserts were intact if they were not desert. Based on this, the extent of each potential natural vegetation type that was not intact was considered restorable. We then ran the outputs through a pixel sieve to eliminate patches below 10 hectares as there were small pixel relics in the Jung et al. (2020) layer. This resulted in 36 natural vegetation type prioritisation features (Figure S4.1).

Table S4.1: The crosswalk used to relate which Kenyan potential natural vegetation types as mapped by van Breugel et al., (2015) correspond to which International Union on the Conservation of Nature (IUCN) terrestrial habitat types as mapped by Jung et al., (2020)

Natural vegetation type	IUCN habitat (level 1)	IUCN habitat (level 2)
Acacia tortilis wooded grassland and woodland	Savannah	Savannah - Dry
Acacia-Commiphora deciduous wooded grassland + Combretum wooded grassland	Savannah	Savannah - Dry
Acacia-Commiphora stunted bushland	Shrubland	Shrubland - Subtropical-tropical dry
Afroalpine vegetation	Shrubland	Shrubland - Subtropical-tropical high altitude
Afromontane bamboo	Forest	Forest - Subtropical-tropical moist montane
Afromontane desert	Desert	Rocky Areas
Afromontane dry transitional forest	Forest	Forest - Subtropical-tropical moist montane
Afromontane moist transitional forest	Forest	Forest - Subtropical-tropical moist montane
Afromontane rain forest	Forest	Forest - Subtropical-tropical moist montane
Afromontane undifferentiated forest	Forest	Forest - Subtropical-tropical moist montane
Catena of North Zambezian Undifferentiated woodland + edaphic grassland on drainage-impeded or seasonally flooded soils	Grassland	Grassland - Subtropical-tropical seasonally wet or flooded
Climatic grasslands	Grassland	Grassland - Subtropical-tropical dry
Coastal mosaic	Forest	Forest - Subtropical-tropical dry
Complex of Afromontane undifferentiated forest with wooded grasslands and evergreen or semi-evergreen bushland and thicket at lower margins	Forest	Forest - Subtropical-tropical moist montane
Desert	Desert	Desert - Hot
Dry combretum wooded grassland	Savannah	Savannah - Dry
Edaphic grassland on drainage-impeded or seasonally flooded soils	Grassland	Grassland - Subtropical-tropical seasonally wet or flooded
Edaphic grassland on drainage-impeded or seasonally flooded soils + freshwater swamp	Grassland	Grassland - Subtropical-tropical seasonally wet or flooded

Edaphic grassland on volcanic soils	Savannah	Savannah - Dry
Edaphic wooded grassland on drainage-impeded or seasonally flooded soils	Grassland	Grassland - Subtropical-tropical dry
Evergreen and semi-evergreen bushland and thicket	Savannah	Savannah - Dry
Freshwater swamp	Wetlands (inland)	Wetlands (inland)
Halophytic vegetation	Forest	Forest - Subtropical-tropical dry
Lake Victoria drier peripheral semi-evergreen Guineo-Congolian rain forest	Forest	Forest - Subtropical-tropical moist lowland
Lake Victoria transitional rain forest	Forest	Forest - Subtropical-tropical moist montane
Mangrove	Forest	Forest - Subtropical-tropical mangrove vegetation
Moist Combretum wooded grassland	Savannah	Savannah - Dry
Montane Ericaceous belt	Shrubland	Shrubland - Subtropical-tropical high altitude
Palm wooded grassland	Forest	Forest - Subtropical-tropical dry
Riverine wooded vegetation	Forest	Forest - Subtropical-tropical dry
Sand	Desert	Desert - Hot
Single-dominant Hagenia abyssinica forest	Forest	Forest - Subtropical-tropical moist montane
Somalia-Masai Acacia-Commiphora deciduous bushland and thicket	Savannah	Savannah - Dry
Somalia-Masai Acacia-Commiphora shrubland + Somalia-Masai semi-desert grassland and shrubland	Shrubland	Shrubland - Subtropical-tropical dry
Somalia-Masai semi-desert grassland and shrubland	Grassland	Grassland - Subtropical-tropical dry
Transitional zone Somalia-Masai Acacia-Commiphora deciduous bushland and thicket and Dry Combretum wooded grassland	Savannah	Savannah - Dry
Upland Acacia wooded grassland	Savannah	Savannah - Dry
Water bodies	Wetlands (inland)	Wetlands (inland) - Permanent freshwater lakes
Zanzibar-Inhambane lowland rain forest	Forest	Forest - Subtropical-tropical dry

Table S4.2: The natural vegetation type prioritisation features and the calculations used to determine their representation target. Nine natural vegetation types that were less than 20% intact required restoration. Values are in hectares (ha). 'Target met by PAs' describes whether the target was met by protected areas (PAs) as (i) National Parks, (ii) Other state-run PAs, (iii) Forest Reserves and (iv) Wildlife Conservancies.

Natural vegetation type	Total potential area (ha)	Proportion intact (%)	Intact extent conservation target (ha)	Restorable extent restoration target (ha)	Target met by PAs
Acacia tortilis wooded grassland and woodland	345,214	83.74%	69,043	-	-
Acacia-Commiphora deciduous wooded grassland + Combretum wooded grassland	460,907	81.72%	92,182	-	-
Acacia-Commiphora stunted bushland	12,193,919	89.82%	2,438,781	-	-
Afroalpine vegetation	22,719	99.18%	4,544	-	i
Afromontane bamboo	370,057	55.32%	74,012	-	i, ii, & iii
Afromontane desert	4,763	100.00%	953	-	i
Afromontane dry transitional forest	406,230	5.39%	21,767	59,479	-
Afromontane moist transitional forest	299,862	3.08%	9,217	50,755	-
Afromontane rain forest	842,902	21.79%	168,565	-	i, ii, & iii
Afromontane undifferentiated forest	2,560,142	17.56%	449,396	62,626	-
Climatic grasslands	38,099	99.46%	7,620	-	i-iv
Coastal mosaic	2,145,432	28.84%	429,087	-	-
Desert	1,132,448	66.13%	226,490	-	-
Dry combretum wooded grassland	1,071,190	33.00%	214,237	-	-
Edaphic grassland on drainage-impeded or seasonally flooded soils	1,634,715	99.19%	10,159	-	-
Edaphic grassland on drainage-impeded or seasonally flooded soils + freshwater swamp	50,792	95.80%	326,943	-	-
Edaphic grassland on volcanic soils	14,948	100.00%	2,990	-	-

Edaphic wooded grassland on drainage-impeded or seasonally flooded soils	1,683,201	49.67%	336,639	-	-
Evergreen and semi-evergreen bushland and thicket	3,144,607	67.89%	628,918	-	i-iv
Freshwater swamp	65,409	53.37%	13,082	-	-
Halophytic vegetation	70,955	61.97%	14,191	-	-
Lake Victoria drier peripheral semi-evergreen Guineo-Congolian rain forest	230,439	0.20%	467	45,621	-
Lake Victoria transitional rain forest	335,607	12.48%	41,899	25,223	-
Mangrove	71,125	59.80%	14,225	-	i, ii, & iii
Moist Combretum wooded grassland	942,995	5.23%	49,105	139,494	-
Montane Ericaceous belt	120,689	91.20%	24,138	-	i
Palm wooded grassland	15,456	11.34%	1,752	1,340	-
Riverine wooded vegetation	644,160	8.76%	56,430	72,402	-
Sand	2,662	89.93%	533	-	i-iv
Single-dominant Hagenia abyssinica forest	10,503	76.54%	2,101	-	i
Somalia-Masai Acacia-Commiphora deciduous bushland and thicket	21,726,892	91.12%	4,345,378	-	i-iv
Somalia-Masai Acacia-Commiphora shrubland + Somalia-Masai semi-desert grassland and shrubland	348,670	86.29%	69,734	-	-
Somalia-Masai semi-desert grassland and shrubland	3,576,057	80.27%	715,212	-	-
Transitional zone Somalia-Masai Acacia-Commiphora deciduous bushland and thicket and Dry Combretum wooded grassland	250	100.00%	50	-	-
Upland Acacia wooded grassland	804,285	74.84%	160,857	-	i-iv
Zanzibar-Inhambane lowland rain forest	1,489	0.00%	0	298	-

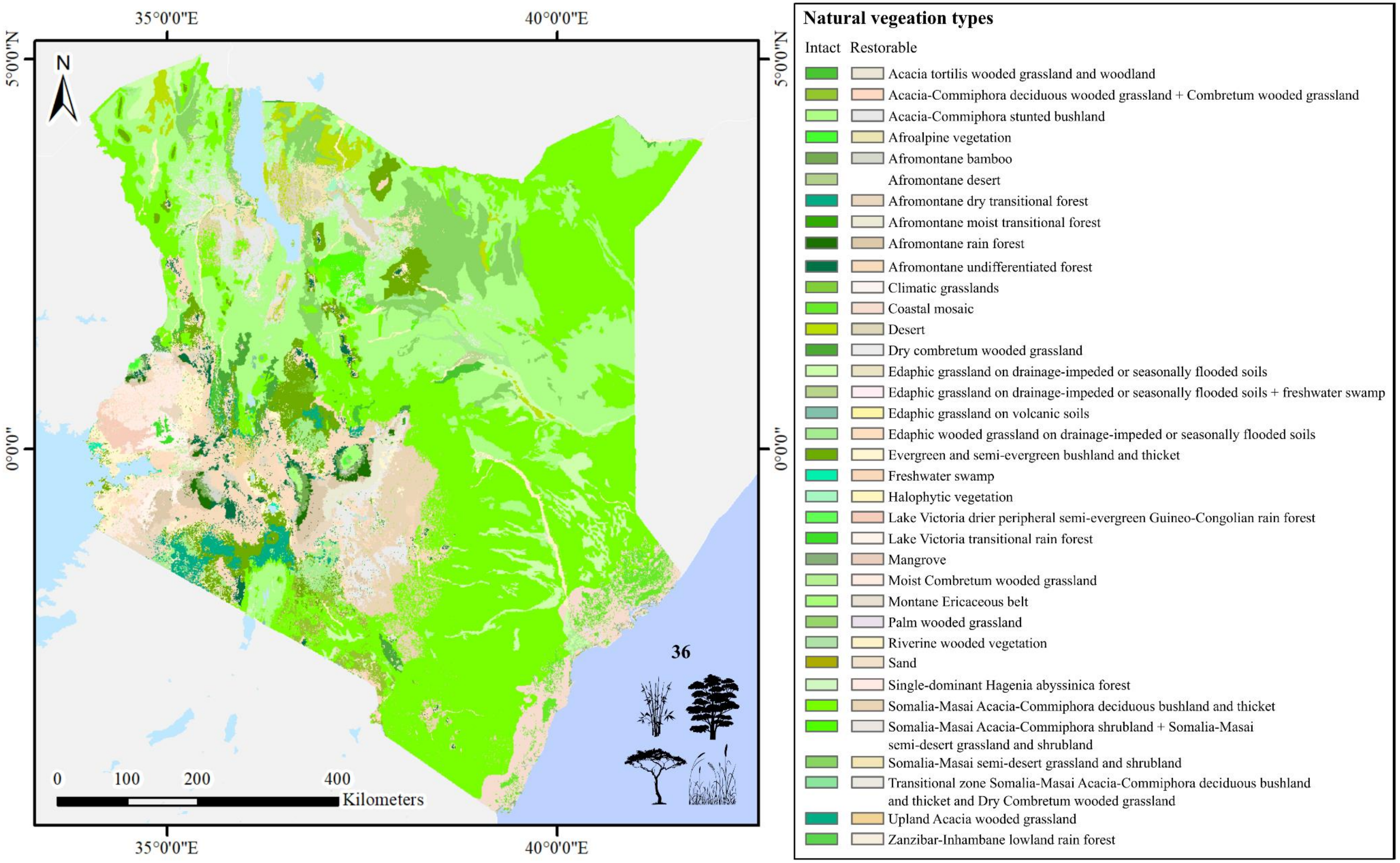

Figure S4.1: The natural vegetation types used as prioritisation features

Near Threatened and Threatened vertebrate species

We used the distribution of terrestrial vertebrates at risk or near risk of extinction as species prioritisation features, mapping their intact and restorable extent. We first selected species via the International Union on the Conservation of Nature (IUCN) Red List web portal classified as Near Threatened, Vulnerable, Endangered, or Critically Endangered. We obtained range maps for mammals, birds, amphibians, and reptiles from the IUCN database (IUCN, 2020) in August 2020. Two reptile species, *Cnemaspis elgonensis* and *Gastropholis prasina*, missing range maps from the IUCN database, were supplemented with data from the Global Assessment of Reptile Distributions (Uetz, 2020). This resulted in 127 Near Threatened and Threatened terrestrial vertebrate species prioritisation features (Table S4.2).

We pre-processed range maps using common practice (Butchart et al., 2015). We first selected species' ranges where they are extant or possibly extant, native or reintroduced, or resident or seasonally present for breeding, non-breeding or during passage whilst migrating (Butchart et al., 2015). We next selected the area of habitat (Brooks et al., 2019) within a species' range where it could potentially exist using data on habitat preference and elevation range. We carried out area of habitat calculation using the 'rredlists' package and the R statistical environment (R Core Team, 2019). We mapped intact species' area of habitat using the Jung et al. (2020) IUCN harmonised habitat map, supplemented with data on exotic timber plantations from the Global Forest Watch (Harris et al., 2021). We were able to select habitat preference to 'level 2' habitats for 115 species, the remaining 12, which had no area of habitat at level 2, were refined to broader 'level 1' habitats. Two species had elevation ranges that led to their having no area of habitat, *Cephalophus silvicultor* and *Grammomys gigas*. Given this we did not refine habitat preference to elevation for these species. We next removed species habitat preferences for open water and artificial habitats, as we wanted to select areas where the conservation of unmodified terrestrial habitats was required. We then used the corresponding potential natural vegetation types (Table S4.1) to calculate the restorable area of habitat within each species' range.

Table S4.3: The International Union for the Conservation of Nature (IUCN) Near Threatened (NT), Vulnerable (VU), Endangered (EN) and Critically Endangered (CR) vertebrate species (N = 127) used as species prioritisation features and their representation target. Their species area target was applied to their Kenya range, where this could not be met by the intact area of habitat (AOH), the remainder was met by their restorable area of habitat (N = 60). 'Target met by PAs' describes whether the target was met by protected areas (PAs) as (i) National Parks, (ii) Other state-run PAs, (iii) Forest Reserves and (iv) Wildlife Conservancies

Binomial name	English name	IUCN category	Taxa	AOH habitat level	Range type	Species-area target (%)	Intact AOH conservation target (ha)	Restorable AOH restoration target (ha)	Target met by PAs
Acinonyx jubatus	Cheetah	VU	Mammal	2	Extant (breeding)	10.00%	1,243,479	-	i
Acinonyx jubatus	Cheetah	VU	Mammal	2	Possibly extant (breeding)	10.00%	3,063,844	-	-
Acrocephalus griseldis	Basra Reed-warbler	EN	Bird	2	Extant (non-breeding)	10.00%	812,037	-	i
Acrocephalus griseldis	Basra Reed-warbler	EN	Bird	2	Extant (passage)	10.00%	811,524	-	i
Adolfus alleni	Alpine meadow lizard	NT	Reptile	2	Extant (breeding)	72.94%	239,711	87,707	-
Afrixalus sylvaticus	n/a	VU	Amphibian	2	Extant (breeding)	66.07%	25,781	39,061	-
Agapornis fischeri	Fischer's lovebird	NT	Bird	2	Extant (breeding)	16.93%	4,669	-	i
Anthreptes reichenowi	Plain-backed sunbird	NT	Bird	1	Extant (breeding)	10.00%	245,107	-	-
Anthreptes reichenowi	Plain-backed sunbird	NT	Bird	1	Possibly Extant (breeding)	41.45%	53,106	-	-
Anthus sokokensis	Sokoke pipit	EN	Bird	2	Extant (breeding)	86.57%	395	38,796	-
Apalis fuscigularis	Taita apalis	CR	Bird	2	Extant (breeding)	100.00%	7,888	190	-
Apalis karamojae	Karamoja apalis	VU	Bird	2	Extant (breeding)	41.45%	10	279	-
Aquila heliaca	Eastern imperial eagle	VU	Bird	2	Extant (non-breeding)	10.00%	18,669	-	-
Aquila heliaca	Eastern imperial eagle	VU	Bird	2	Extant (passage)	10.00%	805,492	-	-
Aquila nipalensis	Steppe eagle	EN	Bird	2	Extant (non-breeding)	10.00%	4,410,668	-	i-iv
Aquila rapax	Tawny eagle	VU	Bird	2	Extant (breeding)	10.00%	5,786,824	-	i-iv
Ardeotis kori	Kori bustard	NT	Bird	2	Extant (breeding)	10.00%	3,633,285	-	i-iv
Arthroleptides dutoiti	Du Toit's torrent frog	CR	Amphibian	2	Extant (breeding)	100.00%	1,606	6,815	-

Atheris desaixi	Ashe's bush viper	EN	Reptile	2	Extant (breeding)	100.00%	20	3,638	-
Balearica pavonina	Black crowned crane	VU	Bird	2	Extant (breeding)	10.00%	195,956	-	-
Balearica regulorum	Grey crowned crane	EN	Bird	2	Extant (breeding)	10.00%	2,511,328	-	i-iv
Bdeogale jacksoni	Jackson's mongoose	NT	Mammal	2	Extant (breeding)	32.11%	731,310	1,002,126	-
Bdeogale omnivora	Sokoke dog mongoose	VU	Mammal	1	Extant (breeding)	45.29%	620,390	547,093	-
Beatragus hunteri	Hirola	CR	Mammal	2	Extant (breeding)	60.32%	638,037	-	-
Bitis worthingtoni	Kenya horned viper	VU	Reptile	2	Extant (breeding)	61.07%	144,115	304,805	-
Boulengerula changamwensis	Changamwensis african caecilian	EN	Amphibian	1	Extant (breeding)	100.00%	10,797	9,812	-
Boulengerula niedeni	Sagalla caecilian	EN	Amphibian	2	Extant (breeding)	100.00%	24	1,216	-
Boulengerula taitana	n/a	EN	Amphibian	2	Extant (breeding)	100.00%	1,045	12,487	-
Bucorvus abyssinicus	Northern ground hornbill	VU	Bird	2	Extant (breeding)	10.00%	1,081,044	-	-
Bucorvus leadbeateri	Southern ground hornbill	VU	Bird	2	Extant (breeding)	10.00%	1,571,513	-	i
Buteo oreophilus	Mountain buzzard	NT	Bird	2	Extant (breeding)	10.00%	748,722	437,397	-
Calamonastides gracilirostris	Papyrus yellow warbler	VU	Bird	1	Extant (breeding)	26.00%	31,417	219,003	-
Callulina dawida	Taita Hills warty frog	CR	Amphibian	2	Extant (breeding)	100.00%	978	8,206	-
Caracal aurata	Golden cat	VU	Mammal	2	Possibly Extant (breeding)	10.00%	555,208	-	i, ii & iii
Cephalophus adersi	Ader's duiker	VU	Mammal	2	Extant (breeding)	73.60%	222,146	32,275	-
Cephalophus silvicultor	Yellow-backed duiker	NT	Mammal	2	Extant (breeding)	10.00%	49,872	-	i, ii & iii
Cephalophus silvicultor	Yellow-backed duiker	NT	Mammal	2	Possibly Extant (breeding)	10.00%	14,859	-	i, ii & iii
Cercocebus galeritus	Tana River mangabey	CR	Mammal	2	Extant (breeding)	100.00%	1,836	10,009	-
Cinnyris usambaricus	Usambara double-collared sunbird	NT	Bird	2	Extant (breeding)	45.71%	6,303	19,056	-
Circaetus fasciolatus	Southern banded snake-eagle	NT	Bird	1	Extant (breeding)	10.00%	806,077	957,213	-
Circus macrourus	Pallid harrier	NT	Bird	2	Extant (non-breeding)	10.00%	5,730,113	-	-
Circus macrourus	Pallid harrier	NT	Bird	2	Extant (passage)	10.00%	700	-	i-iv
Cisticola aberdare	Aberdare cistocola	VU	Bird	1	Extant (breeding)	95.96%	14,278	-	-
Cnemaspis elgonensis	Mt Elgon forest gecko	VU	Reptile	2	Extant (breeding)	39.27%	1,942	19,555	-

Colobus angolensis	Angolan colobus	VU	Mammal	1	Extant (breeding)	10.00%	27,182	-	-
Crocidura allex	East African highland shrew	VU	Mammal	2	Extant (breeding)	68.98%	2,675	-	-
Dendrohyrax validus	Eastern tree hyrax	NT	Mammal	2	Extant (breeding)	40.68%	1,528	16,428	-
Diceros bicornis	Black rhino	CR	Mammal	2	Extant (breeding)	10.00%	5,855,697	-	i-iv
Eidolon helvum	African straw-coloured fruit bat	NT	Mammal	2	Extant (breeding)	10.00%	1,304,644	-	i & ii
Equus grevyi	Grevy's zebra	EN	Mammal	2	Extant (breeding)	26.39%	2,079,612	-	-
Equus quagga	Plains zebra	NT	Mammal	2	Extant (breeding)	10.00%	3,105,665	-	i & ii
Eremomela turneri	Turner's eremomela	NT	Bird	2	Extant (breeding)	100.00%	25,608	3,104	-
Erythrocebus patas	Patas monkey	NT	Mammal	2	Extant (breeding)	10.00%	442,498	-	-
Falco cherrug	Saker falcon	EN	Bird	2	Extant (non-breeding)	10.00%	1,714,072	-	-
Falco concolor	Sooty falcon	VU	Bird	2	Extant (passage)	10.00%	1,460,900	-	i-iv
Falco fasciinucha	Taita falcon	VU	Bird	2	Extant (breeding)	10.00%	1,683,144	-	i-iv
Falco vespertinus	Red-footed falcon	NT	Bird	2	Extant (passage)	10.00%	3,639,394	-	-
Fraseria lendu	Chapin's flycatcher	VU	Bird	2	Extant (breeding)	40.86%	10,337	-	i, ii & iii
Gallinago media	Great snipe	NT	Bird	2	Extant (non-breeding)	10.00%	3,515,926	-	i-iv
Gastropholis prasina	Green keel-bellied lizard	NT	Reptile	2	Extant (breeding)	73.94%	32,619	29,056	-
Geokichla guttata	Spotted ground thrush	EN	Bird	2	Extant (non-breeding)	31.51%	311,538	309,977	-
Giraffa camelopardalis	Giraffe	VU	Mammal	2	Extant (breeding)	10.00%	3,166,529	-	i & ii
Grammomys gigas	Giant thicket rat	EN	Mammal	2	Extant (breeding)	89.32%	117,695	-	-
Gypaetus barbatus	Bearded vulture	NT	Bird	2	Extant (breeding)	10.00%	25,665	-	-
Gyps africanus	White-backed Vulture	CR	Bird	2	Extant (breeding)	10.00%	3,516,328	-	i-iv
Gyps africanus	White-backed Vulture	CR	Bird	2	Possibly Extant (breeding)	10.00%	16,583	-	i
Gyps africanus	White-backed Vulture	CR	Bird	2	Extant (non-breeding)	10.00%	1,149,423	-	i & ii
Gyps rueppelli	Rüppell's vulture	CR	Bird	2	Extant (breeding)	10.00%	2,223,922	-	i-iv
					Extant (non-breeding)	10.00%	1,267,351	-	-
					Extant (passage)	10.00%	26,511	-	i-iv

Hirundo atrocaerulea	Blue swallow	VU	Bird	2	Extant (non-breeding)	10.00%	755,882	111,624	-
Hyaena hyaena	Striped hyena	NT	Mammal	2	Extant (breeding)	10.00%	5,858,842	-	i-iv
Hyperolius cystocandicans	Tigoni reed frog	NT	Amphibian	2	Extant (breeding)	54.59%	68,323	316,170	-
Hyperolius rubrovermiculatus	n/a	EN	Amphibian	1	Extant (breeding)	100.00%	27,921	12,218	-
Kinyongia asheorum	Mount Nyiru bearded chameleon	NT	Reptile	2	Extant (breeding)	100.00%	1,489	73	i, ii & iii
Kinyongia boehmei	Taita blade-horned chameleon	NT	Reptile	2	Extant (breeding)	100.00%	1,069	9,157	-
Kinyongia excubitor	Mt Kenya sentinel chameleon	VU	Reptile	2	Extant (breeding)	96.35%	97,129	17,105	-
Kinyongia tenuis	Usambara Flap-nosed Chameleon	EN	Reptile	2	Extant (breeding)	100.00%	11,336	-	-
Laniarius mufumbiri	Papyrus gonolek	NT	Bird	2	Extant (breeding)	24.99%	9,185	30,257	-
Litocranius walleri	Gerenuk	NT	Mammal	2	Extant (breeding)	10.00%	3,332,750	-	i, ii & iii
Loxodonta africana	African elephant	VU	Mammal	2	Extant (breeding)	10.00%	915,335	-	i
Loxodonta africana	African elephant	VU	Mammal	2	Possibly Extant (breeding)	10.00%	158,534	-	i, ii & iii
Lycaon pictus	African wild dog	EN	Mammal	2	Extant (breeding)	10.00%	698,087	-	i
Macronycteris vittatus	Striped leaf-nosed bat	NT	Mammal	2	Extant (breeding)	10.00%	1,143,896	-	i
Malacochersus tornieri	Pancake tortoise	CR	Reptile	2	Extant (breeding)	29.54%	827,121	-	-
Mertensophryne lonnbergi	Lönnbergs toad	VU	Amphibian	2	Extant (breeding)	62.76%	332,619	280,069	-
Montatheris hindii	Kenya montane viper	NT	Reptile	2	Extant (breeding)	71.34%	44,116	-	-
Necrosyrtes monachus	Hooded vulture	CR	Bird	2	Extant (breeding)	10.00%	2,367,536	-	i-iv
Neophron percnopterus	Egyptian vulture	EN	Bird	2	Extant (breeding)	10.00%	78,731	-	-
Neophron percnopterus	Egyptian vulture	EN	Bird	2	Extant (non-breeding)	10.00%	4,938,319	-	i-iv
Neotis denhami	Denham's bustard	NT	Bird	2	Extant (breeding)	10.00%	824,306	-	i-iv
Numenius arquata	Eurasian curlew	NT	Bird	2	Extant (non-breeding)	10.00%	6,295	-	-
Oryx beisa	Beisa oryx	EN	Mammal	2	Extant (breeding)	10.00%	3,756,266	-	i, ii & iii
Otomops harrisoni	Harrison's large-eared giant mastiff bat	VU	Mammal	2	Extant (breeding)	10.00%	4,140,244	-	i-iv
Otomys barbouri	Barbour's vlei rat	EN	Mammal	2	Extant (breeding)	100.00%	7,433	-	i & ii
Otus ireneae	Sokoke scopps owl	EN	Bird	1	Extant (breeding)	100.00%	26,958	1,234	-

Panthera leo	Lion	VU	Mammal	2	Extant (breeding)	10.00%	723,658	-	i
Panthera pardus	Leopard	VU	Mammal	2	Extant (breeding)	10.00%	3,120,129	-	i & ii
Phataginus tricuspis	White-bellied pangolin	EN	Mammal	2	Extant (breeding)	10.00%	38,804	146,760	-
Phrynobatrachus irangi	Irangi puddle frog	EN	Amphibian	2	Extant (breeding)	100.00%	1,114	951	-
Phrynobatrachus kinangopensis	Kinangop river frog	VU	Amphibian	2	Extant (breeding)	59.40%	251,492	461,127	-
Phrynobatrachus ungujae	n/a	EN	Amphibian	2	Extant (breeding)	100.00%	17,528	2,053	-
Piliocolobus rufomitratus	Tana River red colobus	CR	Mammal	2	Extant (breeding)	100.00%	2,773	16,224	-
Ploceus golandi	Clarke's weaver	EN	Bird	2	Extant (breeding)	100.00%	22,861	46,877	-
Poeoptera femoralis	Abbott's starling	VU	Bird	2	Extant (breeding)	73.55%	126,218	110,677	-
Polemaetus bellicosus	Martial eagle	EN	Bird	2	Extant (breeding)	10.00%	5,747,980	-	i-iv
Prionops poliolophus	Grey-crested helmetshrike	NT	Bird	2	Extant (breeding)	34.44%	338,440	264,331	-
Psittacus erithacus	Grey parrot	EN	Bird	2	Extant (breeding)	10.00%	85,592	40,798	-
Pternistis atrifrons	Black-fronted francolin	EN	Bird	1	Possibly Extant (breeding)	85.19%	245	-	-
Redunca fulvorufula	Mountain reedbuck	EN	Mammal	2	Extant (breeding)	10.00%	955,528	-	-
Rhinolophus deckenii	Decken's horseshoe bat	NT	Mammal	2	Extant (breeding)	14.60%	21	3,633	-
Rhynchocyon chrysopygus	Golden-rumped sengi	EN	Mammal	2	Extant (breeding)	79.13%	71,954	208,684	-
Sagittarius serpentarius	Secretary bird	EN	Bird	2	Extant (breeding)	10.00%	5,405,248	-	i-iv
Scleroptila elgonensis	Elgon francolin	NT	Bird	2	Extant (breeding)	52.59%	655,067	229,787	-
Scleroptila streptophora	Ring-necked francolin	NT	Bird	2	Extant (breeding)	12.24%	1,857	1,086	-
Sheppardia gunningi	East coast akalat	NT	Bird	2	Extant (breeding)	33.69%	105,637	325,724	-
Smutsia gigantea	Giant ground pangolin	EN	Mammal	2	Extant (breeding)	10.00%	6,057	12,137	-
Smutsia temminckii	Temminick's pangolin	VU	Mammal	2	Extant (breeding)	10.00%	3,675,089	-	i-iv
Stephanoaetus coronatus	Crowned eagle	NT	Bird	2	Extant (breeding)	10.00%	829,880	-	i
Streptopelia reichenowi	White-winged collared-dove	NT	Bird	2	Extant (breeding)	17.84%	15,477	-	-
Struthio molybdophanes	Somali ostrich	VU	Bird	2	Extant (breeding)	10.00%	2,415,323	-	i & ii
Suncus aequatorius	Taita dwarf shrew	EN	Mammal	2	Extant (breeding)	100.00%	271	3,332	-

Syncerus caffer	African buffalo	NT	Mammal	2	Extant (breeding)	10.00%	4,715,725	-	i-iv
Taphozous hildegardeae	Hildegarde's tomb bat	EN	Mammal	2	Extant (breeding)	38.03%	167,446	505,060	-
Tauraco fischeri	Fischer's turaco	NT	Bird	1	Extant (breeding)	36.56%	520,219	250,719	-
Terathopius ecaudatus	Bateleur	EN	Bird	2	Extant (breeding)	10.00%	5,747,974	-	i-iv
Thelotornis usambaricus	Usambara vine snake	VU	Reptile	2	Extant (breeding)	79.16%	3,902	8,608	-
Thrasops schmidti	Schmidt's bold-eyed tree snake	EN	Reptile	2	Extant (breeding)	73.38%	15,184	420,233	-
Torgos tracheliotos	Lappet-faced vulture	EN	Bird	2	Extant (breeding)	10.00%	2,829,284	-	i & ii
Torgos tracheliotos	Lappet-faced vulture	EN	Bird	2	Extant (non-breeding)	10.00%	2,850,359	-	i-iv
Tragelaphus eurycerus	Bongo	NT	Mammal	2	Extant (breeding)	10.00%	53,471	-	i & ii
Tragelaphus imberbis	Lesser kudu	NT	Mammal	2	Extant (breeding)	10.00%	4,473,567	-	i-iv
Trigonoceps occipitalis	White-headed vulture	CR	Bird	2	Extant (breeding)	10.00%	1,700,188	-	i & ii
Trigonoceps occipitalis	White-headed vulture	CR	Bird	2	Possibly Extant (breeding)	10.00%	1,616,504	-	i-iv
Trioceros kinangopensis	Aberdare dwarf chameleon	NT	Reptile	2	Extant (breeding)	100.00%	339	-	i
Trioceros marsabitensis	Marsabit one-horned chameleon	NT	Reptile	1	Extant (breeding)	100.00%	5,783	974	-
Trioceros narraioca	Mount Kulal stump-nosed chameleon	NT	Reptile	2	Extant (breeding)	100.00%	110	303	i, ii & iii
Trioceros schubotzi	Mount Kenya dwarf chameleon	NT	Reptile	2	Extant (breeding)	100.00%	5,432	15	i
Turdus helleri	Taita thrush	CR	Bird	2	Extant (breeding)	100.00%	721	9,005	-
Turdus helleri	Taita thrush	CR	Bird	2	Possibly Extant (breeding)	100.00%	107	125	-
Zosterops kulalensis	Kulal white-eye	NT	Bird	2	Extant (breeding)	99.66%	1	3,745	i, ii & iii
Zosterops silvanus	Taita white-eye	EN	Bird	2	Extant (breeding)	92.94%	1,550	19,721	-

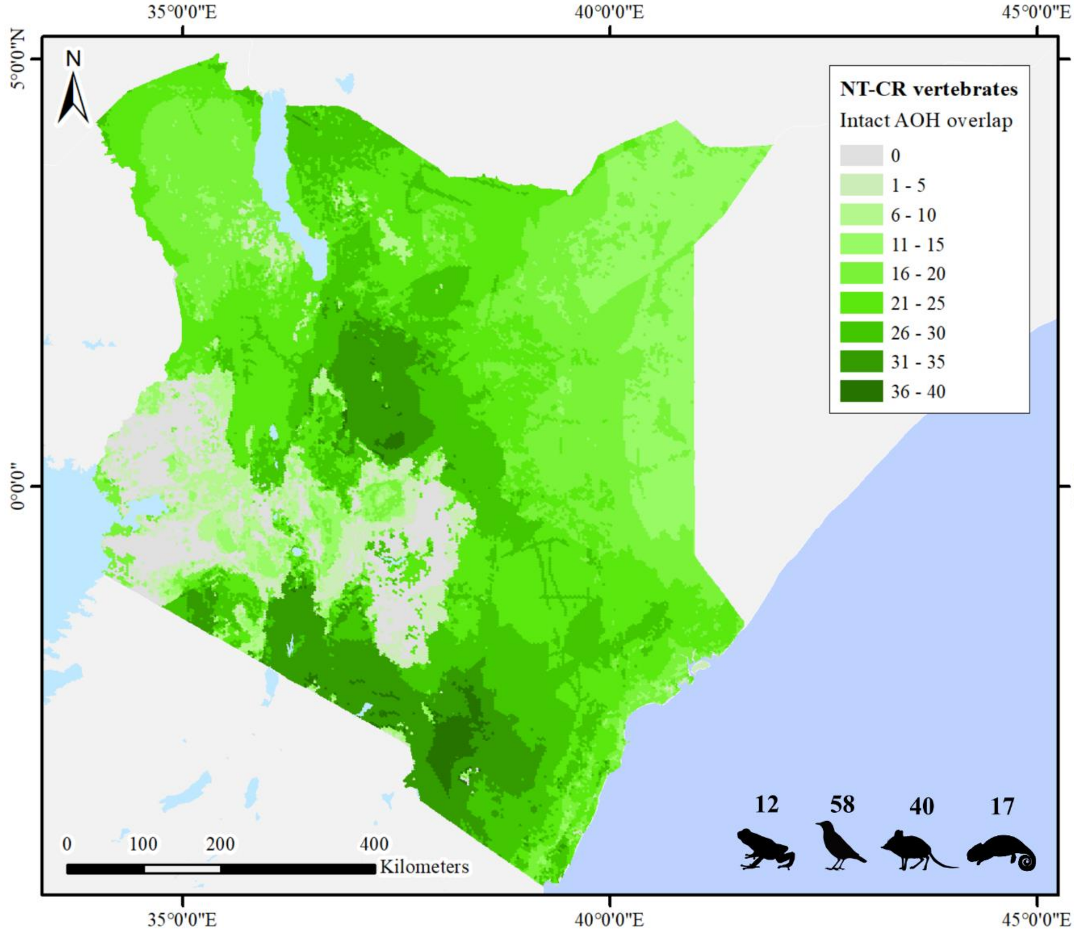

Figure S4.2: Intact area of habitat overlap for the Near Threatened, Vulnerable, Endangered and Critically Endangered (NT-CR) terrestrial vertebrate species (N = 127) used as prioritisation features. Taxa silhouettes show the species contained 12 amphibians, 58 birds, 40 mammals and 17 reptiles

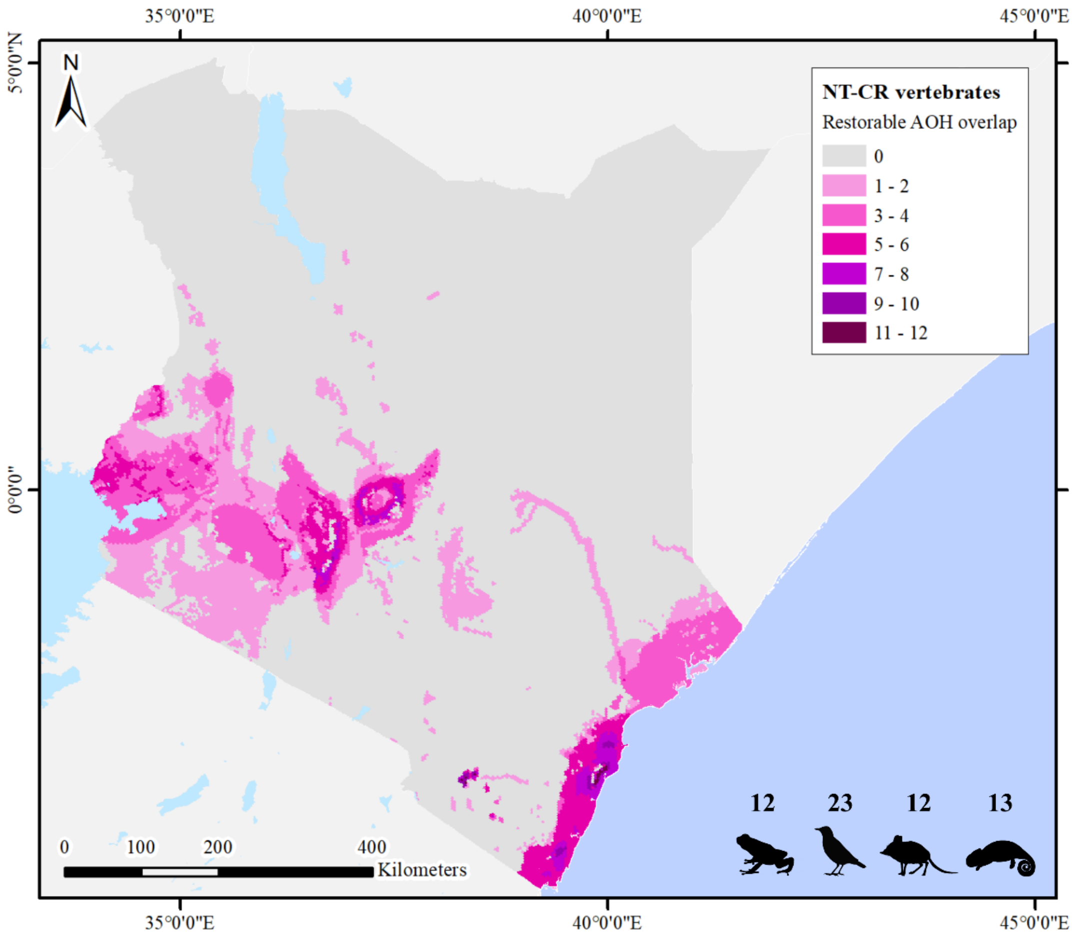

Figure S4.3: Restorable area of habitat overlap for the Near Threatened, Vulnerable, Endangered and Critically Endangered (NT-CR) terrestrial vertebrate species used as prioritisation features (N = 60). Taxa silhouettes show the species contained 12 amphibians, 23 birds, 12 mammals and 13 reptiles

S4.2 Producing the planning system

We selected areas to meet targets by dividing Kenya's land surface into planning units, the basis for compiling data on the abundance of prioritisation features within the planning region (Nhancale & Smith, 2011). We first created hexagonal polygons with an area of 10km^2, as hexagons produce the most reliable results (Ibid). We then clipped these to the national boundary. We next accessed spatial data on the location of Kenya's PAs from the August 2020 version of the World Database on Protected Areas (IUCN and UNEP-WCMC, 2021) and updated Forest Reserves with more comprehensive spatial data from the Kenya Forest Service and Wildlife Conservancies with data from the Kenya Wildlife Conservancies Association. These were handled according to best practice guidelines (https://www.protectedplanet.net/c/calculating-protected-areacoverage). We removed marine protected areas and lakes over 10 km^2 due to our focus on terrestrial systems. We finally dissolved and overlaid protected areas to remove overlaps using the hierarchy i) National Parks, ii) Other state-run PAs, iii) Forest Reserves, and iv) Wildlife Conservancies. We then unioned these to the hexagonal planning units creating the final layer of 64,445 planning units, which divided Kenya into hexagons and sub-sections that matched protected area boundaries (Figure 4.1).

We calculated the cost for including a planning unit in the Conserve zone as the conservation management costs of intact vegetation types and secondary habitats, as these have value for intact species prioritisation features. Costs were taken at ecoregion and biome level (Moore et al., 2004) and accounted for inflation and multiplied by three to make them comparable to the three year period for restoration (Table S4.4). The cost for selecting a planning unit for the Recover zone was the restoration cost for all restorable vegetation types and the conservation management costs of all vegetation types either intact or restored. Restoration costs were based on the most comprehensive database available (de Groot et al., 2012) and scaled to a cost for active tropical forest restoration over three years from the Kenya Forest Service. To calculate the restoration cost for other habitats, we accounted for inflation, then calculated the distance within the range of tropical forest restoration project costs that this value was from the mean. We then used this as a ratio to scale costs for all other biomes (Table S4.4). We finally used the Zonal Statistics function in ArcMap to sum the Conserve and Recover zone costs for each planning unit (Figure S4.5; Figure S4.6) and set the Wider landscape zone cost as \$ 0.001.

Table S4.4: Conservation management and restoration costs over three years for each natural vegetation type. Costs for restoration were scaled to a cost for tropical forest restoration from the Kenya Forest Service for tree planting and monitoring all costs accounted for inflation to United States Dollars (USD) in November 2020.

Natural vegetation type	**Ecoregion as described in Moore et al., (2004) and georeferenced from Dinerstein et al., (2017)**	**Conservation management cost (per hectare in 2020 USD)**	**Biome as described in de Groot et al. (2013)**	**Restoration cost (per hectare in 2020 USD)**
Acacia tortilis wooded grassland and woodland	Southern Acacia-Commiphora bushlands and thickets	$31.44	Woodlands	$568.78
Acacia-Commiphora deciduous wooded grassland + Combretum wooded grassland	Southern Acacia-Commiphora bushlands and thickets	$31.44	Woodlands	$568.78
Acacia-Commiphora stunted bushland	Northern Acacia-Commiphora bushlands and thickets	$29.96	Woodlands	$568.78
Afroalpine vegetation	East African montane moorlands	$6.99	Woodland/Grasslands	$603.15
Afromontane bamboo	East African montane forests	$6.99	Tropical forest	$2,051.26
Afromontane desert	East African montane moorlands	$6.99	Woodland/Grasslands	$603.15
Afromontane dry transitional forest	East African montane forests	$6.99	Tropical forest	$2,051.26
Afromontane moist transitional forest	East African montane forests	$6.99	Tropical forest	$2,051.26
Afromontane rain forest	East African montane forests	$13.46	Tropical forest	$2,051.26
Afromontane undifferentiated forest	East African montane forests	$15.89	Tropical forest	$2,051.26
Climatic grasslands	Northern Acacia-Commiphora bushlands and thickets	$31.44	Woodlands	$568.78
Coastal mosaic	Eastern African lowland forest/grassland mosaics	$29.96	Tropical forest	$2,051.26
Desert	Masai xeric grasslands and shrublands	$29.96	Woodland/Grasslands	$603.15

Dry combretum wooded grassland	Northern Acacia-Commiphora bushlands and thickets	$31.44	Woodlands	$568.78
Edaphic grassland on drainage-impeded or seasonally flooded soils	Northern Acacia-Commiphora bushlands and thickets	$29.96	Woodlands	$568.78
Edaphic grassland on drainage-impeded or seasonally flooded soils + freshwater swamp	Northern Acacia-Commiphora bushlands and thickets	$29.96	Woodlands	$568.78
Edaphic grassland on volcanic soils	Northern Acacia-Commiphora bushlands and thickets	$29.96	Woodlands	$568.78
Edaphic wooded grassland on drainage-impeded or seasonally flooded soils	Southern Acacia-Commiphora bushlands and thickets	$11.72	Woodlands	$568.78
Evergreen and semi-evergreen bushland and thicket	Northern Acacia-Commiphora bushlands and thickets	$13.46	Woodlands	$568.78
Freshwater swamp	Freshwater wetland	$6.99	Inland wetland	$16,028.61
Halophytic vegetation	Saline wetland	$6.99	Inland wetland	$16,028.61
Lake Victoria drier peripheral semi-evergreen Guineo-Congolian rain forest	Victoria Basin forest-savanna	$6.99	Tropical forest/Grasslands	$995.81
Lake Victoria transitional rain forest	East African montane forests	$6.99	Tropical forest/Grasslands	$995.81
Mangrove	East African mangroves	$75.73	Coastal wetlands	$132,877.55
Moist Combretum wooded grassland	Victoria Basin forest-savanna mosaic	$13.46	Tropical forest/Grasslands	$995.81
Montane Ericaceous belt	East African montane moorlands	$13.46	Woodland/Grasslands	$603.15
Palm wooded grassland	Eastern African lowland forest/grassland mosaics	$15.89	Tropical forest/Grasslands	$995.81
Riverine wooded vegetation	Eastern African lowland forest/grassland mosaics	$15.89	Tropical forest/Grasslands	$995.81

Sand	Masai xeric grasslands and shrublands	$6.99	NA	
Single-dominant Hagenia abyssinica forest	East African montane moorlands	$11.72	Tropical forest/Grasslands	$995.81
Somalia-Masai Acacia-Commiphora deciduous bushland and thicket	Northern Acacia-Commiphora bushlands and thickets	$6.30	Woodlands	$568.78
Somalia-Masai Acacia-Commiphora shrubland + Somalia-Masai semi-desert grassland and shrubland	Masai xeric grasslands and shrublands	$6.30	Woodlands	$568.78
Somalia-Masai semi-desert grassland and shrubland	Masai xeric grasslands and shrublands	$6.30	Woodland/Grasslands	$603.15
Transitional zone Somalia-Masai Acacia-Commiphora deciduous bushland and thicket and Dry Combretum wooded grassland	Northern Acacia-Commiphora bushlands and thickets	$6.30	Woodlands	$568.78
Upland Acacia wooded grassland	Southern Acacia-Commiphora bushlands and thickets	$7.03	Woodlands	$568.78
Zanzibar-Inhambane lowland rain forest	Eastern African lowland forest/grassland mosaics	$8.47	Tropical forest/Grasslands	$995.81

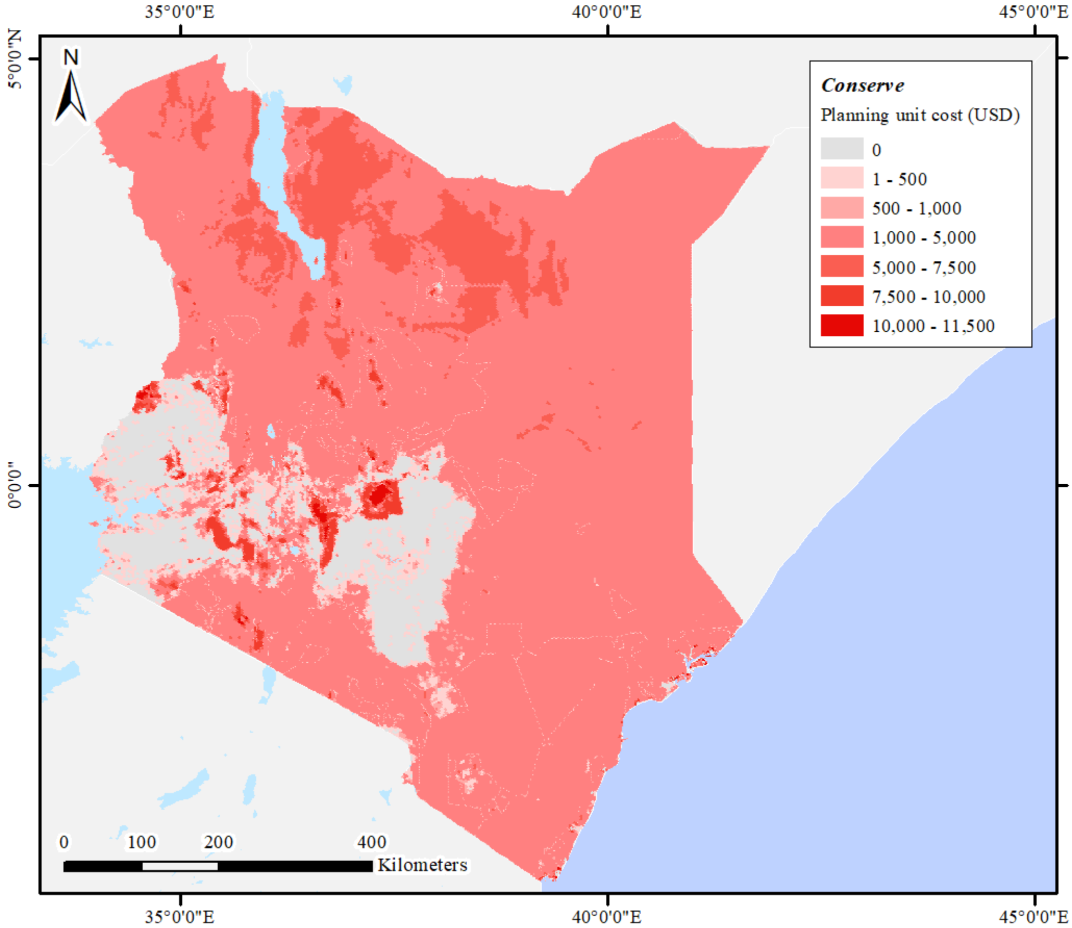

Figure S4.4: The cost for including each planning unit in the Conserve zone, taken as the conservation management costs for intact and secondary natural vegetation types

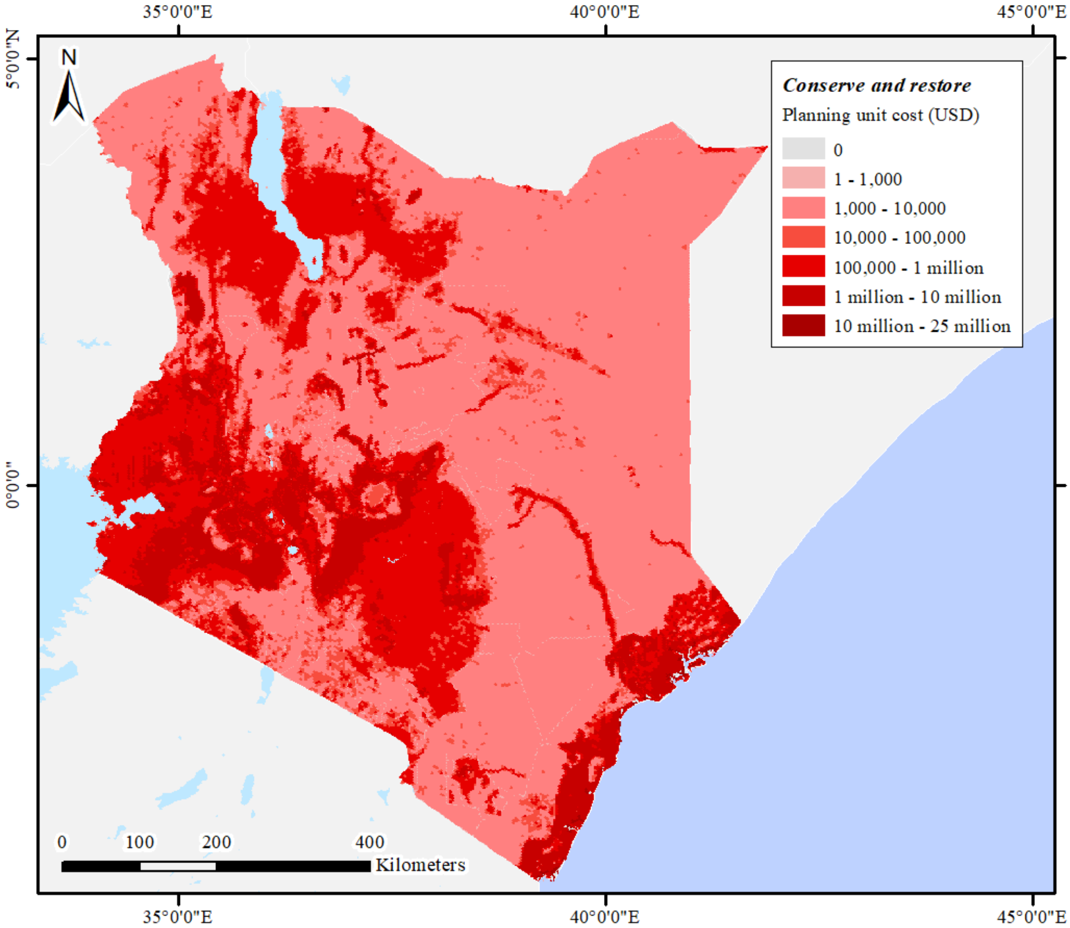

Figure S4.5: The cost for including each planning unit in the Recover zone taken for each natural vegetation type as the restoration cost and the conservation management cost of all restored and intact vegetation types

S4.3 Sensitivity analysis and calibration

We undertook sensitivity analysis and calibration using standard approaches (Serra et al., 2020). We calibrated the number of iterations looking for the optimal trade-off between run time based on the number of iterations and the median score across 20 runs. We varied the number of iterations from 1,000,000 to 2,000,000,000 (see Figure S4.6). The score began to level off after 100,000,000 iterations. We therefore ran further calibrations with 200,000,000 iterations and 20 runs and final runs with 1,000,000,000 iterations and 100 runs.

We calibrated the feature penalty factor, also known as the species penalty factor (Serra et al., 2020), by starting at one and then increasing it by an order of magnitude up to 1,000,000. We found all targets were met with a factor of 10,000 but reduced the factor to 85, at which all targets were met except for five restoration features that were above 99.9% met. We then set a 'MISSLEVEL' of 0.999, so the optimisation algorithm considered these targets met. We did this to avoid having an excessively high factor. However, in the further restoration scenarios, ten species intact extent prioritisation features which required selection of all their available AOH did not meet their targets. We, therefore, increased these feature penalty factors until the target was met. This resulted in eight having a factor of 170, one a factor of 255 and one a factor of 340.

We calibrated the zone boundary costs, also known as the boundary length modifier, to select viable patch sizes that were not too fragmented (Ball et al., 2009). We started by calibrating the boundary cost between the Conserve zone and the Wider landscape zone until an appropriate clustering was observed. This resulted in a zone boundary cost of 0.45 between the Conserve zone and the Wider landscape zone. We then used the median cost difference between PUs selected for the Conserve zone and the Recover zone to set a boundary cost for the Recover zone and the Wider landscape zone. This was then adjusted until an appropriate clustering was observed at a zone boundary cost of 165.

We also put constraints on the planning unit allocation process to avoid Marxan with Zones selecting planning units for the wrong zone, which happens when a planning unit contained no features or had the same cost for multiple zones. Therefore, planning units were excluded from the Conserve zone if they had a zone cost equal to the Wider landscape zone and excluded from the Recover zone if they had a zone cost equal to the Conserve zone or the Wider landscape zone.

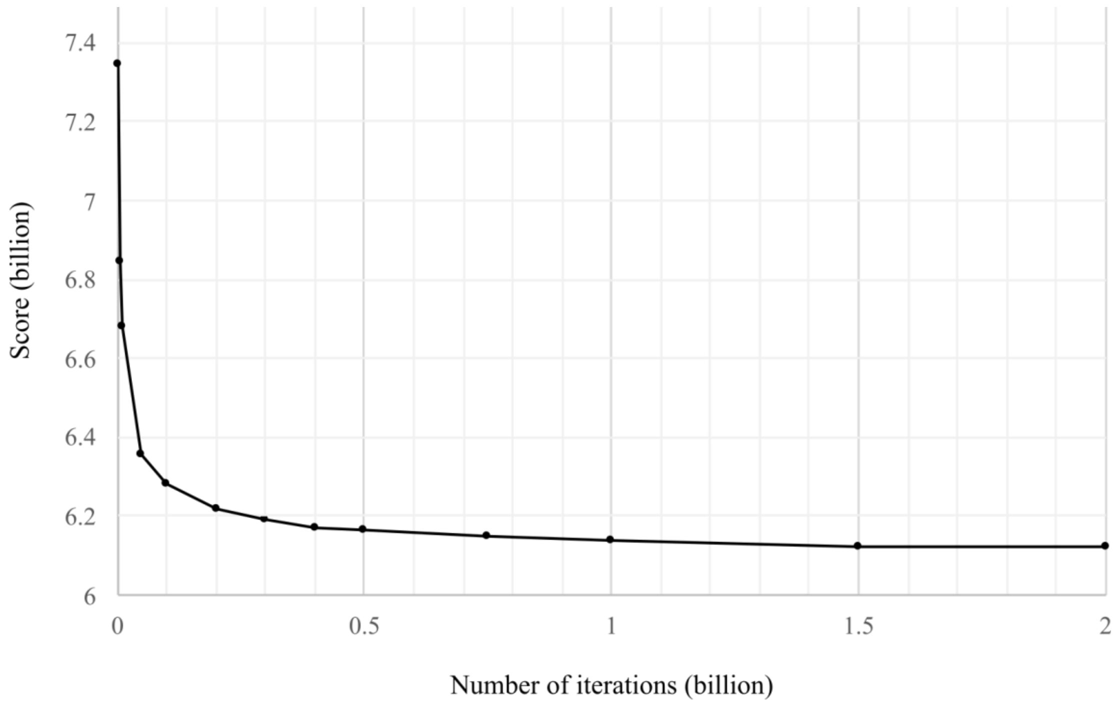

Figure S4.6: The relationship between the number of iterations in each Marxan with Zones run and the portfolio score, the objective function, used for calibrating the number of iterations

S4.4 Prioritisation scenarios

We estimated human population using LandScan's global population distribution model for the year 2019 (Rose et al., 2019). Data represent the ambient population (average over 24 hours) calculated in raster format at a pixel resolution of approximately 835m^2, or 30 by 30 arc seconds (Figure S4.7), which we resampled and recalculated to 1 hectare.

We estimated carbon storage and potential capture using global maps of aboveground and belowground biomass stocks and soil organic carbon density to 1 metre in depth (Soto-Navarro et al., 2020). This dataset created a harmonized global map of carbon storage for the year 2010 using five carbon datasets, providing estimates at a 300 m resolution. These were resampled to our 1 hectare mask as two layers. Firstly, above and belowground carbon stored in biomass (Figure S4.8) and secondly soil carbon stored to a depth of 1 metre (Figure S4.9). These were summed for areas requiring conservation attention and those requiring ecological restoration, assuming that intact primary vegetation types in planning units in the Conserve and Recover zones would not be converted. We then estimated the potential carbon captured under habitat restoration by calculating the mean carbon stored in intact natural vegetation types and assigned these to restorable natural vegetation types (Figures S4.10) and summing this for the areas requiring ecological restoration.

Water provision was estimated using cubic metres of potential clean water provision calculated through WaterWorld (Mulligan, 2013) and Co$ting Nature (Van Soesbergen & Mulligan, 2014). This measures the availability of water as the accumulated upstream water balance from rainfall, fog, and snowmelt minus actual evapotranspiration. Clean water is then assessed using the Human Footprint on Water Quality index (HWFQ), which measures the extent of downstream contamination from human land-uses, including urban, roads, mining, oil and gas, croplands and pastures. We calculated the amount of clean water available under current land-use using current habitat map compiled of the Jung et al. (2020) layer and plantation data from the Global Forest Watch at a 1km resolution. We then summed the amount in each planning unit to estimate the current clean water that would be safeguarded, assuming planning units in the Recover zone would not be further polluted (Figure S4.11).

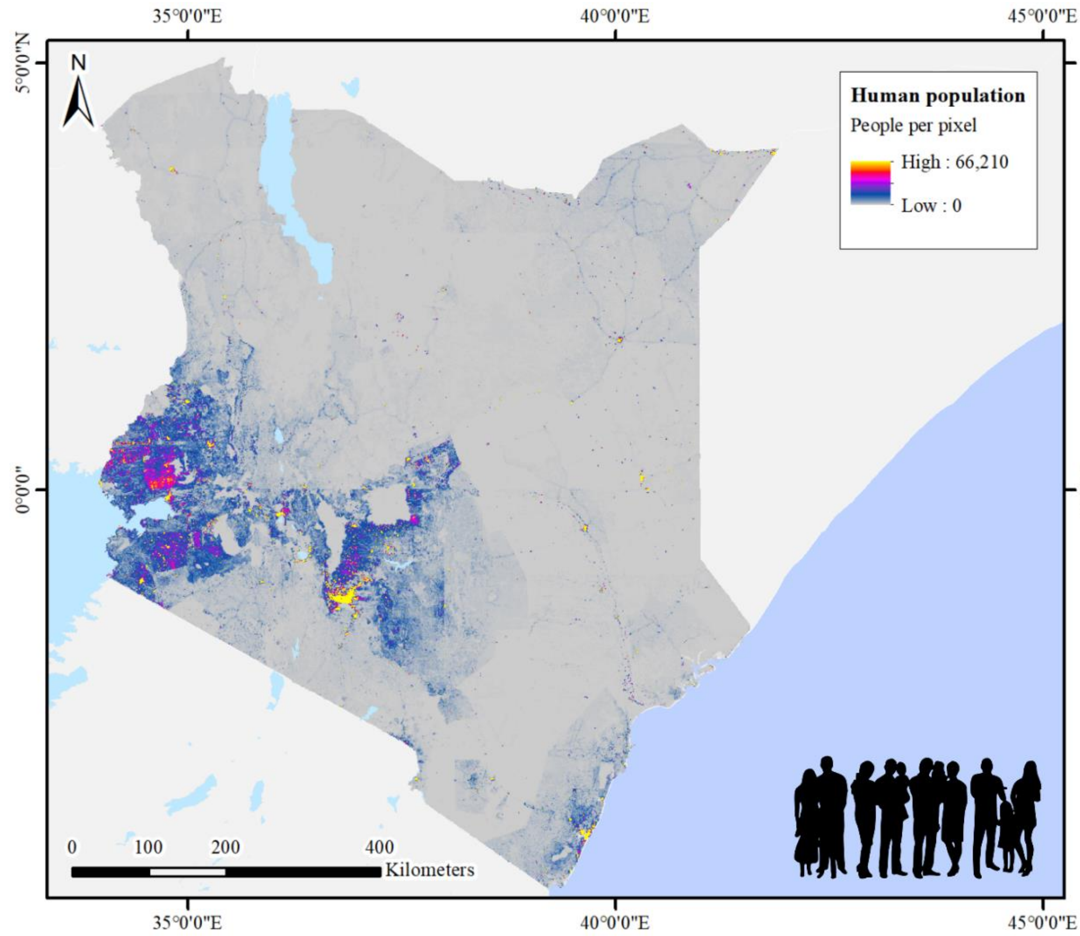

Figure S4.7: Estimated human population in the year 2019 at a pixel resolution of approximately 835m^2, or 30 by 30 arc seconds

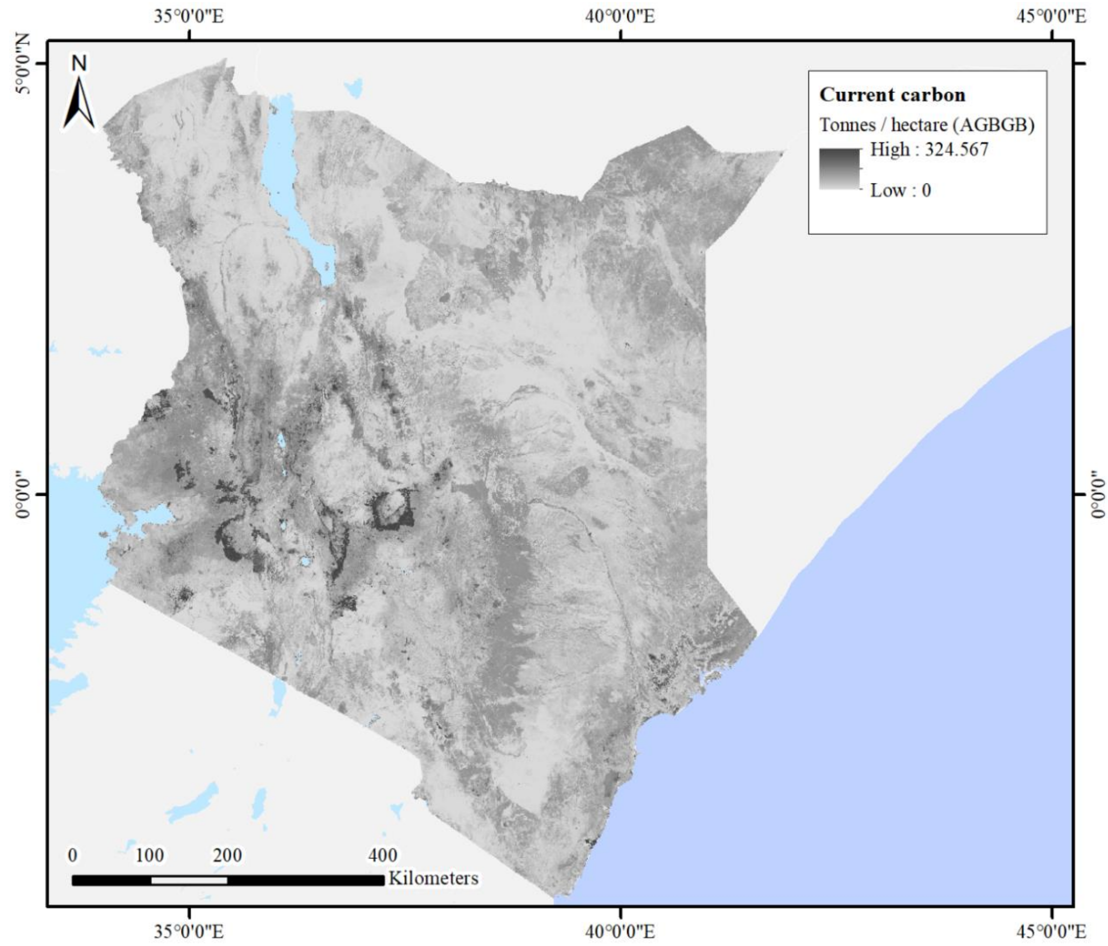

Figure S4.8: Estimated intact carbon stored in aboveground and belowground biomass (AGBGB)

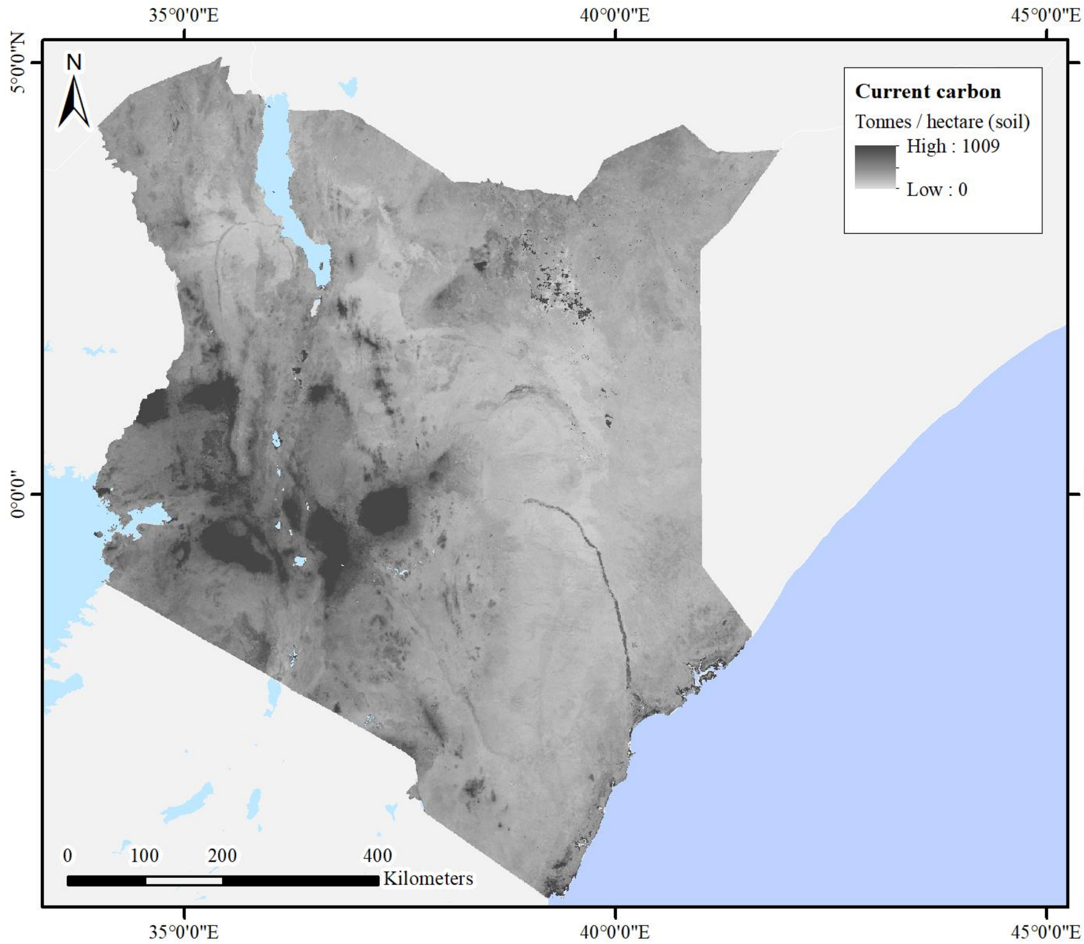

Figure S4.9: Estimated intact carbon stored in organic soil carbon to 1 metre in depth

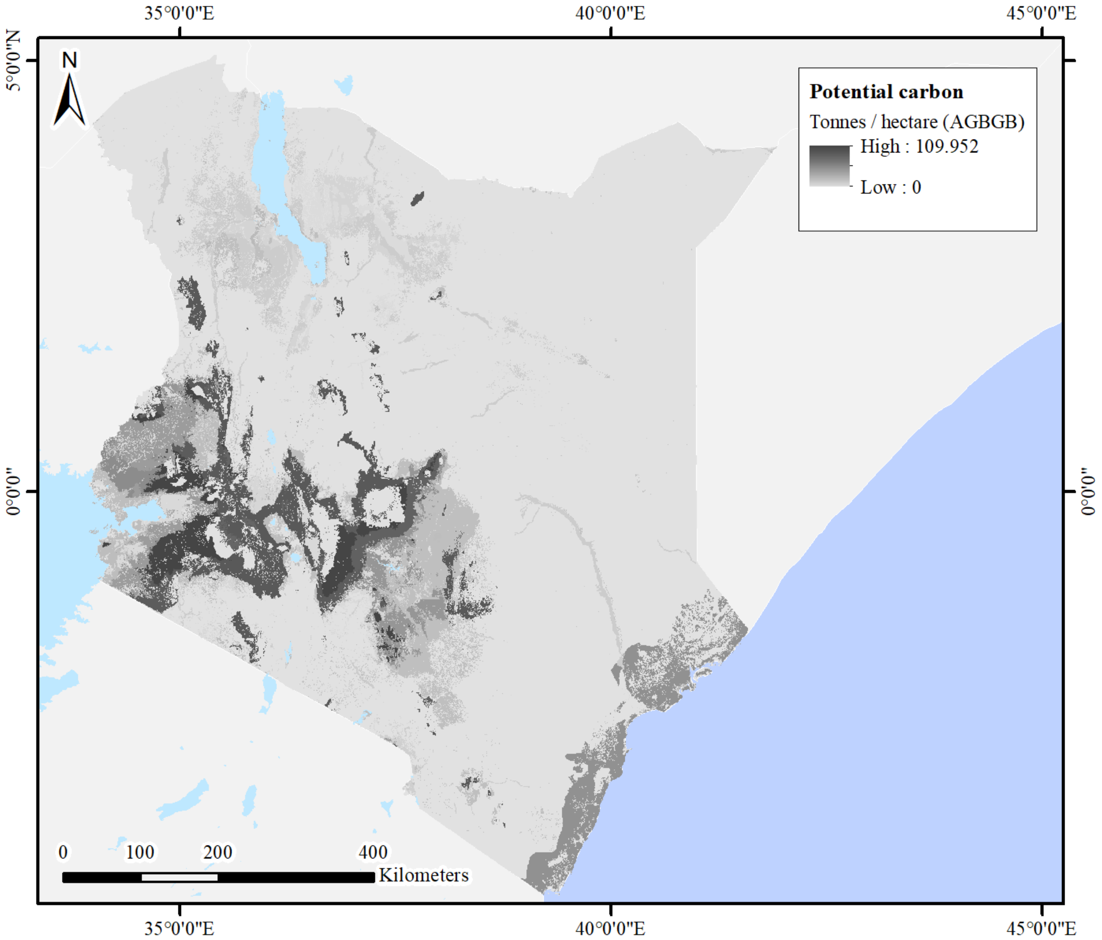

Figure S4.10: Estimated potential carbon capture in aboveground and belowground biomass (AGBGB)

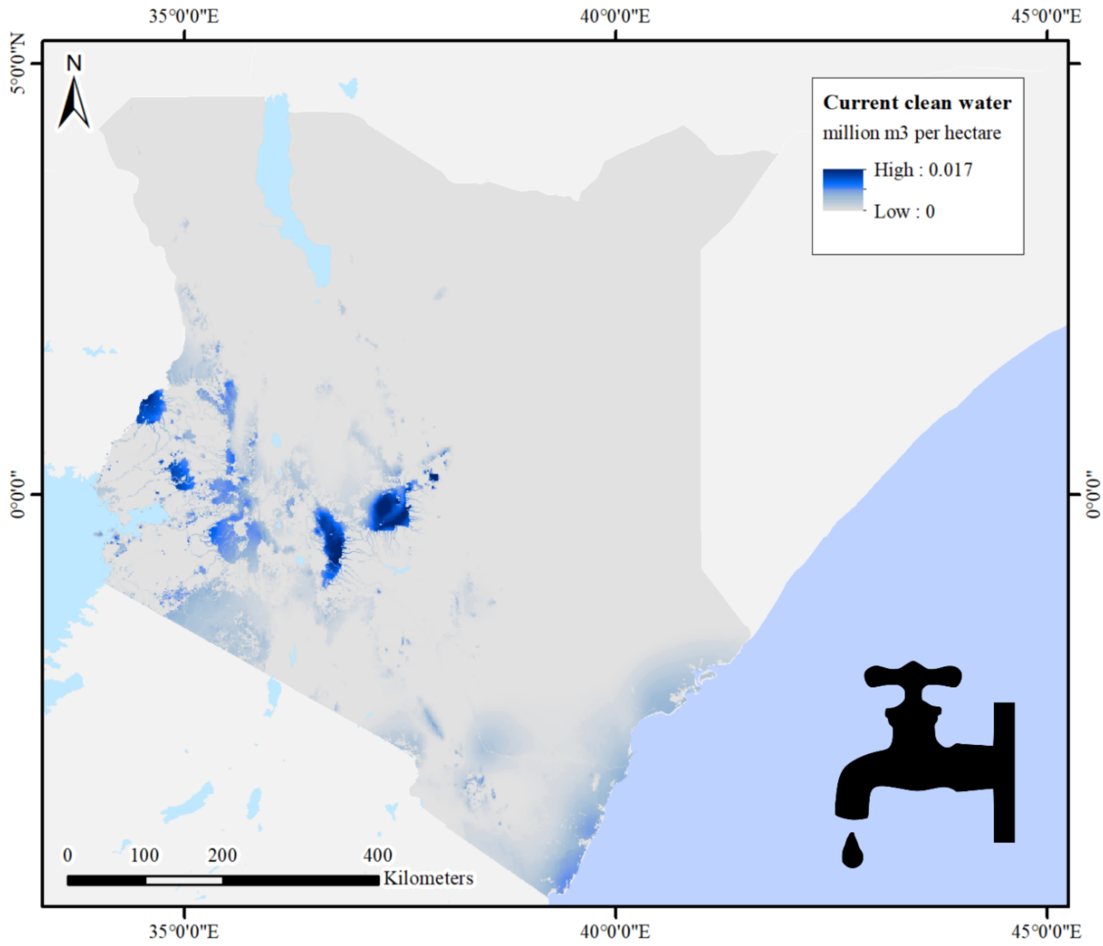

Figure S4.11: Estimated clean water provision

S4.5 Results

Table S4.5: The cost of conservation management and restoration, as well as the land area and the proportion intact habitats in each protected area (PA) type

Protected area type	Conservation management cost (million USD)	Restoration and conservation cost (million USD)	Land area		Intact habitats (%)
			km^2	%	
National Parks (n = 23)	8.07	94.84	28,682	4.98%	98.41%
Other state-run PAs (n = 33)	5.89	521.78	18,284	3.18%	93.12%
Forest Reserves (n = 327)	8.89	2,092.53	16,874	2.93%	71.35%
Sub-total across state-run PAs	22.85	2,709.16	63,840	11.10%	87.63%
Wildlife Conservancies (n = 108)	12.35	1,536.40	46,699	8.12%	97.32%
Sub-total across all PAs	35.2	4,245.56	110,539	19.21%	92.94%
Total across Kenya:	144.32	21,632.72	575,373		83.54%

Chapter 5 Discussion

5.1 Introduction

This thesis explores issues related to spatial planning for conservation and restoration, focusing at the regional and national level for terrestrial Kenya. Signatories to the Convention on Biological Diversity (CBD) have made bold commitments to put nature on a pathway to recovery by 2030 and to be living in harmony with nature by 2050 (Convention on Biological Diversity, 2020). Achieving this will require governments to develop interventions that incentivise and enable local communities and wider civil society with unprecedented urgency, so the work presented here shows how to engage with stakeholders when planning for such changes and investigates the implications of such change. **Chapter 2** shows the benefits of landscape perspectives in identifying priority sites for improving connectivity between and within PAs. **Chapter 3** finds that diverse local communities preferred a landscape where biodiversity is conserved and restored, rather than the continuation of business as usual, or the expansion of intensive consumptive uses. Finally, the spatial prioritisation in **Chapter 4** identifies conservation and restoration networks that would meet the target of 30 x 30 (Waldron et al., 2020). The results show how Kenya’s PAs contribute towards biodiversity conservation and the trade-offs involved when developing ecological networks through conservation and restoration.

Kenya is home to extraordinary levels of biological and cultural diversity, but the issues discussed here relating to implementing conservation and restoration interventions in priority areas will have parallels in many other nations. In this final chapter, I discuss some dominant themes of this thesis: managing landscapes for multifunctionality, the values of integrating ecological restoration with conservation, the need to engage and empower stakeholders in wider decision-making, and how systematic conservation planning fits into this framework. I also suggest options for future research and provide recommendations for policy and practice in Kenya and more widely.

5.2 Contribution to the research field

5.2.1 Landscapes should be managed for multifunctionality

Landscape-scale approaches are important for conservation to integrate with wider land-use plans. Principles for landscape approaches should involve adaptive management practices, focusing on multiple scales, engaging multiple stakeholders, recognising their rights and promoting resilience through multifunctional landscapes (Armitage et al., 2020; Sayer et al., 2013; Tauli-Corpuz et al., 2020). In this thesis, the theme of managing for landscape multifunctionality first emerged early on from the stakeholder interviews that formed the basis of **Chapter 2** and **Chapter 3**. I used these to map a landscape where many consumptive and non-consumptive land-uses occur in varying degrees of co-existence and conflict with biodiversity conservation efforts (Crego et al., 2021; Evans & Adams, 2016; Kamweya et al., 2012a). These outputs created future land-use options as exploratory scenarios, capturing distinct and divergent land-use options, and investigated policy and broader implications for connectivity (McKenzie et al., 2014; Peterson et al., 2003). These go beyond a tradition where connectivity scenarios map only the status quo or specific conservation interventions, showing how broader policy and other social or historical constraints define wildlife movements. In **Chapter 2**, I also provided the first application of expert elicitation through the Delphi Method to undertake connectivity analyses.

5.2.2 Integrating restoration with conservation at the local level

Integrating ecological restoration within area-based conservation also requires a landscape approach (e.g. Strassburg et al., 2019). Throughout this thesis, I contribute to the literature by showing how incorporating restoration within conservation and wider environmental planning can benefit people and nature (Chazdon & Brancalion, 2019; Le et al., 2012; Pritchard & Brockington, 2019). **Chapter 2** confirms findings that decentralised restoration is happening in Kenya (Smucker et al., 2020) and shows how focusing it in small areas produces affordable gains for connectivity and thus biodiversity conservation. I went further than many connectivity analyses that identify the singular most cost-efficient pathway, looking at within linkage dynamics and identifying precisely where further restoration would be most beneficial.

Chapter 3 then uses the future land-use options to map the implications for ecosystem services, which is the first application of a multi criteria decision analysis of this kind in

Kenya. I show that local communities support the restoration and conservation of multifunctional landscapes for a broad range of reasons that reflect their diverse values. This is counter to the general opinion that local people oppose such efforts, but supports broader calls for making community-led approaches the method of choice for targeted interventions in forest landscapes (Erbaugh et al., 2020; Fa et al., 2020; Wiens & Hobbs, 2015). Kenya's history of PA encroachment and eviction (Gathaara et al., 1999; Vanleeuwe, 2004) has fostered resentment from local communities (Kamweya et al., 2012a), and the literature shows that displacement has negative social ramifications and can increase environmental destruction elsewhere (Barr & Sayer, 2012; Latawiec et al., 2015). Therefore, as I discuss in later sections, conservation and restoration will inherently mean different things in different socio-political and biogeographic contexts but must centre on building stewardship from local communities (Armitage et al., 2020).

5.2.3 Spatial prioritisation

National-level planning plays a major role in how countries implement policy and their CBD commitments (e.g. Bicknell et al., 2017). In **Chapter 4**, I use a novel application of a spatial prioritisation to provide an understanding of where is important for integrating conservation and restoration at a national level. The results show the contribution of the existing PAs, emphasising the key role of state PAs for maintaining ecosystem services (Viviroli et al., 2007), but finding that many of these require habitat restoration. We also provide more evidence on the importance of community- and privately-managed PAs for representing biodiversity that is missing from the state-run PAs (O'Bryan et al., 2020; Tyrrell et al., 2019; Western et al., 2020). These existing PAs cover 19% of Kenya, but our analysis identifies that an additional 10% of the country needs to be managed for conservation, and 6% needs to be restored. I then use prioritisation scenarios to show how integrating restoration around intact habitat fragments can produce better conservation networks, but increases management costs and the number of people who would be impacted.

This type of analysis is recognised as the best means to meet the CBD's post-2020 targets (Woodley et al., 2019). I contribute to the literature by carrying out an analysis that recognises the trade-offs between costs and other implications when integrating conservation with restoration (Possingham et al., 2015). Other studies have investigated these when prioritising restoration systematically (e.g. Strassburg et al., 2019), but to the best of my knowledge, this represents the first application of a national-level prioritisation to investigate these effects. More so, when compared to other examples of systematic conservation planning from the scientific literature we take a comprehensive approach to integrating restoration, which is not seen in other regional (e.g. Drummond et al., 2010, Strassburg et al., 2019) or national analyses (e.g. Yoshioka, et al., 2014; Wendt et al., 2016; Bicknell et al., 2017; Tulloch et al., 2021). The benefit of this integration within a national context, also means it is most appropriate to inform the implementation of commitments to the CBD. Interestingly I also identify species requiring restoration that could coexist within multifunctional landscapes (Table S4.3), though obviously, many other species would not be tolerated or tolerate production landscapes in Africa (e.g. Graham et al., 2010; Vanthomme et al., 2018).

Conservation has long focused on frontier mitigation or wilderness protection (Sacre et al., 2019), an approach that continues with contentious ideas such as the ‘half-earth’ proposal (Schleicher et al., 2019; Wilson et al., 2016). My findings highlight that context-specific conservation and restoration interventions are needed that benefit people and the planet (e.g. Pritchard and Brockington, 2019; Chazdon et al., 2020). In implementing them, governments will also have to avoid the dangers of corruption such as elite capture and wider poor governance practices (Smith et al., 2003). Integrating restoration with conservation in multifunctional landscapes can move away from a paradigm of preservation and encouraging ecosystem dynamism and diversity. This can put biodiversity and people on a pathway to recovery.

5.3 Future research

Throughout the course of undertaking this thesis, I identified important knowledge gaps where further research could be focused to provide interesting and important insights for conservation and restoration in Kenya and more broadly.

5.3.1 Elephants as a surrogate for connectivity

Maintaining connectivity using elephants as a surrogate for broader connectivity is an important part of Kenyan policy (Government of Kenya, 2017) and research (e.g. Ihwagi et al., 2019; Bastille-Rousseau and Wittemyer, 2020; Crego et al., 2021), as well as being the case across the wider continent (Purdon et al., 2018; Wall et al., 2021). I show that elephants use one restored linkage within Kenya’s Central Highlands, but further research should investigate how many individuals use this and other linkages, providing insights on how important they are for metapopulation maintenance. Further research could also investigate where fence or other barrier removal would improve connectivity (e.g. Osipova *et al.* 2018; Crego *et al.* 2021).

Another interesting question will be to investigate whether natural resource availability in Kenya's mountain forests is leading to the autoregulation of their elephant populations. Elsewhere in Africa's savannah systems, elephant populations follow a boom-and-bust pattern, which drives habitat change and human-elephant conflicts (Di Minin et al., 2021; Purdon et al., 2018). This does not seem to be the case in Kenya's mountain parks, where the elephants of the Aberdare Range have been entirely isolated for decades (Vanleeuwe & Gitau, 2020; Vanleeuwe & Lambrechts, 2017).

Elephants regularly use suboptimal habitats for crop-raiding and passage (e.g. Graham et al., 2009; Kamweya et al., 2012), which provides two interesting research areas relating to their management. The first is research into what aspects of restored habitats best encourage movement – i.e. whether the multifunctionality of the existing corridor (Green et al., 2018) inhibits movement and more habitat refuges are needed throughout the corridor. The second is the degree to which projects to strengthen linkages for elephants facilitate the movement and dispersal of other species (e.g. Jackson and Sax, 2010; Damschen et al., 2019).

This further research is needed as restoration of connectivity needs to be based on the socio-political and ecological context (Keeley et al., 2018). Poorly designed or inappropriately located linkage endeavours may fail or, worse, can increase edge effects (Haddad et al., 2014), help the spread of invasive or otherwise antagonistic species (Daskin & Pringle, 2018), exacerbate human-wildlife conflicts (Buchholtz et al., 2020) or have negative impacts on environmental and social justice (Goldman, 2009).

5.3.2 Engaging stakeholders

Using different forms of expertise is important (Wheeler & Root-Bernstein, 2020, Pascual et al., 2021), and here I used diverse knowledge, from people who grew up dealing with crop-raiding elephants to western researchers who collected movement data with GPS collars. Throughout my thesis, I highlight the value of local understanding and the need for further stakeholder engagement. When investigating stakeholder values, we found broad support for habitat conservation and restoration. However, more research is needed to look at the context of spatial scale in these decisions, considering what happens to people's value judgements when conservation or restoration happens on their land, to their neighbours, or further afield.

5.3.3 Spatial prioritisation

Targets best perform when quantifiable, realistic, clear, and scalable (Green et al., 2019). Target-based approach outputs are, after all, dependent on the values that underpin them (Levin et al., 2015; Svancara et al., 2005). We produced robust prioritisations using detailed habitat data and selecting the habitat preference of Near Threatened and Threatened species (Brooks et al., 2019; Visconti et al., 2019), which have been shown to provide cost-effective coverage to other species (Drummond et al., 2010). However, those developing such plans should consider the implications of choosing different prioritisation feature data, such as endemic species or those identified by national red lists, something that does not currently exist for Kenya. The inclusion of additional taxa such as plants and invertebrates will also produce different outcomes. More so, future spatial planning should incorporate data on bio-abundance (Williams et al. 2020; Mair et al. 2021) and genetic diversity (Butchart et al., 2015). Analyses like ours are also prone to errors of commission and omission (Di Marco et al., 2017; Rondinini et al., 2006). While we reduced this by accounting for habitat and elevation associations when calculating the area of habitat (Brooks et al., 2019), validating and correcting these distribution models would improve the spatial prioritisation results.

More broadly, in spatial prioritisations, further research needs to be done to incorporate the different threats within restoration and conservation priority areas. This includes aspects such as climate change (Groves et al., 2012; Jones et al., 2016) and habitat transformation risk (Sacre et al., 2019). Future prioritisations are also needed that better incorporate agricultural opportunity costs and account for equity, identifying areas where restoration may be better achieved through agroforestry and silvopasture.

5.4 Recommendations for policy and practice

5.4.1 Invest in protected areas

My comprehensive mapping of Kenya's PA estate showed that many PAs are highly degraded (Table S4.5). Efforts should be made to avoid the existence of paper parks, which provide perverse outcomes such as underachievement due to misallocated protection (Barnes et al., 2018; Rodrigues et al., 2004) or overstatement due to the existence of paper parks, which can be no more effective than unprotected land (Bertzky et al., 2012; Hockings et al., 2004). From my personal experience of Kenya, these are always of lower priority and protected status. Two key protected area types to look at are the 'Other state-run PAs' (mainly National Reserves) and Forest Reserves. Both have low levels of active management, mostly being managed from local towns and receiving little or no funding from the state. Four that are likely paper parks are Losai National Reserve, the Diani-Chale Marine National Reserve, Ngai Ndaathia National Reserve and the Ngurumun Forest Reserve.

Africa's PAs have long needed better funding (Bruner et al., 2004; Moore et al., 2004), with the Covid-19 pandemic again highlighting the precarity of PA funding (Lindsey et al., 2020). These authors suggested 'green grants' for Africa. In Kenya this could work, with payments for ecosystem services being found to be most appropriate due to land-tenure and practice (Curran et al., 2016). To remedy the issues of poor performance due to underfunding, where appropriate, governments could look at funding and co-management partnerships (Baghai et al., 2018) and wider investments from high-income countries (Balmford & Whitten, 2003). Though these may be contentious, well-functioning PAs can benefit local communities and economies (Naughton-Treves et al., 2005). However, conservation in Africa is massively diverse in its practice and in places, very different strategies exist to the comparatively mainstream "if it pays, it stays" model of eastern and southern Africa (Bauer et al., 2021). The vulnerability of which was shown during the Covid-19 pandemic (Lindsey et al., 2020), although tourism does benefit local livelihoods around PAs (Naidoo et al., 2019).

5.4.2 Promote protected area and wider resilience to climate change

Improving PA management effectiveness will be particularly important under climate change, with temperatures on protected land set to increase by as much as 3 degrees Celsius by 2050 (Maxwell et al., 2020). Countries must urgently manage their PA networks to promote resilience to climate change, ensuring governance and management structures implement realistic policies for adaptation (e.g. van Kerkhoff et al., 2019; Elsen et al., 2020). This will be particularly urgent as OECMs are brought into conservation frameworks, as these sites may be less engaged with wider scientific and practitioner communities.

5.4.3 Use restoration to connect protected areas

Ecosystem restoration should be a means to promote this resilience. Africa's PAs are currently poorly connected (Saura et al., 2019; Ward et al., 2020), so restoring and reconnecting them would aid resilience and species dispersal (e.g. Jackson and Sax, 2010). However, this needs to be managed carefully in a country such as Kenya, which ranks near the top of all African countries for human-wildlife conflicts (Di Minin et al., 2021). Fencing has become commonplace in Kenya (Dupuis-Désormeaux et al., 2016; Evans & Adams, 2016; Osipova et al., 2018). This is a sad reality, but it will be an essential part of managing multifunctional landscapes in a continent with some of the largest remaining megafaunal populations (Ripple et al., 2016). Policy incentives that promote the restoration of multifunctional landscapes (e.g. agroforestry and silvopasture) will benefit biodiversity, ecosystems and people, and can change the trend of environmental degradation. Kenya should therefore look at how their priority species could be maintained in production landscapes and build this into their planning processes (e.g. Wilson et al., 2010).

5.4.4 Engage stakeholders in further systematic conservation planning

The key challenge for implementing commitments made to the Convention on Biologically Diversity will be formalising the goals it outlines through targeted interventions that incentivise local communities in sound environmental practices. Well-designed spatial prioritisations can help governments identify where to focus specific policies and practices. Kenya can improve on our analyses, which used standard methods from the literature for setting targets (Rodrigues et al., 2004; SANBI, 2018), rather than using a stakeholder-driven approach. Now that we have developed a national conservation planning system, the next step is to work with Kenyan policy-makers, conservation practitioners and the wider environmental management and academic communities to produce outputs that are specifically designed to inform action on the ground. This should also benefit from using the wealth of available data from wildlife censuses and citizen science efforts to set species representation targets based on population viability assessments.

5.4.5 Use restoration to mainstream biodiversity

Countries such as Kenya need to think carefully about how to implement restoration. Promoting multifunctionality will be a key part of this, rather than monoculture agriculture or blanket restoration strategies that can both have negative socio-economic consequences and lead to habitat and biodiversity loss elsewhere (Barr & Sayer, 2012; Grass et al., 2020; Latawiec et al., 2015). In particular, there is a need to incentivise industries and the wider economy in these sound environmental practices to mainstream biodiversity. Frameworks such as the Mitigation and Conservation Hierarchy hold a lot of promise (Arlidge et al., 2018;

Milner-Gulland et al., 2021) but must account for the impacts on key elements of biodiversity, framing goals that are ambitious and offer a true net gain over a clear timeframe and space (Maron et al., 2021).

5.4.6 Monitoring for effectiveness and equity

In designing their conservation networks and best integrating restoration, countries should continue to monitor PA effectiveness. This is important to ensure they function to the best of their abilities (Coad et al., 2019; Geldmann et al., 2019). However, efforts to monitor equity are also important (Zafra-Calvo et al., 2019; Zafra-Calvo & Geldmann, 2020). The scale of implementing commitments to the CBD means that these efforts should be further expanded and adapted to include OECMs and restoration efforts (Erbaugh & Oldekop, 2018; Monaco et al., 2012; Watson et al., 2017). This will require different strategies for monitoring the fundamental differences between rangeland restoration (e.g. Kimiti et al., 2020) and reforestation (e.g. Viani et al., 2017).

Implementing conservation and restoration in Kenya, as well as in other countries, will be about national and county level spatial planning that empowers and enables those on the ground in sound environmental stewardship but also provides robust and accountable monitoring structures. Solutions based on developing multifunctional landscapes are more likely to be supported by local communities. This will come at a cost to some species, but if implemented appropriately to promote human wildlife coexistence, it will be best placed to promote good environmental stewardship. Doing so requires equitable decision making and governance (Armitage et al., 2020). This method is a challenge to Kenya’s status quo. However, devolved county government is supposed to be a means to remedy injustices where rural peoples, particularly those of the arid and semi-arid lands, have been marginalised politically and economically (Smucker et al., 2020). Kenya, therefore, needs big picture national plans that feed into county plans and then also incentivise landowners, but as with all governance issues, this will be prone to inequities such as elite capture, which is why it needs proper oversight from informed and accountable government agencies.

5.5 Conclusions

This thesis came together as we move into a decade where governments, industries and wider civil society have acknowledged the scale of the climate and wider ecological crises and are now agreeing on how to act (Bhola et al., 2021; Convention on Biological Diversity, 2020; Milner-Gulland et al., 2021). I found a diverse local community to support biodiversity conservation and ecosystem restoration actions. I then identified priority sites for biodiversity conservation and ecosystem restoration, showing the much higher cost of restoration, but discussing the trade-offs between this and the gains for biodiversity and ecosystem service conservation, as well as people. These findings provide important lessons that are widely applicable and show that we need to be explicit about our goals and explicit about benefiting people when targeting interventions in priority landscapes. If local people and economies are put at the centre of conservation and restoration efforts, this will produce an optimistic, effective, and equitable way to turn the tide on environmental destruction and degradation.

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