Conservation in a crowded place

Forest and people on Mount Elgon, Uganda

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Abstract

A growing world population has important consequences for forests. In this study I investigate how conflicting goals by different actors under different historical contexts impacted the protected area of Mt Elgon, Uganda, and I consider what this means for conservation. Mt Elgon is an important water catchment area for Uganda and Kenya with important biodiversity values. The forest on Mt Elgon is also a source of agricultural land, timber, fuel wood and other forest resources for local communities. In this study I explore the factors that influenced local people's motivations for forest clearing, the impacts of local forest use, including as a source of fuelwood, on Mt Elgon, Uganda. I also evaluate the use of radar satellite data to estimate above ground biomass on Mt Elgon. Finally I discuss the implications for the design of interventions that seek to reconcile the needs of local people and forest conservation.

A major wave of deforestation on Mt Elgon, Uganda took place in the 1970s and 1980s and by 2009, 25% of the forest on Mt Elgon was lost. However, locally, there were areas of recovery. This study demonstrated that agricultural expansion on Mt Elgon cannot simply be linked to individual drivers such as population or high crop prices, and these were not always associated with increased deforestation. By analysing local variations, I found that it is the *context* (institutional, social, political) under which drivers such as population, wealth or commodity prices operate, rather than the drivers *per se*, that influences outcomes for forest cover.

I found that local forest uses strongly influenced forest structure, even where people had a collaborative management agreement with the park authorities. The type of resources collected varied with the land use systems around the park: small stem-harvesting affected regeneration in areas where people grew crops that require supports such as bananas and climbing beans, and seedlings were almost absent where in-forest cattle grazing was important. Studying the characteristics and impacts of fuelwood harvesting revealed high levels of fuelwood collection and depletion of dead wood on the edge of the park. Human impacts affected highly preferred and used tree species. Allowing the collection of fuelwood or other non-timber products creates opportunities for more destructive activities such as timber harvesting or charcoal making. On the other hand it helps to improve relations between local people and park staff, which this study showed helps limit agricultural encroachment. I also found indications that trees on people’s own land can provide alternative sources of fuel.

Mt Elgon has a history of conservation and development projects in an attempt to better reconcile local livelihood improvement and forest conservation. The most recent include pilot REDD+ schemes both inside and outside the protected area. Such schemes need consistent biomass estimations. I used a cost-effective field method for direct basal area estimation that yielded consistent estimates of above ground biomass (AGB), which reached above 800Mg/ ha on Mt Elgon’s northern slopes. Radar (ALOS PALSAR) data produced realistic classifications of the different vegetation types. However, using radar backscatter values in combination with field estimated AGB data to produce a biomass map had limited success. This was likely linked to the sampling strategy and topography.

Our study showed that simple theoretical models based on single drivers of deforestation cannot explain local variation, nor can simple models that lead to “simplified institutional prescriptions” lead to sustainable solutions, as they do not reflect complex local social and ecological realities. This has important implications for the design of more locally adapted and ecologically and socially sustainable management arrangements on Mt Elgon and elsewhere. These are necessary because current practices appear to lead to forest degradation and resource depletion. Building trust between stakeholders and developing alternative resources are vital to support more sustainable forest management. Both international conservation actors, as well as forest management authorities need to recognise that incentives that influence people’s motivation for action vary locally and can therefore not be designed globally.

Key words: tropical forest, conservation management, local livelihoods, forest cover change, disturbance, fuelwood, forest structure, species richness, biomass, Mount Elgon

Contents

Chapter 1

General introduction

1.1. Background

1.1.1. Drivers of forest change

At the global scale, the need to provide food and energy for a growing world population is leading to changes in land use with important consequences for remaining forest areas (Lambin and Meyfroidt 2011). At regional or national scales, poor and land-hungry farmers are considered the main threat to forest conservation. Population and poverty seemingly increase dependence on and the need for forest land and resources (Uusivuori et al. 2002, Lung and Schaab 2010), although in frontier areas with fewer people, farmers also expand into forests because they lack the resources to intensify (Angelsen and Kaimowitz 2001). However, it is generally not just population pressure and poverty that drive forest loss, but rather the interaction at the global to local level of various political and socio-economic contexts and forces (Angelsen and Kaimowitz 1999, Lambin et al. 2001, Geist and Lambin 2002, Carr et al. 2005).

Important questions are debated on global agendas, such as: Will intensifying agricultural production lead to decreased deforestation (the Borlaug hypothesis) or will the opposite occur and at which scales? What is the influence of global commodity markets on land use? and, What are the trade-offs between increased human welfare and ecosystem services[1]?. Most studies focus on forest loss, whereas in some places forest is expanding or regenerating (Fairhead and Leach 1996, Rudel 1998, Rudel et al. 2005). Secondary or degraded forests are important for conservation because of their extent and their proximity to remaining old growth forests (Wright 2005, Wright and Muller-Landau 2006, Chazdon et al. 2009). Factors that lead to regeneration are just as important as those leading to deforestation and can help understand how to prevent further forest loss. Also, most focus so far has been on loss of forest cover, whereas forest degradation is often more extensive, especially in protected areas. But forest disturbance is harder to measure than outright deforestation. The role of protected forests and the impacts of these forces on conserving the products and services they provide locally need to be clarified. Low cost methods are needed that can help improve monitoring of impacts in places where financial resources for conservation are limited.

1.1.2. Conservation and local people: changing paradigms

The dominant paradigm concerning most protected areas in the world still emphasizes the importance of 'untouched nature' (Philips 2003, Borgerhoff Mulder and Coppolillo 2005).

[1] These include global environmental services provided by forests such as water catchment values etc and local products and services to forest dependent people.

Management is geared towards the preservation of wildlife and the attraction of tourists, at the exclusion of local people who often depend on these areas for their livelihoods (WRI 2005). Not unexpectedly this can lead to conflict (Hough 1988, Balmford et al. 2001). In Africa, the 'fortress conservation' model (Brockington 2002) fitted well with the administration style of the colonial powers and subsequent post-colonial governments, concentrating the control of natural resources in the hands of the central administration (Philips 2003). Colonial powers created reserves and parks in vast areas of forest and savanna considered to be untouched pristine lands, without consideration of historical local land uses (Adams and McShane 1996, Chatty and Colchester 2002). Historically many conservationists have seen people and the local use of natural resources as incompatible with conservation (e.g. Oates 1999, Terborgh 1999, Locke and Dearden 2005).

Local people are often those most affected by forest degradation or conversion and restrictions on forest use impact their livelihoods, but their views and priorities frequently remain unheard by decision makers (Sharpe 1998, Lawrence et al. 2000, Sheil et al. 2006). However, over the past decades, thinking on protected areas has changed towards a more socially equitable and integrated paradigm where protected areas also have social and economic objectives and are managed by a wider range of actors, including local communities (Philips 2003). Protected areas are seen as part of a wider socio-ecological system, with various land uses and functions, subject to various external pressures and influences (Sayer and Campbell 2004).

1.1.3. New approaches to forest management and conservation

Approaches that integrate conservation and development goals, support sustainable use and devolved forest management have emerged as alternatives or as complementary to strict conservation (Wells and McShane 2004, Lele et al. 2010). The underlying assumption being that devolving forest management to local communities will provide incentives for more sustainable forest use (Agrawal et al. 2008). Several global meta-analyses of published case-studies show evidence that forests managed by local communities are equally or more effective at maintaining forest cover than those under stricter protected area regimes (Persha et al. 2011, Porter-Bolland et al. 2011). However, conservation and development and community forest management projects often have pre-determined conservation goals – e.g. boundaries are not usually negotiable (Sharpe 1998), and insufficient powers are often transferred to local institutions (Ribot 2002). Decentralization of forest management may also lead to resource capture by local elites if effective supporting institutions are not in place (Larson and Soto 2008, Persha et al. 2011).

There has been extensive research on the effectiveness of protected areas (e.g. Bruner et al. 2001, Struhsaker et al. 2005, Hayes 2006, Laurance et al. 2012), and a number of assessments have been conducted on the outcomes of conservation projects on conservation and neighbouring communities (e.g. Plumptre et al. 2004, Fisher et al. 2005, West et al. 2006, Sayer et al. 2007). But in practice win-win solutions are rare (Chan et al. 2007, McShane et al.) and more evidence is needed on the conditions that lead to successes or failures for both people and conservation.

1.1.4. Finding a balance

The consensus concerning the need to integrate local people into conservation management emerges alongside the increasing pressures on natural resources that arise from the rapid increase in population. A balance needs to be found. To maintain the resources that local people use as well as those that conservationists and other actors value, ways need to be found in which multiple and sometimes conflicting uses or values of natural resources can be combined or negotiated (Kaimowitz and Sheil 2007, McShane et al. 2011, Sayer et al. 2013). Even when there is no outright clearing, most tropical forests, even those in protected areas, are influenced by human activity (Olupot et al. 2009, MacKenzie et al. 2012). But there is a lack of understanding on the drivers for forest clearing and regrowth under different political and socio-economic contexts.

Overall, conservation related interventions, including the establishment of protected areas, the development of non-timber forest products, tourism and other alternative sources of income to local communities, as well as collaborative management arrangements, have failed to take the complexity and dynamics of underlying contexts and drivers of forest change sufficiently into account as well as the importance of these in influencing local people's decisions (Putz and Romero 2012).

Such understanding is vital in the context of payments for environmental services (PES) schemes such as the planned implementation of the Reduced Emissions from Deforestation and Forest Degradation in Developing Countries (REDD+[2]) policy mechanism under the United Nations Framework Convention on Climate Change (UNFCCC). Payments made under such schemes could, in principle, provide additional incentives for forest conservation and the restoration of degraded forests. But technical and institutional challenges need to be addressed: how to set reference emission levels in degraded forest

[2] "REDD+ goes beyond deforestation and forest degradation, and includes the role of conservation, sustainable management of forests and enhancement of forest carbon stocks" (UN-REDD Programme http://www.un-redd.org/AboutREDD/tabid/102614/Default.aspx)

used by people and monitor change? And how to resolve questions of rights over land and resources and the dilemmas in common pool resource management? (Chhatre and Agrawal 2009, Reynolds 2012). Such dilemmas are no different in REDD+ projects than in other projects based-on collaborative management of forest resources (Dietz et al. 2003, Reynolds 2012). In particular when forests are owned by the state, such as in the case of most protected areas, local communities who are involved in collaborative management schemes may feel little ownership over the resource, leading to overuse and degradation (Chhatre and Agrawal 2009). I further discuss this in light of my results in section 6.3.

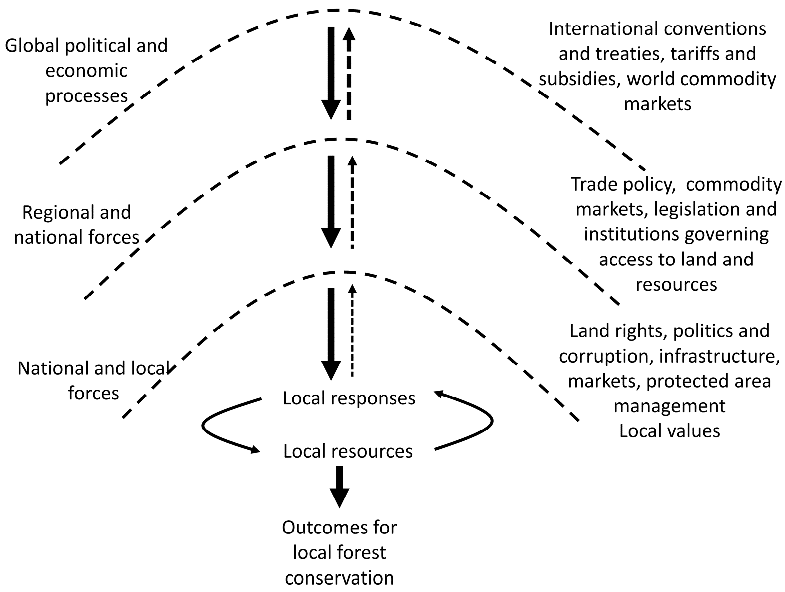


Figure 1.1. Interactions between processes, policy and drivers of resource use, local responses and outcomes for forest conservation (modified from Giller et al. 2008).

1.1.5. Understanding interactions and contexts

Perceived benefits, weighed against costs incurred through access or institutional barriers, largely determine people's use of and impact on that resource (Schweik 2000, Lynam et al. 2004, Norgrove and Hulme 2006). The outcomes of these interactions are reflected in the state of the environment and in the products and services it provides and may vary across scales (VanWey et al. 2005, Hersperger et al. 2010). More local studies are needed to understand how human and biophysical factors interact in time and space, for which outcomes: *i.e.* what the impacts are of competing demands on forest land and resources. Contexts under which such interactions take place include global and national level political and economic processes, institutional frameworks governing access to resources, the values of stakeholders and the ecology of the resources (Figure 1.1). Tools such as remote sensing in combination with field-based research can help to improve the understanding of interactions and ultimately inform management decisions (Nagendra et al. 2003, Ostrom and Nagendra 2006, Southworth et al. 2006).

Mount Elgon (henceforth Mt Elgon), on the border of Kenya and Uganda provides a relevant case to study these interactions. Mt Elgon has a history of protection under various more or less exclusionary management regimes, numerous boundary demarcation exercises and growing pressure exercised by high population densities (up to 1000p.km^{-2}). The protected area has known forest loss due to widespread encroachment but in some places forest has also recovered over time (Otte 1991, van Heist 1994, Norgrove and Hulme 2006, Soini 2007). Information on forest degradation due to local uses or past logging is limited and largely anecdotal. On the Kenyan side a study of the impacts of logging on forest structure and species was conducted in the early 2000s (Hitimana et al. 2004). The whole area – including the Kenyan side - has experienced historical conflicts over resource access and use (Norgrove and Hulme 2006, MERECP 2007). Multiple conservation and development related projects have been tested since the 1990s and REDD+ related schemes that include both local livelihoods and conservation objectives have been piloted since 2010 (UWA 2000, LVBC 2009).

1.2. Mt Elgon

1.2.1. Biophysical characteristics

Mt Elgon is a solitary extinct volcano straddling the Uganda-Kenya border (4321m). It is located between 0^{o}52’ and 1^{o}25’N, and between 34^{o}14’ and 34^{o}44’E. Mt Elgon is the oldest of the large East African volcanoes, probably of Tertiary origin (Davies 1952). The last major eruption probably took place about 12 million years ago, with smaller ones up to 2 million years ago. Hot springs can still be found in the caldera. Mt Elgon is a shield volcano, with an overall gentle slope of around 4°. However, the lower slopes in the west and north form a characteristic stepped topography with spectacular cliffs sometimes more than 300m high. A 20 km long ridge over 2000 m high reaches out to the west (Figure 2.1 in Chapter 2) (Dale 1940). Mt Elgon’s caldera is around 8 km across, making it one of the largest calderas in the world (Davies 1952). Average annual rainfall is between 1200 and 2000 mm, varying with elevation and side of the mountain (Dale 1940, Soini 2007). Numerous streams come off the mountain, originating in the caldera and increasing in volume on their way downslope, feeding into the Turkwell and Lake Turkana system, the Lake Victoria Basin, Lake Kyoga and the Nile Basin (IUCN 2005).

The vegetation of Mt Elgon can be stratified into broadly three zones, although they are not strictly elevational zones: an afroalpine and ericaceous zone, an afromontane forest zone (below about 3200 m) and an afromontane rainforest zone (Dale 1940, Hedberg 1951, Langdale-Brown et al. 1964, van Heist 1994). The afroalpine and ericaceous zone - also called moorland and heath zone - is composed of bogs, shrub- and grasslands that are rich in (endemic) shrubs and herbs (including the endemic *Lobelia elgonensis*, *Alchemilla elgonensis* and *Senecio elgonensis*) (Dale 1940). *Philippia* spp. thickets and woodlands form the transition to afromontane forest communities. Frequent fires have affected vegetation structure and composition in these areas (Hamilton and Perrott 1981, Beck et al. 1987, Wesche et al. 2000). The afromontane forest zone is characterised by higher elevation forest types and bamboo forest (*Arundinaria alpina*), but patches of dominant species occur next to each other (e.g. *Hagenia abyssinica*, *Cornus volkensii* or *Podocarpus milianjianus*) (Dale 1940, Hamilton and Perrott 1981). Of the lower elevation afromontane rainforest (e.g. *Aningeria adolfi-friedericii* and *Strombosia Schefflerii*.) only small and patchy areas remain on the edges of the protected area in the south and on the western ridge (van Heist 1994).

1.2.1. Conservation values

About 1464 vascular plant species have been identified for Mt Elgon, of which 43 are regional endemics and 39 only recorded from Mt Elgon (MUIENR and NMK 2005). Mt Elgon hosts 120 species of mammals, representing 12 out of 13 orders of mammals recorded for Africa. IUCN has listed 37 faunal species on Mt Elgon as "globally threatened" of which nine are endemic (IUCN 2005). Mt Elgon is considered one of the richer forests in terms of bird species, including some with a limited range (Davenport et al. 1996). Previous studies found indications that forest specialist birds are being negatively affected by human disturbance (Katende and et al. 1990, Project Elgon 1997a).

Mt Elgon derives most of its biodiversity importance from species that are rare or have limited distributions in Uganda and in East Africa (Table 1.1), notably in the higher elevation moorlands (Howard 1991, Davenport et al. 1996, MUIENR and NMK 2005). Mt Elgon was ranked among the top 20 most important forests in Uganda in terms of overall biodiversity importance, and number 6 in terms of species rarity value (Howard et al. 2000). Mt Elgon is also part of the East African Mountains and contributes to preserving the biodiversity characteristic of these ecosystems. A number of studies have described Mt Elgon's vegetation in general, listing species or studying altitudinal zonation (e.g. Dale 1940, Langdale-Brown et al. 1964, Tweedie 1976, Hamilton and Perrott 1981), but there seem to be a lack of information on species distributions (a study, including specimen collection, was conducted in 1993 but the data was never fully published).

Table 1.1. Biodiversity importance of five indicator taxa surveyed in Mt Elgon. Adapted from Davenport et al. (1996).

	Trees and shrubs	**Birds**	**Small mammals**	**Butterflies**	**Large moths**
No. of known forest species for Mt Elgon	273	296	30	171	71
No. of restricted-range species (known from ≤ 5 Ugandan forests)	50	40	6	17	10
No of regional endemics	-	0	1	1	1
No of species recorded by Davenport et al. 1996)	254	163	20	115	67
Species diversity	**	**	**	**	*
Species conservation value	****	***	***	***	****

Star ratings indicate values relative to the other 64 Ugandan forests investigated by Davenport et al. (1996):
**** top 10% of sites; *** top 11-25% of sites; ** mid-ranking 26-74% of sites; * bottom 25% of sites.
Regional endemics refer to species restricted to Uganda, the Albertine Rift and/or the Somali-Masaai region.

Mt Elgon is an important water catchment area for several million people in the surrounding districts in Uganda and Kenya (Dale 1940, UWA 2000, Akotsi and Gachanja 2004). The high intensity land use around the mountain and significant impacts of local uses make the forest on Mt Elgon vulnerable to degradation, especially in terms of water catchment. Erosion due to deforestation upstream may have important impacts further downstream although this would need to be further assessed. Due to high rainfall and typical soil properties and stratification, many slopes on Mt Elgon are naturally unstable, even on relatively gentle slopes. However, deforestation and slope excavation e.g. for house-building were found to be major preparatory factors for landslides on Mt Elgon. Small landslides have been reported for over a century but major landslides with numerous casualties occurred in 1933, 1964, 1970, 1997, 2010[3] and 2012[4] (Knapen et al. 2006, Mugagga 2011). In recent years Mt Elgon has developed a 40-kilometre crack with a width of between 30 to 35 cm, leading to fears of further casualties[5]. The Ugandan National Environment Regulations for Mountainous and Hilly Areas Management established strict regulations, that for example prohibit cultivation of slopes steeper than

[3] http://www.bbc.co.uk/news/world-africa-18595913
[4] http://www.reuters.com/article/2012/06/25/us-uganda-landlside-idUSBRE85O0MZ20120625
[5] http://www.ipsnews.net/2012/07/overpopulation-on-ugandas-mount-elgon-kills-hundreds/

15% but such rules are unrealistic is such densely populated areas (Kajura, 2001 in Knapen et al. 2006).

Mt Elgon also has important cultural values for the people living on its slopes. Values vary a little according to the ethnic group. The Bagisu and Sabiny have different histories in relation to the mountain that are reflected in the values they attribute to it. For the Bagisu Mt Elgon has significance in relation to the origin of mankind, and in relation to traditional circumcision ceremonies (UWA 2000 and local informants). The Sabiny recognise special sites for rituals and ceremonies. Both ethnic groups reportedly have ancient burial sites on the mountain, inside what is now the national park. Ancient rock paintings have been found in various caves (Wright 1961 cited in Weatherby 1965).

1.2.3. Land use and forest management

The forests and higher elevation areas of Mt Elgon on the Ugandan side of the mountain are protected by a national park. In Kenya, there is a national park, a forest reserve and a national reserve. Nevertheless, these areas are an important source of natural products for a large proportion of the people living in their vicinity. They also have cultural and religious significance as they host traditional sites (e.g. for circumcision) and ancestors' graves (Scott 1994a, Scott 1998). Monetary benefits from tourism are limited and dependence on the forest for subsistence is likely to remain in the long term (Scott 1998). Poverty is widespread (40-70% of people are below the poverty line in Kenya, 30-40% in Uganda (Soini 2007). The cropping systems are diverse. Depending on the area, cash crops include coffee, tea, pyrethrum, sunflowers, maize, cotton, fruits and vegetables (Kayiso 1993). Livestock rearing is important. In 1990, between 40 and 60% of households owned cattle (Kayiso 1993).

In this study I focus on the Ugandan side of Mt Elgon. A more detailed description of the population and land use around the park in Uganda is given in Chapter 2, as well as a detailed history of its management. Initially a Forest Reserve, Mt Elgon became a National Park in 1993 (Chapter 2). Projects trying to address both conservation and development have been implemented in both Kenya (MEICDP[6], MERECP[7]) and Uganda (MECDP[8], MERECP) since the late 1980s (UWA 2000, LVBC 2009). In Uganda the policy framework allows for community participation in management and benefit sharing (Vedeld et al. 2005). After two pilot agreements in 1996, collaborative management arrangements were

[6] Mount Elgon Integrated Conservation and Development Project (1998-2001?)
[7] Mount Elgon Regional Ecosystem Conservation project (includes both countries) (2005-2009)
[8] Mount Elgon Conservation and Development Project (1990-2002)

set-up with about 30 parishes (Hinchley et al. 1998, Scott 1998, Kato and Okumu 2006). In 2011, 66 agreements varying from boundary management agreements, to resource use and beekeeping agreements were in place in 36 parishes (UWA, unpublished data).

The implementing institutions for conservation in Uganda are the Uganda Wildlife Authority (UWA) and the National Forest Authority (NFA). The International Conservation Union (IUCN) is the main international conservation organization that influences the conservation and development agenda through its projects, in collaboration with research organizations such as the World Agroforestry Center (ICRAF). MERECP, implemented by the Lake Victoria Basin Commission (LVBC), aims to establish Mt Elgon as transboundary protected area including both the Ugandan and Kenyan sides. This process is largely driven by international conservation actors and donors including IUCN and the Norwegian Agency for International Development (NORAD) (details in Petursson 2011). In Uganda the FACE Foundation, together with UWA ran a carbon offsetting project that helped restore parts of the previously encroached areas of the park by replanting native species. The project was controversial from a human rights point of view because in some areas it led to the forceful eviction of people who had settled inside the official park boundary (see Lang and Byakola 2006 and ensuing discussions in the press). In these areas most planted trees were later destroyed whereas in others they were left to regenerate (see Chapter2). Other conservation and/or people oriented NGOs include the World Rainforest Movement (WRM), Climate and Development Initiatives Uganda, Action Aid Uganda.

1.2.4. Previous research

A number of studies describe institutional and socio-economic processes on Mt Elgon, particularly in Uganda. They found that numerous boundary demarcations, corruption in the land allocation process of resettlement schemes, and evictions of people from the park have fuelled conflicts between local people and protected area management, which amplified illegal use of forest resources and land (e.g. Himmelfarb 2006, Lang and Byakola 2006, Norgrove and Hulme 2006). Collaborative management approaches were found to be promising, also to improve relations between park management and local communities, but suffering from a number of problems including weaknesses in local management institutions, a lack of ownership and problems of monitoring and local rule enforcement (White and Hinchley 2001, Sletten 2004, Kato and Okumu 2006). Various student projects have found that local communities surrounding the park are highly reliant on park resources for their livelihoods, in particular as safety nets, and that the change in management regime from forest reserve to national park reduced their access to forest resources (Jewsbury 2001, Katto 2004, Namugwanya 2004, Gosalamang et al. 2008).

Petursson (2011) found that, considering the institutional complexities on both sides of the mountain, establishing a transboundary protected area management regime on Mt Elgon would increase the marginalisation of local community interests and rights.

Only a few studies measured human impacts on forest resources. One study reports the impacts of bamboo harvesting (Scott 1994b), one the impacts of grazing and former cultivation on plant communities (Reed and Clokie 2000) and others the impacts of grazing and former settlement on birds and small-mammal populations (Project Elgon 1996, 1997a, b, Reed and Clokie 2000). The latter five studies were restricted to one area to the north of the mountain, with a history of grazing within the forest. In the early 1990's Scott (1994a) made an assessment of the use of park resources by local communities (Uganda) including profiles per parish for 6 neighbouring parishes. She determined that the extraction of most non-timber forest products on the Ugandan side of Mt Elgon was likely sustainable at the time of her study, but that there were risks of depletion associated with increased commercialisation of timber products (Scott 1994b, Scott 1998). Forest cover change and its drivers were assessed on the Ugandan side in the 1990s (Otte 1991, van Heist 1994) and on the Kenyan side in the late 2000s (Petursson et al. 2012), but these studies did not seek to explain local variation.

1.3. Study objectives and approach

I based this project and my methodological approach on the DEED (Describe, Explain, Explore and Design) framework of the Competing Claims on Natural Resources programme of Wageningen University (Giller et al. 2008). I sought to describe and explain the dynamics between various larger scale processes (governance, policy, management, socio-economic drivers), their influence on local motivations for forest use and the impacts thereof on local forest cover and conservation in Mt Elgon (Figure 1.1). I then explore the implications of my findings in light of existing theories on human-environment interactions, the global discussion on drivers of forest change, and the role of local access to forest resources. Finally, I consider how this affects the design of current and future interventions or management options such as collaborative management, alternative resources projects or PES schemes related to REDD+. The research also contributed to a project by the Centre for International Forestry Research (CIFOR) and the World Agroforestry Centre (ICRAF) to integrate improved livelihoods and biodiversity conservation goals in tropical landscape mosaics.

I aimed to assess how conflicting goals by different actors (conservationists, local communities, politicians etc.) led to various outcomes for the forest on Mt Elgon, Uganda under different historical contexts and what this means for long term conservation efforts. I also aimed to explore the factors that influence local people's motivations to respect rules and regulations (Chapter 2), their use of forest resources (Chapter 3), their dependence on the forest as a source of fuelwood and the impacts thereof on forest conservation (Chapter 4). I explore the use of simple field methods in combination with radar remote sensing technology for mapping and monitoring carbon stocks on Mt Elgon in REDD+ projects (Chapter 5).

The goal of this study was to draw lessons for and contribute to the wider debate on conservation and development (Minteer and Miller 2011), as well as provide empirical evidence for human-environment interaction theories beyond simple models of population and poverty (Lambin et al. 2001). The analysis was conducted both for the whole of Mount Elgon in Uganda, including the historical larger scale contexts and influences that have taken place, and at the local scale, where different types of livelihood strategies and access to resources are assumed to have different impacts on these resources. The analysis of changes in forest cover and of the impacts of local uses on forest structure and diversity in relation to changing institutional and socio-economic contexts contributes to the understanding of these relations. It then provides a basis for the exploration of future options and scenarios for more ecologically and socially sustainable management arrangements. I also explore implications of my findings for the potential role of PES schemes such as REDD+ in degraded, human impacted forests under pressure.

Institutional, socio-economic and field data were linked to remote sensing and GIS in order to investigate the interactions between forest cover, changing contexts, local livelihoods and impacts on forest structure and diversity (Liverman et al. 1998, Fox et al. 2003). The specific objectives of the study were to:

1. Understand what affected forest conservation outcomes locally, and to unravel the success of conservation strategies under different political, institutional and socio-economic contexts.
2. Examine how long-term use by local neighbouring communities has influenced forest structure and diversity using field plots on transects into the park, in four sites.
3. Characterise and assess the effects of fuelwood collection and other practices on the availability and distribution of dead wood in field plots into the park in four sites.

4. Assess the above ground biomass and carbon content of the degraded and less degraded forest in Mt Elgon National Park.
5. Reflect on the implications of these results for forest conservation on Mt Elgon as well as in other places with high population densities around protected areas and with major competing claims on land and resources between conservation and local livelihood goals.

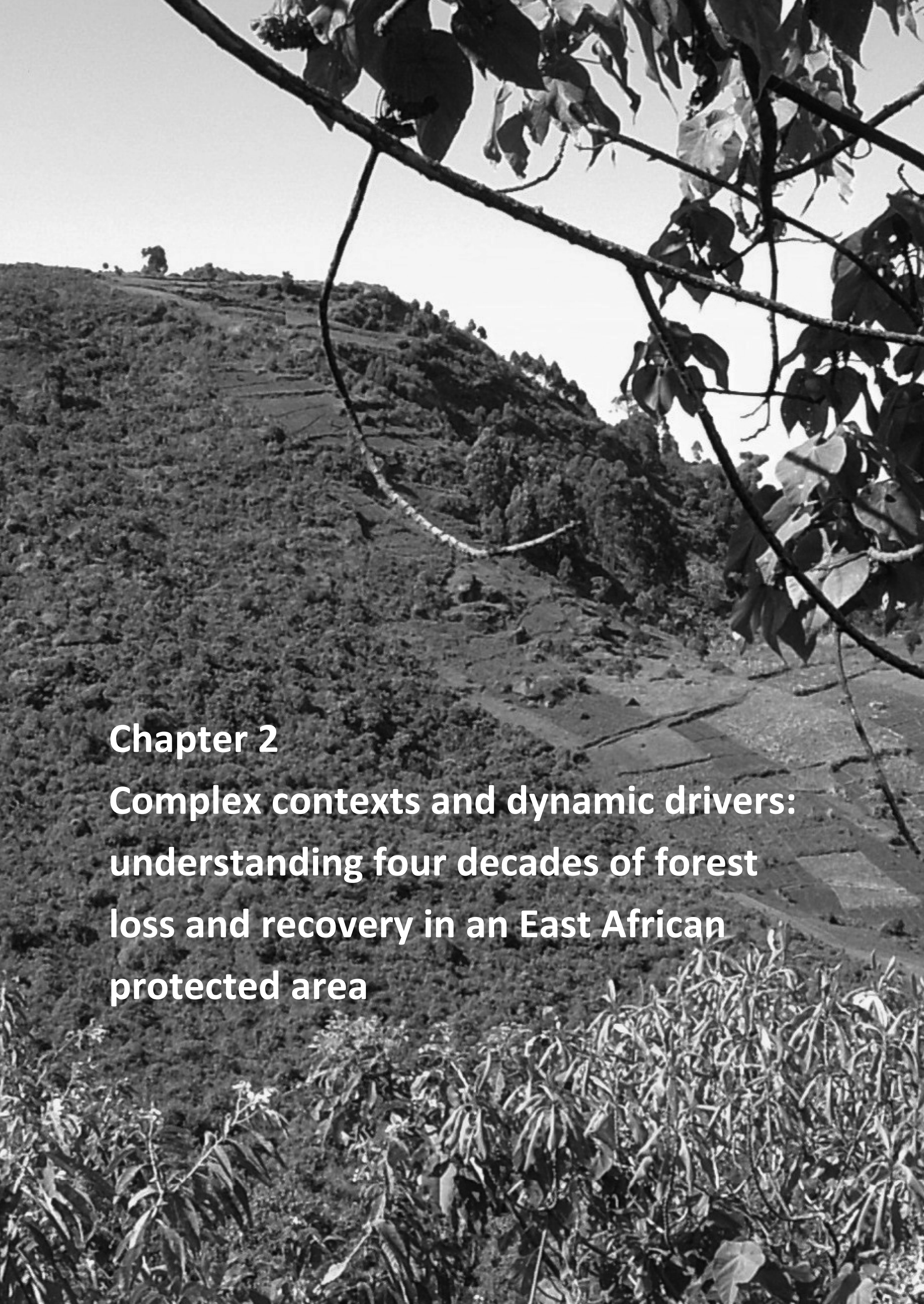
In **Chapter 2** I analyse the causes of local forest cover loss and recovery on Mt Elgon, Uganda. I created a series of four forest cover maps from multi-date satellite image classification of Landsat images between 1973 and 2009. Forest cover and forest cover change was then linked to regional to local historical contexts, population data, socio-economic and livelihoods information acquired in 14 villages with different livelihood types and histories and from interviews with the protected area management authorities (UWA).

In **Chapter 3** I examine how local scale variation in human impacts influenced forest structure and tree species richness on Mt Elgon. For this, I assessed basal area (BA), stem density, diameter at breast height (dbh) and indicators of human activity (e.g. signs of fire, tree-cutting, grazing etc.) in 343 plots on transects into the park in four study sites.

In **Chapter 4** I study the characteristics and impacts of firewood collection. I interviewed 192 households on firewood use and collection and on tree planting. Additionally I surveyed dead wood species and availability using line 48m intercept methods in 81 of the 343 plots described in Chapter 3. I link species preferences for firewood, volumes of dead wood and basal area of standing trees for those species to evaluate depletion in each site.

In **Chapter 5** I assess the above ground biomass (AGB) on Mt Elgon. I explore the use of field-derived AGB data in combination with satellite L-band synthetic aperture radar backscatter data (ALOS PALSAR) to monitor the impact of forest degradation on biomass in relation to planned local REDD+ schemes. I also explore the potential of direct basal area estimations as an effective field method to assess carbon stocks.

Lastly, **Chapter 6** discusses the complexity of human impacts on forest conservation in a densely populated environment under changing contexts of policy and governance. In this chapter I reflect on the impacts that the *competing claims* on land and resources between conservation actors and local communities have had over time on Mt Elgon. I discuss the findings presented in the previous chapters in light of existing theories on human-environment interactions and what this means for the design of more locally appropriate and inclusive approaches to forest management, for the importance of alternative resources and the potential of “new” interventions. I discuss the lessons that these offer to the wider conservation and development debate.



Chapter 2
Complex contexts and dynamic drivers: understanding four decades of forest loss and recovery in an East African protected area

Abstract

Protected forests are sometimes encroached by surrounding communities. But patterns of such cover change can vary even within one given setting – understanding these complexities can offer insights into the effective maintenance of forest cover. Using satellite image analyses together with historical information, population census data and interviews with local informants, we analysed the drivers of forest cover change in three periods between 1973 and 2009 on Mt Elgon, Uganda. More than 25% of the forest cover of the Mt Elgon Forest Reserve/National Park was lost in 35 years. In periods when law enforcement was weaker, forest clearing was greatest in areas combining a dense population and people who had become relatively wealthy from coffee production. Once stronger law enforcement was re-established forest recovered in most places. Collaborative management agreements between communities and the park authorities were associated with better forest recovery, but deforestation continued in other areas with persistent conflicts about park boundaries. These conflicts were associated with profitability of annual crops and political interference. The interplay of factors originating at larger scales (government policy, market demand, political agendas and community engagement) resulted in a “back-and-forth” of clearing and regrowth. Our study reveals that the *context* (e.g. law enforcement, collaborative management, political interference) under which *drivers* such as population, wealth, market access and commodity prices operate, rather than the drivers *per-se*, determines impacts on forest cover. Conservation and development interventions need to recognise and address local factors within the context and conditionalities generated by larger scale external influences.

Keywords: Forest cover change, conservation management, protected areas, local livelihoods, coffee, Mt Elgon

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2.1. Introduction

Even though protected areas in the tropics have generally reduced deforestation within their boundaries (Bruner et al. 2001, Naughton-Treves et al. 2005), forest loss still continues in many (DeFries et al. 2005, Nagendra 2008). This deforestation threatens the provision of forest-derived services. These services range from climate regulation and biodiversity conservation, to water-catchment protection, to providing local populations with food and timber (Millennium Ecosystem Assessment 2005). Protected forests in East Africa, for example, often serve as important water catchments supporting high densities of people. They also attract substantial tourism and host rich biodiversity. One of these forests, on Mt Elgon (Kenya, Uganda), provides water for more than 2 million people in the surrounding districts (Figure 2.1) and has a rich and remarkable history of both forest loss and forest recovery (van Heist 1994, KWS et al. 2001).

Across the tropics the underlying drivers and proximate causes of deforestation have been the subject of numerous studies (Geist and Lambin 2002). Population pressure and rural poverty, leading to agricultural expansion, dominate the global discussion on the causes of forest loss in the tropics (e.g. Allen and Barnes 1985, Uusivuori et al. 2002, Lung and Schaab 2010). By contrast, reviews show that these factors are seldom the principal determinants of when and where forest cover is lost (e.g. Rudel and Roper 1996, Angelsen and Kaimowitz 1999). Multiple political, institutional, economic and social forces operating at the local, national and global level interact to determine the patterns of tropical deforestation (Angelsen and Kaimowitz 1999, Lambin et al. 2001, Geist and Lambin 2002, Carr et al. 2005). The significance of different management arrangements, including the degree of community involvement, remains debated (Hayes 2006, Southworth et al. 2006). Deforestation by small scale farmers reflects marginal choices about whether and where to clear (Sheil and Wunder 2002). Such choices depend on the availability of the resources needed for agricultural production, infrastructure, markets, perceived costs and benefits and alternative options outside agriculture (Kaimowitz and Angelsen 1998, Angelsen et al. 1999, Maeda et al. 2010). These factors are often time and location specific, but local studies using longitudinal data and linking people and place can clarify their role (see e.g. Fox et al. 2003).

We assess how changing contexts in combination with more local drivers can influence forest cover within one protected area (see also Gaveau et al. 2009, Nagendra et al. 2010). We examine cover in Mt Elgon National Park, Uganda between 1973 and 2009. Previous studies emphasized the deforestation during the civil unrest of the 1970s and 1980s (Otte 1991, van Heist 1994). Some forest recovered subsequently though clearance has remained a local concern (UWA 2000). We use a combination of data and methods to investigate the diversity of factors that affected forest clearance and recovery within a single national park (as in Ostrom and Nagendra 2006). We examine three periods broadly corresponding to weak enforcement, strong enforcement and community engagement periods, and investigated the effects of changing political, economic and social factors.

2.2. Study area

2.2.1. Mt Elgon

Mt Elgon is an extinct 4321m high Miocene volcano, shared between Kenya and Uganda. Its slopes are generally gentle (averaging less than 4 degrees), with characteristic natural terraces cut by sheer cliffs in the north, and steep slopes in the south and south-west. A parasitic vent formed the 20 km long ridge that extends towards the west. The protected area covers approximately 1120 km^2 in Uganda and 1400 km^2 in Kenya (Figure 2.1). Dry north-easterly and moist south-westerly winds determine the climate. July-August and December-February are relatively dry, although rain falls in all months (Figure 2.2). Annual precipitation in the protected area is between 1500 and 2000 mm. More rain falls on the western and south-western slopes and most falls mid-slope at between 2000-3000 m altitude (m.a.s.l.) (Dale 1940, IUCN 2005).

The mountain is an important water catchment area for the Turkwell and Lake Turkana systems, the Lake Victoria Basin, Lake Kyoga and the Nile Basin (IUCN 2005). The vegetation is composed of an afromontane forest belt (*Podocarpus* spp., *Cornus volkensii, Schefflera* spp., *Hagenia abyssinica, Olea* spp., *Prunus africana*) with large areas of bamboo (*Arundinaria alpina*) on average between 2000 and 3000 m, followed by heathers (*Philippia* spp.), and high altitude moorland (*Senecio* spp., *Lobelia* spp., *Alchemilla* spp.) (Dale 1940, van Heist 1994). Fire on the moorlands plays a role in determining the upper forest boundary (Hamilton and Perrott 1981). Mt Elgon hosts biodiversity of global significance, including 39 endemic species of vascular plants specific to Mt Elgon and many species with limited distributions, such as *Lobelia elgonensis* and *Senecio elgonensis, Hypericum afromontanum, Juniperus procera* and *Euphorbia obovalifolia* (for details see Davenport et al. 1996, IUCN 2005).

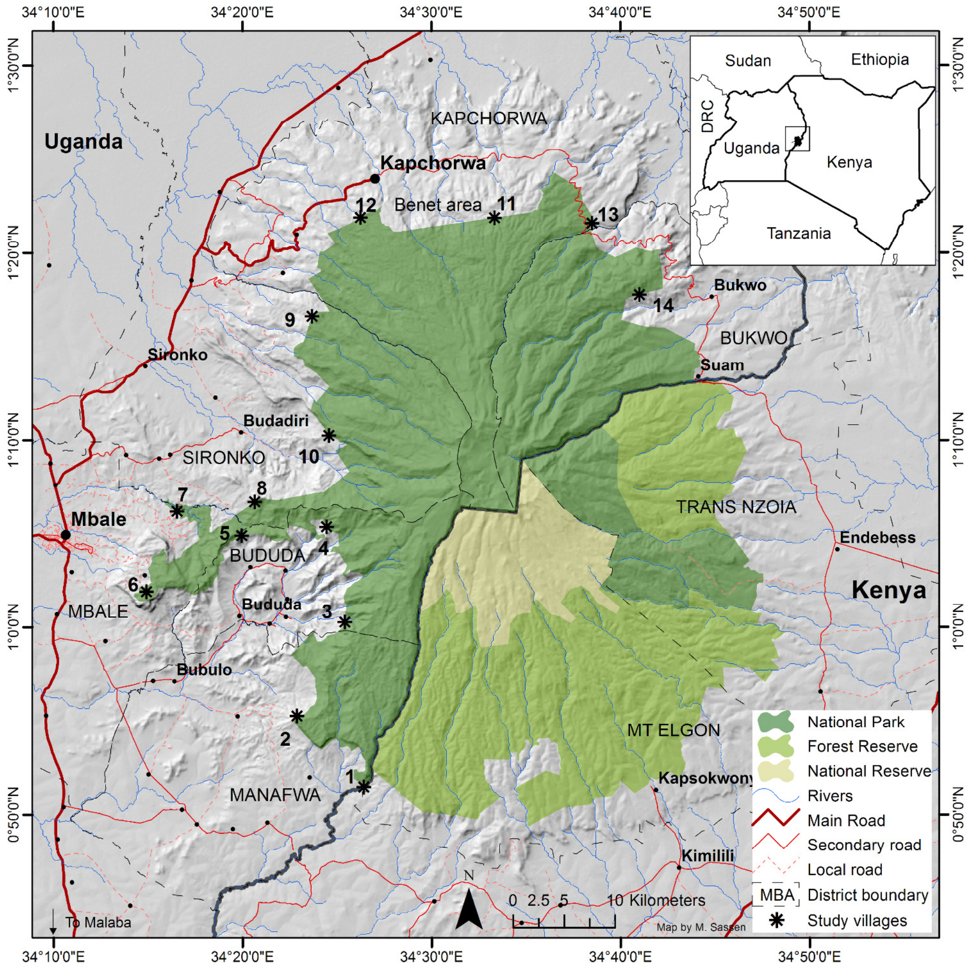

Figure 2.1. Map of Mt Elgon, Uganda and Kenya, with the location of the 14 study villages.

2.2.2. Land use

In the period covered by this study, nearly all land within 20 km from the protected area was grazed or under cultivation (van Heist 1994, IUCN 2005) (see images in Appendix 2.A). The region's volcanic soils are fertile and support intensive mixed agriculture in the south and west, known as the "coffee-banana farming system" (Kayiso 1993, ILRI 2007). Coffee (*Coffea arabica*) is commonly grown in combination with multipurpose shade trees, while stream valleys are often planted with *Eucalyptus* woodlots. On the north and north-eastern slopes extensive maize, potatoes, wheat and pasture dominate (ILRI 2007), while trees are scarce, especially nearer the park boundary. The western and south-western slopes of Mt Elgon in Uganda have been among the most densely populated and cultivated in the country since before 1960 (McMaster 1962). In 2002, human population densities in the surrounding parishes ranged from 150 p/km^2 in the north and northeast to more than 1000 p/km^2 in the west. Average annual population growth rates ranged between 2.5% and 4.3% (UBOS 2002a, b, d).

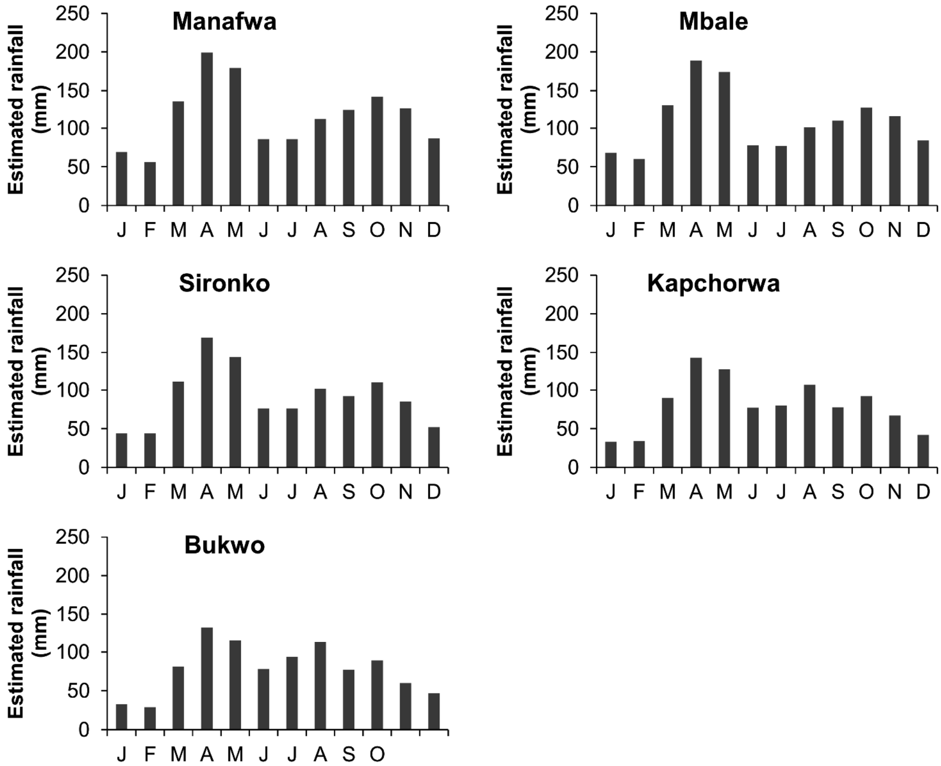

Figure 2.2. Estimated average monthly rainfall in five districts around Mt Elgon (2007-2011). See Figure 2.1 for district boundaries. Data for Manafwa district includes Bududa. (Data source: FAO/GIEWS 2012).

Two ethnic groups predominate around Mt Elgon in Uganda: The Bagisu, of Bantu origin, in the south and south-west and the Sabiny, of Nilo-Cushtic origin in the north and north-east. Coffee was introduced in 1912 on the north-western slopes of Mt Elgon. The crop helped the agriculturalist Bagisu gain substantial economic and political power and according to Bunker (1987), the region had the second highest per capita income in Uganda in the 1950s. Much of this power was lost during and after the political upheaval of the 1970s and 1980s and many coffee farmers diversified into additional subsistence or cash crops (Bunker 1987). The Sabiny were originally pastoralists dwelling in both the semi-natural forest grasslands and high altitude moorlands and in the lower northern plains. Since the 1980s, they turned to agriculture for cash, often under the influence of Bagisu or immigrants from the plains. Cattle remain very important (Scott 1998). Maize was introduced to Uganda before colonial times and it became an important food crop on the northern flanks of Mt Elgon (McMaster 1962). From 2003, a new murram road from Mbale to Kapchorwa has improved their access to markets, but historically the Sabiny have lagged behind in terms of education, transport, access to agricultural support and credit (Kasfir 1976).

Major local markets for agricultural produce exist in the west, northwest and on the borders with Kenya (see towns in Figure 2.1). Transport costs are high as many roads are unusable during heavy rains. Land degradation is a concern and landslides occur regularly on the relatively steep slopes on the Ugandan side of Mt Elgon (Knapen et al. 2006).

2.2.3. Management history

Uganda's forests were brought under government control from 1929, as colonial powers were concerned that expanding agricultural activities would cause forest loss and damage water catchments (Turyahabwe and Banana 2008). The management objectives of the forest reserve on Mt Elgon were protection and timber extraction. From 1955, forest clearing started on the north-eastern side of the mountain to establish pine and cypress plantations (Synnott 1968). A system of resident cultivation attracted neighbouring people who settled inside the reserve by the northernmost plantation (Government of Uganda 1996, Scott 1998, Médard 2006, Luzinda 2008). In 1968, forest management in Uganda was centralized and reserve boundaries were officially demarcated (Table 2.1).

During the years of Idi Amin (1971) and Milton Obote (1978), who took power through successive military coups, there was a general breakdown in national and local forest management institutions. In 1975, in a drive to increase national agricultural production, Idi Amin declared all land public and open for settlement (Hamilton 1985, Turyahabwe and Banana 2008). In the plains north of Mt Elgon, cattle raiding activities intensified due to the increased availability of firearms (Otte 1991, Scott 1998).This phase of instability lasted until 1986 when peace was restored under President Yoweri Museveni. From 1987 onwards, a conservation and development project and forest restoration projects were implemented on Mt Elgon (Table 2.1) (UWA 2000).

Table 2.1. Historical timeline of larger and local scale contexts for forest conservation and management on Mt Elgon in Uganda (sources: Synnott 1968, Bunker 1987, Government of Uganda 1996, Scott 1998, Luzinda 2008, Turyahabwe and Banana 2008).

Dates	National / regional context	Local events
±1910	Coffee introduced as a cash crop by colonial power	Start of Arabica coffee cultivation by the Bagisu on Elgon (northwest)
1929-1951	First national forest policy Afforestation programs for fuelwood outside existing forests Management by the Forest Department	Establishment of Mt Elgon Forest Reserve, subsequently: Mt Elgon Crown forest, Central Forest Reserve and Demarcated Protection Reserve
1955		Forest clearing for plantations in northeast. Resident cultivation system (“Taungya”)
1962	Independence from Britain	
1968	Centralization of forest management	Forest Reserve boundary marked Loss of local forest management institutions
1971-1986	Idi Amin-Obote coups, political instability. Withdrawal of foreign funding. Breakdown in forest management institutions Dysfunctional coffee buying cooperative All land declared public	Encroachment into the Forest Reserve Peak in coffee prices Bagisu-Sabiny conflicts Start diversification of cash-crops Intensified cattle raiding from the north
1983		Benet resettlement (> 1000 people)
1986	Museveni president	
1986-1988	Coffee production improvement program.	Raises in coffee prices to producers to reflect world market prices
From 1987	Rehabilitation of the forestry sector. Support for protected area management by international donors	Mt Elgon Conservation and Development Project (MECDP) supported by IUCN
1988	New Uganda forest policy	Start forest rehabilitation, re-establishment of boundaries
1989	Collapse International Coffee Organization (ICO) agreement	Low coffee prices Eviction of settlers near plantations
1992-1995	Liberalization of the Uganda coffee market Coffee price boom on world markets	Eviction of encroachers High local coffee prices
1993	Policy decision to increase the protection status of 5 forest reserves, under (UWA)	Forest Reserve converted to National Park. Boundary survey
1994-2002		Forest restoration project (UWA-FACE) funded through carbon emissions mitigation
1996		Collaborative management pilot projects
2001-2009	New forest policy, decentralization including collaborative forest management	Evictions and temporary resettlement inside the boundary in Benet (± 9km[2)]
2004-2005	Rise in coffee-prices	Boundary tracing and conflicts in southwest and south
2006	Multi-party elections and campaigns	Conflicts and encroachment
2008-2009	Increase demand for maize in Kenya and Sudan: prices doubled	Evictions in the north and northeast leading to conflicts. Temporary resettlement inside the boundary in Benet

In 1993 the forest was re-gazetted as a National Park and brought under the management of Uganda National Parks (UNP, now Uganda Wildlife Authority, UWA), except for a small protuberance north of the western arm (Figure 2.1). Increasing restrictions on local people led to the disintegration of existing indigenous forest resource management systems and sparked conflict (Scott 1998). In the early 1990s, UWA policy shifted to include more collaborative and participatory approaches to park management (Hinchley 2000). From 1999 onwards, agreements between local people and park management in the form of resource use agreements, boundary management agreements and beekeeping agreements were initiated in a number of parishes surrounding the park (Scott 1998, UWA 2000). In other places, recurrent evictions created strong tensions between local people and park management (Table 2.1).

In 1983 the Forest Department allocated land to be excised for the resettlement of forest dwelling Sabiny. This area in the north of the reserve is now called the "Benet resettlement area" (Scott 1998, Luzinda 2008) (Figure 2.1). Lowland Sabiny who had settled on the forest edge to escape cattle raiding from neighbouring tribes were also included (Government of Uganda 1996, Banana and Gombya-Ssebajjwe 2000). The process suffered from a number of problems. Land was illicitly acquired by members of the land allocation committees, some intended beneficiaries received little or no land while others preferred to stay in the forest (Government of Uganda 1996, UWA 2000, IUCN 2005). The 1993 boundary survey found that 1500 ha more land than planned had been given out. The eviction of settlers from these 1500 ha led to court cases and conflicts between local communities and park management that remain unresolved (Scott 1998, Himmelfarb 2006). In the meantime, people live in temporary settlements both inside and outside the contested area (Table 2.1). Conflicts also occur when politicians promise people land inside the park to gain support. This was exacerbated after multi-party politics were re-established in Uganda in 2005 and competition among candidates increased (Banana et al. 2010).

2.3. Materials and methods

2.3.1. Forest cover change

We used Landsat images for February 1973 (MSS), 1988, 2001 (TM) and January 2009 (ETM) (Table 2.B.1). Image dates correspond to the second half of the dry season, when differences between evergreen and seasonal vegetation are greatest. Recurrent cloud cover on Mt Elgon limited the number of useful images and prevented a more frequent time-series. We also used forest boundaries for 1967 (Department of Lands and Surveys 1967), a vegetation map with data from approximately 1990 (van Heist 1994), a 90 m digital elevation model (Jarvis et al. 2008) and conducted detailed field surveys in 2009 and 2010.

Image processing was done using the ENVI 4.0 (RSI) software. Each image was co-registered to the 2001 image using Nearest Neighbour resampling with a root mean square error (RMSE) of less than 0.4 pixels (Schowengerdt 1997). The analysis focused on the Afromontane and the Afromontane Rain Forest Zones inside the protected area boundaries, as defined by van Heist (1994). We will further refer to this area of approximately 860 km^2 as “the forest zone”.
Clouds and their shadows partly obscured the higher altitudes in the 1988 and 2001 images and were masked out using visual interpretation and classification (e.g. Martinuzzi et al. 2007) (see Appendix 2.B for details). Softwood plantations (near Village 13) were also masked-out and labelled “non-forest”. An unsupervised classification on the remainder of each image (Schowengerdt 1997) helped to identify natural spectral clusters. We then selected training areas using visual interpretation of natural clusters from the unsupervised classification, false colour composites of the images and the maps (Foody and Hill 1996). Finally a supervised classification was run using a Maximum Likelihood Classification algorithm (Schowengerdt 1997). This resulted in 4-7 classes per image that were combined into two: “forest” (minimum canopy cover of 30%, based on van Heist (1994)) and “non-forest” (see Appendix 2.B for details). Where possible, gaps from clouds and their shadows were filled manually using the 1990 map (1988 image) and Google Earth (2001 image) as references.

We used the 1967 and 1990 forest cover maps, high resolution imagery from Google Earth (2003, 0.5-2.5 m resolution) and field observations as references to validate the accuracies of the four classification maps. We then generated four confusion matrices by allocating either forest or non-forest classes to additional randomly selected sample points on each classification map and its reference (see Appendix 2.B for details). Overall accuracies ranged between 91 and 95%, with kappa coefficients between 0.79 and 0.88 (Table 2.B.2) (Congalton and Green 1999). Quantity and allocation disagreements following Pontius and Millones (2011) gave values of 5%, 8%, 4% and 1% quantity disagreement for 1973, 1988, 2001 and 2009 respectively, while allocation disagreement was 0%, 1%, 6% and 5%.

2.3.2. Population data

Using administrative boundaries and population numbers from databases at the International Livestock Research Institute (ILRI 2007) and Uganda Bureau of Statistics (UBOS 2002a, b, d), we established GIS layers of administrative boundaries for 1991 and

2002, population data at parish level for 1969, 1991 and 2002, and data on immigrants for 2002. The finest scale for which population data was available was the parish, which usually consists of one to four villages. Between 1969 and 1991 and then 2002, some parishes became sub-counties that were then subdivided into new parishes. For 1969, visual inspection of topographic maps of the 1960s helped match administrative unit names and boundaries between 1969 and 1991. After 1991, subdivisions were more frequent and involved more boundary shifting. Sometimes pieces of former parishes were divided among different new sub-counties or (groups of) parishes. Therefore we calculated population density for 2002 using the area corresponding to the parish boundaries of 1991. In all cases where boundaries had changed, we assumed that the relative population in a parish, within the larger sub-county or a group of parishes, remained constant over time. We then used the 1991 proportions as a basis to estimate population numbers over a corresponding area in other years. This method is also used by the Uganda Bureau of statistics in their population projections.

Population density for a parish *p* in the census year of interest *y* was then estimated as:

$P_{p,y} = (P_{p,1991} / P_{s,1991}) \times P_{s,y}$

With $P_{p,y}$, the unknown population of the parish of interest in year *y*, $P_{p,1991}$ and $P_{s,1991}$ the known population of the parish and its sub-county in 1991 and $P_{s,y}$ the known population of that sub-county in census year *y*. When possible we used a group of parishes instead of the larger sub-county as a reference.

2.3.3. Local livelihoods' survey

We collected livelihood-related information from 14 villages around the boundary of the protected area. The sites were spread evenly around the boundary to represent as much variation as possible. Fifteen points were originally located on the boundary line, using a GIS (ArcGIS 10). These were later located in the field using a GPS (Garmin 60CSx). As all villages are settled up to the park boundary each point was always located in a village. That village was identified in the field. The village at point 15 was excluded as the local leader was uncooperative and demanded payment. In each village, we applied a range of survey techniques including village meetings with semi-structured discussions collecting basic data on ethnicity, education, wealth, infrastructure, distance to markets, cropping and livestock feeding systems, boundary conflicts, historical changes in land use and collaborative management agreements (McCracken et al. 1988). Scoring exercises were used to gauge and understand people's perceptions on the relative importance of agricultural land and forest (see Sheil and Liswanti 2006 for a review and discussion of these methods). We conducted the exercise separately with groups of men and women volunteers in each village. Villages identified different land and forest types. For comparison among villages, we aggregated the scores into two classes: agricultural land and forest. Informal conversations with villagers during our stay in each village (3 days) allowed us to cross-reference information gathered during the more formal meetings. We also conducted semi-structured interviews with UWA personnel based at Mt Elgon.

2.3.4. Data analysis

To analyse changes in forest cover against local population and livelihoods data, their area of influence had to be determined. We drew a 2 km wide zone parallel to the boundary (after excision) inside the protected area. This captured the area where the initial wave of encroachment for agriculture took place in the 1970s and 1980s (Hamilton 1985 and this study), while at the same time avoiding much of the upslope cloud cover and bamboo areas, which experienced natural death and regrowth during the study periods. All villages around the park boundary are adjacent, and the areas they use in the forest overlap (Scott 1994a). Traditional clan boundaries, corresponding to historical use zones in the forest, did not apply during the land rush that took place during the 70s and 80s (villagers, personal communication). As it was not feasible to exactly delimit their area of influence, we assigned each study village an area using the Euclidean allocation tool in ArcGIS 10, which allocated each cell in the buffer zone to the nearest sample point (Figure 2.1). Forest cover change is expressed as a net % change between end and base year (positive = net gain, and negative = net loss). The three periods are: 1973-1988, 1988-2001 and 2001-2009. The % change is then converted to an annual rate of change (assuming a simple linear rate).

Rate of change (%) in region a: $R_a = (100 \times (F_a2 - F_a1) / S_a) / (t_2 - t_1)$

Where F_a1 is the area of forest in region a at time t_1, F_a2, the area of forest in region a at time t_2, S_a the area of region a, t_1 the base year and t_2 the end year of the period. Remaining areas of cloud cover that could not be filled with other images were removed from all dates.

We used inter-battery factor analysis (Tucker 1958), also known as principal component-based coinertia analysis (Dray et al. 2003) to investigate the relationships between two groups of variables. The first group included the livelihood variables, population density in 1969, population density change between 1969, 1991 and 2002 and scores for forest or agricultural land types. The second group consisted of the forest cover and forest cover change rates. Inter-battery factor analysis was chosen instead of canonical correlation analysis, because of the small number of villages (14) compared with the number of variables (31+7). Inter-battery factor analysis searches for normalized linear combinations of the variables in each set, such that their covariance is maximized. The statistical significance of the relationship was tested using Monte Carlo permutation (999 permutations). The results are presented in a biplot with arrows and points for variables and villages (Gabriel 1982). We focus on the correlations between the two sets of variables – livelihoods and forest cover – and also show the variation of livelihood variables among villages. All variables were standardized to zero mean and unit variance. The analysis and statistical test were carried out using the R-package ade4 (Dray and Dufour 2007) and the graph was made with Canoco for Windows (ter Braak and Šmilauer 2002). We tested for differences in forest cover change between villages with and without collaborative management agreements separately using non-parametric Mann-Whitney U tests in the SPSS software package (IBM SPSS 18).

2.4. Results

2.4.1. Forest cover change on Mt Elgon, Uganda

In the north and northeast of Mt Elgon around 50 km^2 of forest had already been cleared before 1973, around the plantations and on the northern edge of the reserve (NF-NF in Figure 2.3). Between 1973 and 2009, patterns of forest loss and recovery in the protected area varied considerably among locations (Figure 2.3 and Table 2.2). The annual average forest loss and recovery rates in the forest zone for each period are summarized in Figure 2.4. During the 1970s and 1980s, more than a quarter of the remaining forest cover in the forest zone was lost, at a rate of almost 12% per year (F-NF in Figure 2.3 and Figure 2.4). Between 1988 and 2001, many formerly-encroached areas on the western side started recovering (NF-F in Figure 2.3). Overall annual rates of forest recovery (6%) compensated new losses (4.5%) past the officially mapped 1993-boundary in the Benet resettlement in the north (Figure 2.3 and Figure 2.4). Recovery continued in many places of the northwest and west between 2001 and 2009, but the trend was reversed to the southwest of the mountain. The changes seen on the higher slopes reflected natural bamboo dying and its regeneration (Figure 2.3).

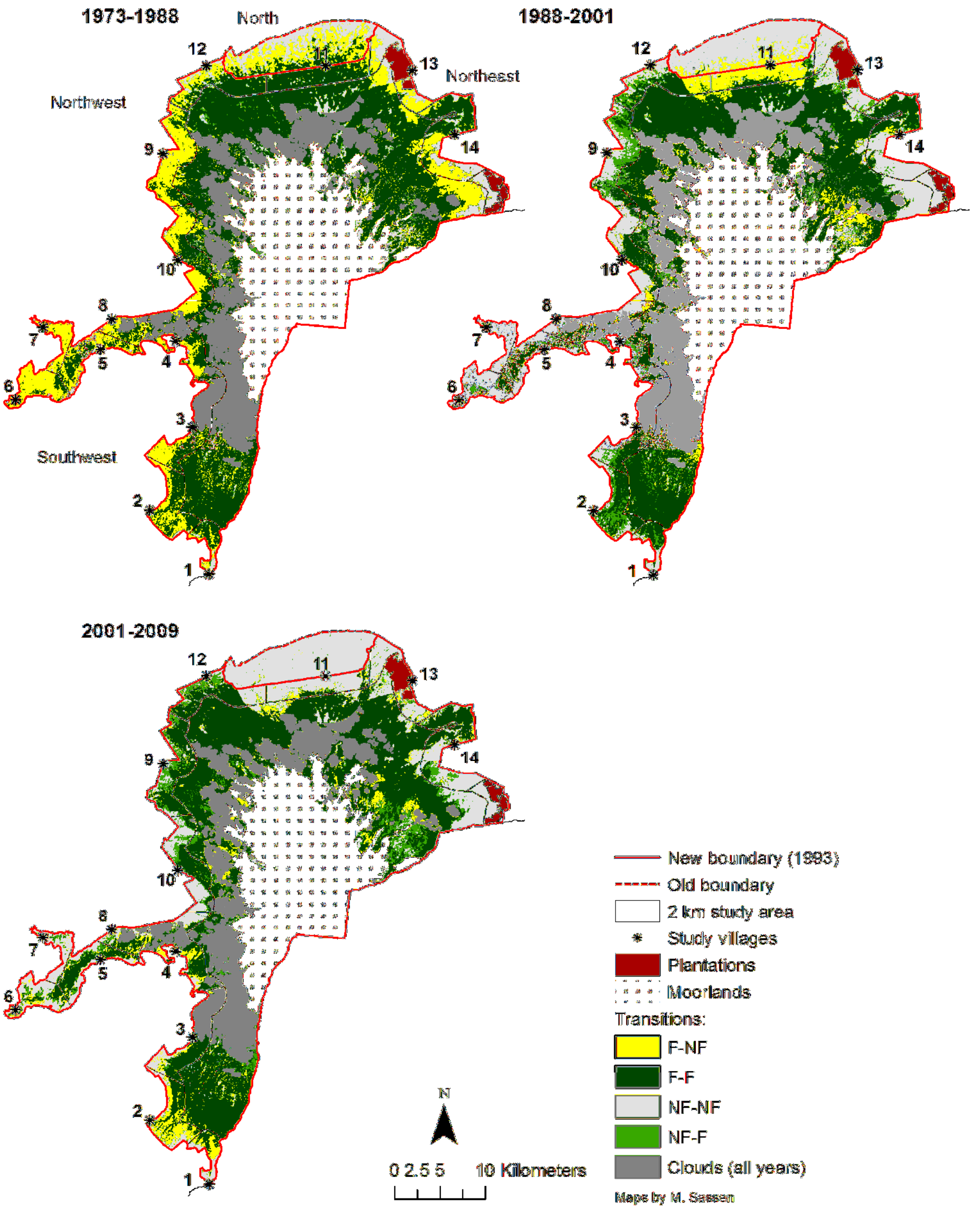

Figure 2.3. Forest cover on Mt Elgon, Uganda in 1973, 1988, 2001 and 2009 derived from classification of Landsat satellite images (see text for details and accuracy of the methods employed). Transitions at higher altitudes are related to die-off and regrowth of bamboo areas.

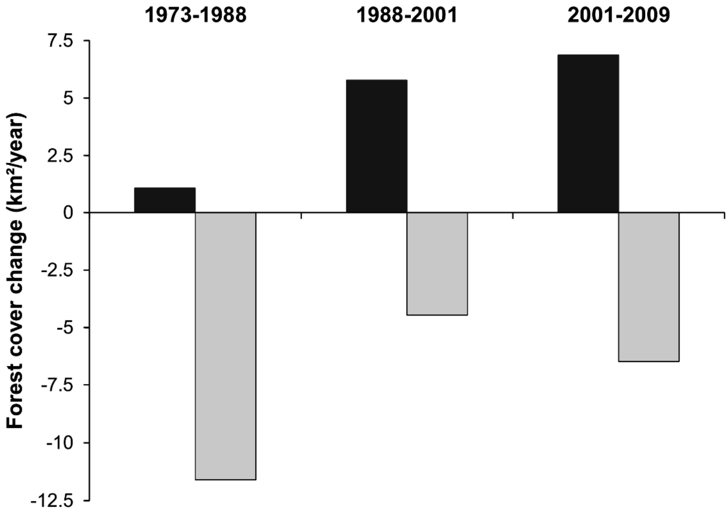

Figure 2.4. Annual average rates of forest loss (negative rates) and recovery (positive rates) on Mt Elgon, Uganda during three time periods between 1973 and 2009.

Table 2.2. Forest cover, proportion of forest cover and forest cover change for the study villages.

	Forest cover (km^2)				Forest cover (%)				Rate of forest cover change (%/y)		
Village	**1973**	**1988**	**2001**	**2009**	**1973**	**1988**	**2001**	**2009**	**1973-1988**	**1988-2001**	**2001-2009**
1	7.9	5.3	6.2	3.0	79	53	62	30	-1.7	0.7	-3.9
2	21.8	12.4	17.8	7.8	95	54	78	34	-2.7	1.8	-5.5
3	15.3	8.0	11.9	10.3	95	50	74	64	-3.0	1.9	-1.2
4	20.2	10.2	13.0	9.6	97	49	62	46	-3.2	1.0	-2.0
5	17.8	8.2	10.9	12.0	87	40	53	59	-3.2	1.0	0.7
6	16.7	2.7	5.1	5.1	94	15	29	29	-5.3	1.0	0.0
7	8.9	0.6	0.5	1.5	93	6	5	16	-5.8	0.0	1.3
8	7.9	3.6	4.6	4.1	96	44	56	49	-3.5	0.9	-0.8
9	27.9	7.1	17.8	23.6	96	24	61	81	-4.8	2.8	2.5
10	24.7	13.0	11.1	14.8	92	49	41	55	-2.9	-0.6	1.7
11[a]	22.9	20.9	5.4	3.5	96	87	23	14	-0.6	-5.0	-1.0
12[a]	21.8	14.5	14.1	17.7	84	56	54	68	-1.9	-0.1	1.7
13[a]	9.1	0.7	2.5	1.6	52	4	14	9	-3.2	0.8	-0.6
14[a]	27.9	17.0	17.8	19.3	75	46	48	52	-1.9	0.1	0.5

[a] Sabiny dominated villages, the others are Bagisu dominated.

2.4.2. Population, local livelihoods and forest cover

The area cleared under the resettlement exercise is not included in the following results as it is formally outside the protected area (Figure 2.1). From here we focus on the 2 km study zone as described in the methods.

Livelihood variables (Appendix 2.C), including population (Table 2.3) and forest cover and cover change were significantly correlated (permutation test based on inter-battery factor analysis, $p = 0.017$). A biplot (Figure 2.5) summarizes the correlations between the livelihood variables and forest cover and cover change, as well as the variation in the livelihood variables among the villages (see also Table 2.4).

The first (horizontal) component represents 57% of the sum of squared correlations and the second (vertical) component 23%, together 80%. The third component is not shown as it adds only 13%. The first component largely characterizes the two main land use systems (Figure 2.5): on the left hand side, maize-based villages, for both cash and staple, with pastures and high scores for forest and, on the right hand side, older coffee-based villages, with banana or maize as a main staple and high scores for agricultural land and high population density. Maize-based villages were poorer with a larger proportion of thatched roofs, using mainly pasture and the forest as a source of fodder (Table 2.4). Villages with coffee as the main cash crop had access to formal credit and were generally wealthier with more metal roofs and planted grass to feed their mainly stall-fed livestock (Table 2.4). Component two is associated with accessibility expressed by distance to roads and markets, better education and the estimated number of tree species planted in the village (Figure 2.5).

In 1973 forest cover was still high even in the densely populated areas on the western and south-western slopes (Table 2.2, Figure 2.5). However, between 1973 and 1988 most forest was cleared near these densely populated, older, wealthier and coffee-(banana)-based villages (Figure 2.5). Population increased a great deal in maize-based villages during this period (Figure 2.5), as immigrants were attracted to the resettlement and plantation areas (Villages 11 and 13) (Table 2.3). This did not significantly affect forest cover near the still forested resettlement area as loss inside the new boundary was limited (Figure 2.5). Between 1988 and 2001 however, population continued to increase in the resettlement area and forest was cleared beyond the intended boundary (Figure 2.3 and Figure 2.5). While in 2001 forest cover had somewhat recovered in coffee- based villages, it was cleared again in the south (Villages 1 till 4) between 2001 and 2009 (Figure 2.5).

During this period, recovery continued in coffee-banana villages, especially those with more educated people (Figure 2.5).

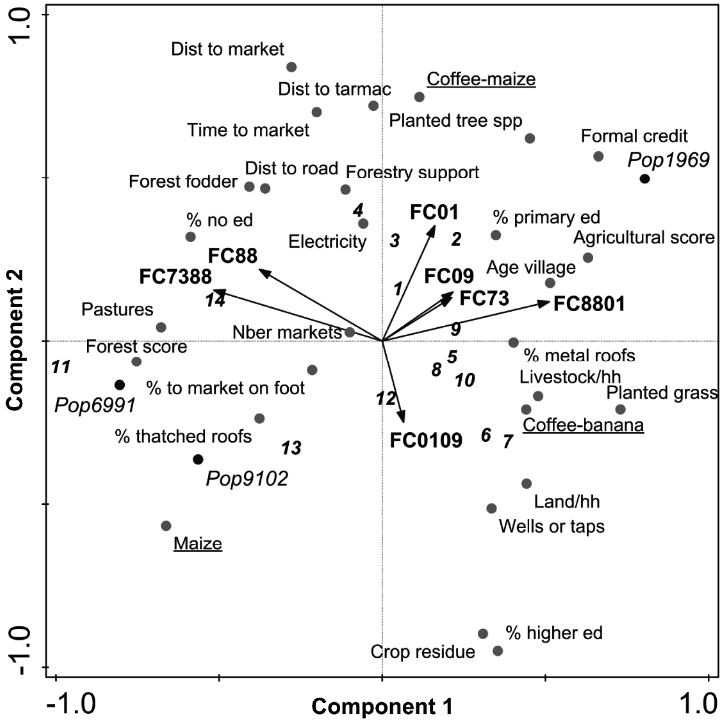

Figure 2.5. Results of the co-inertia analysis of livelihoods variables, population and forest cover and cover change with villages (numbers). Forest cover and forest cover change variables are indicated with the abbreviation FC, followed by a shortened year or period indication (e.g. FC73 or FC7388). Population density is indicated with “pop” followed by the year (e.g. pop1969) or the period (e.g. pop0109). In the plot the forest cover arrows point - from their average value - in the direction of their steepest increase and their lengths express the displayed variance in forest cover. Less than average values, in particular negative change (loss) in forest cover between dates can be obtained by extending the arrows beyond the plot origin. The approximate correlation between the variables and forest cover or cover change can be read by projecting the variables onto the line overlaying the FC arrows. The absolute value of the inferred correlation is proportional to the distance of the projection point from the plot origin. The correlation is positive if the projection point is on the same side of the origin as the arrow-head and negative otherwise. The approximate ranking of the values of variables in villages can be read by projecting the villages onto the imaginary line running through the variable point and the origin of the plot. Villages 1-10 are Bagisu and villages 11-14 are Sabiny dominated.

Table 2.3. Population, population density and proportion of immigrants in the parishes where the study villages are located.

	Population numbers			Population density (/km^2)			Immigrants (%)
Parish	**1969**	**1991**	**2002**	**1969**	**1991**	**2002**	**2002**
1	5415	11420	17126	242	511	766	0.53
2	5167	7317	10127	322	456	631	0.36
3	1922	3293	4444	314	539	727	0.20
4	3058	4398	7967	259	372	673	0.26
5	5165	8093	16145	374	587	1170	0.52
6	2967	5564	8651	247	464	721	0.34
7	4418	9966	18244	316	714	1307	0.87
8	7279	9723	13581	335	448	625	0.61
9	5837	8633	11245	369	546	712	0.77
10	2101	2393	2758	314	358	413	0.26
11	284	5268	16682[b]	10	188	448	2.11
12	1809	2935	4336	211	342	506	0.74
13	781	1998	3082	62	158	244	3.54
14	1194	2277	3575	125	238	374	0.56

[b] The enumeration area for this parish in 2002 is larger than in previous years as it included a new area of temporary resettlement inside the official boundary and data on the split of population between old and new parish boundaries was unavailable. Adding this area and assuming no-one lived there in 1969 and 1991 would reduce population densities in those years by 25% but is not realistic. People were living inside the forest at low densities but not necessarily "counted".

Table 2.4. Values for livelihoods variables per group of villages.

	Coffee		
Variables	**Banana (n=7)**	**Maize (n=4)**	**Maize (n=3)**
Age of the village (years)	148	107	44
% people with a metal roof	0.7	0.9	0.3
Distance to the nearest road (km)	6.2	2.9	6.4
% of people with no education	13.1	19.7	24.3
% of people with secondary or higher education	0.2	0.3	0.1
Average area of land (acres)	1.7	1.3	0.9
Average number of livestock (equivalents)	1.2	1.5	1
Fodder: planted grass	++	+	-
Fodder: forest	+	-	++
Access to formal credit	+	+	-
Number of tree species planted in the village	5.4	4.0	2.0
Forest scores	36.4	36.6	59.5

2.4.3. Collaborative management

Of the 14 study villages, five had a resource use agreement in 2009-2010 (Villages 6 and 9-12). Informants in four of the villages with no agreement said that having an agreement would enable them to access resources without conflict with UWA (village meeting). The main reason cited for not renewing past agreements (Villages 2 and 4) or refusing any form of agreement (Villages 1 and 13) were the villager's own feelings of entitlement to access and their resentment with the park authorities concerning boundary conflicts. Forest recovery near villages with an agreement, tended to be higher (+1%/y on average) than in those without (-1.6%/y on average) (U=34, n = 13, P = 0.045). We found no significant ($p < 0.05$) relation between the presence of a resource use agreement and the first two components of the inter-battery factor analysis.

2.5. Discussion

In the following discussion we first review the quality of the cover data and analyses. Then we examine how the contexts that characterised each study period influenced how drivers such as population, wealth, market-access and commodity prices impacted on forest cover. Finally, we summarise our findings and their implications for improving conservation effectiveness.

2.5.1. Forest cover change mapping on Mt Elgon, Uganda

We achieved high overall classification accuracies (91 - 95%), which was aided by using broad aggregated cover classes (Appendix 2.B). The kappa coefficient has been widely criticized (Foody 1992, Pontius 2000). We therefore calculated an alternative classification error measure that combines pixel quantity and quality errors, as proposed by Pontius and Millones (2011). Because we compared net forest cover over time, errors in the location of forested pixels were less important than errors in total quantity of forest at our four image dates. Quantity disagreement between our images and the reference data was small (1-8%). The 1973 and 1988 maps combined the highest quantity disagreement (5% and 8% respectively) and the least allocation disagreement (0% and 1%), which consisted mainly of forest pixels being classified as non-forest. This is because the reference maps for those years had lower resolution (were more "generalized") than the image classification. For the same reason, the user accuracy was lower for non-forest than for forest in 1973 (69%) and 1988 (77%).

Any study such as ours, based on widely spaced images from distinct sources, has a limited ability to examine the finer details of landscape change. The study of change dynamics on Mt Elgon would benefit from the use of a more regular time-series, fuzzy sets and continuous mapping instead of discrete classes as used in this study (see e.g. Woodcock and Gopal 2000, Southworth et al. 2004, Southworth and Gibbes 2010, Hartter et al. 2011). The extended time intervals between our images likely obscures a more fluctuating and dynamic pattern of forest loss and recovery than we can observe from our data. Recurrent cloud cover prevented a more frequent time-series. Study areas 3, 4 and 8 were most affected by cloud and cloud shadow cover on deforested areas (Figure 2.3), and cloud cover was most important in the 1988 image. This may have caused an underestimation of forest loss in these areas, but it did not affect overall patterns (see Table 2.2). In all other areas, clouds were concentrated at the upper forest boundary (see e.g. Foody and Hill (1996) on using ancillary information to aid classification interpretation). Moreover, the deforestation results (30%) for the first period (1973-1988) of our study are comparable with those found by van Heist (1994), who reported that 28% of the forest on Elgon, Uganda had been encroached in 1989/90, and Otte (1991), who found that by 1985 30% of the forest on the western slopes had been cleared for agriculture. An overview of the main contexts and factors that affected forest cover during the study periods in the different areas of Mt Elgon is provided in Figure 2.6 and discussed below.

2.5.2. Population, wealth and agricultural expansion

The forest inside the reserve boundary was largely intact on the southern and western sides of the mountain until the mid-70s (Otte 1991, Scott 1998) (see also Figure 2.3). The breakdown in law enforcement of the 1970s and 1980s led to a *de facto* free access to most forests in Uganda and widespread encroachment into forest reserves (Turyahabwe and Banana 2008). Our data indicates that relatively wealthy people with strong agricultural traditions and high population density were more likely to clear forest for agriculture compared with their neighbours who were poorer and less densely settled (Figure 2.5). In an older study, Scott (1994a) also found that wealthier, more educated households on Mt Elgon consumed more forest products than poorer households. Moreover, a recent study of 8000 households in 40 sites across the tropics, relating poverty and the environment, suggests that wealth rather than poverty drove faster deforestation (CIFOR 2011). A combination of other factors added to the drive for agricultural expansion on the western slopes during this period. In 1979, Bagisu who had been progressively migrating into Sabiny areas were violently evicted and on returning to their home areas exacerbated pressure on land (Scott 1998). Also, despite a rise in prices (Figure 2.6), returns from coffee in the early 1980s were poor because of the collapse of the main cooperative (Table 2.1), which led to a diversification of cropping systems and additional demands for land (Bunker 1987). Only seasonal crops were planted inside the reserve due to the risk of eviction and crop slashing by UWA (local villagers, 2009, personal communication).

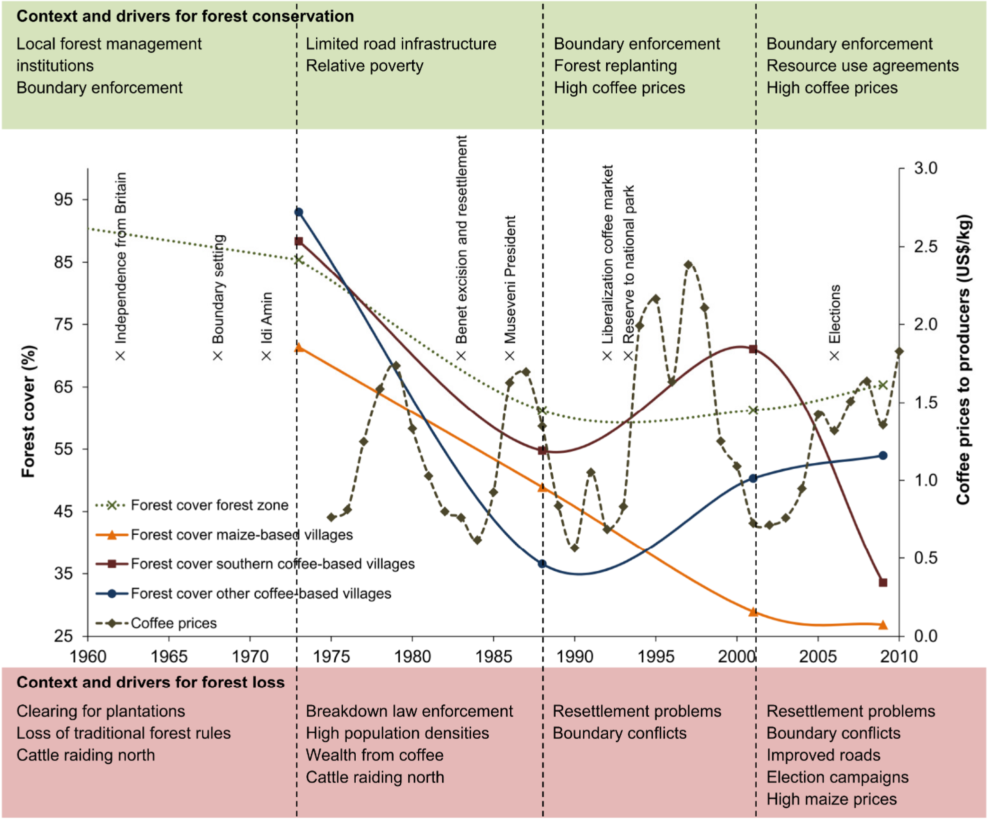

Figure 2.6. Forest cover over time in the entire forest zone of the park and in the coffee-based and maize-based study villages (2 km zone) and coffee prices. We separated forest cover for villages in the south to illustrate the difference with other coffee-based villages in the latest period. In the top rectangle are contexts and drivers that contributed to forest conservation in each period, while the lower rectangle lists contexts and incentives for forest clearance. Forest cover prior to 1973 was estimated based on 1967 topographic maps. Coffee prices to growers were corrected for inflation.

2.5.3. Law enforcement and coffee prices

Otte (1991) predicted that Mt Elgon's forests would be entirely cleared by 1990. However, between 1988 and 2001 the forest near densely populated coffee growing areas showed signs of recovery (Figure 2.3). Increased law enforcement and the change in status to national park coincided with the coffee-price boom in 1994-1995. Under a liberalized market (Table 2.1 and Figure 2.6), Bussolo et al. (2007) found that in other coffee producing districts in Uganda, gains from the coffee boom helped farmers diversify their agriculture and start other activities that helped them cope with later price drops. In Village 9, local informants said that after eviction, they decided to concentrate on their existing land outside the park, which was made easier because of good returns from coffee (local villagers, 2010, personal communication). Contrary to the previous period, this time high returns from coffee, followed by diversification during a subsequent time of low prices, did not lead to expansion into the protected area. Conversely, law enforcement did not have the same effect in the Benet resettlement scheme where conflicts about the boundary were associated with continued forest loss.

2.5.4. Commodity markets, collaborative management and elections

From the early 2000s, stiff competition among coffee buyers drove local prices upwards again (Figure 2.6). During the same period (2001-2009) coffee growing areas in the southwest (Villages 1 and 2) where forest had previously recovered, now showed important new losses (Table 2.2). According to local informants in these villages, government-promoted vanilla and chilies replaced coffee when prices went down around 2000 (Figure 2.6). They also felt that their coffee bushes were old and unproductive and the proximity of the Kenyan border provided ready markets for seasonal cash crops such as onions and maize (villagers, 2009, personal communication). Increased demand from Sudan and Kenya caused maize prices in Uganda to double between 2008 and 2009 (IFPRI 2009). In Kenya maize is the main staple but crop failures and political turmoil caused shortages.

Although the area near Village 10 has also known long standing boundary conflicts that have precluded any forest recovery (Figure 2.3), conflicts between local communities and park management over boundaries were more marked in the south (R. Matanda, Community Conservation Warden Mt Elgon National Park, 2009, personal communication). These conflicts were mainly fuelled by political interference during parliamentary election campaigns in 2001 and 2006 (Norgrove and Hulme 2006 and villagers, 2009, personal communication). The protected area is large - 1120 km^2 and 288 km of boundary - with limited staff (187 armed personnel in 2010, A. Bintoora, Conservation Area Manager, 2010, personal communication) and law enforcement alone cannot stop people encroaching into the park (Norgrove and Hulme 2006). Interactions between local people and park rangers were often negative, but seemed less so in villages with collaborative management agreements. This was observed by Sletten (2004), local rangers, villagers (2009-2010, personal communication) and M. Sassen (2009-2011, personal observation). We found that study villages with agreements tended to have better forest recovery. Nevertheless, cause and effect are hard to determine as agreements can only be implemented effectively with willing communities.

In non-coffee growing areas of the north, people gave high scores to the forest compared with agricultural land, yet forest clearing remained important (Figure 2.6). Government decisions to remove people from the forest led to a sense of alienation and to the disappearance of local forest management institutions (Scott 1998, Turyahabwe and Banana 2008 and local informants Village 14, 2009, personal communication). Cutting down or burning trees also maintains accessible pasture on the edges of the park (M. Sassen, 2009-2011, personal observation), as people lost access to traditional grazing areas inside the forest when the national park was created. Most villages in the north remain dependent on the forest for wood products because of a lack of alternatives and demand is likely to grow as population is increasing at an average rate of 4.3% per year (Kapchorwa District 2007). During the most recent study period however, clearing seemed to have slowed (Figure 2.6).

2.5.5. Lessons for conservation and development

Studies in Indonesia have shown negative impacts of rising coffee prices on forest conservation (as in this study's first period) and that effective law enforcement can reduce these impacts (O'Brien and Kinnaird 2003, Gaveau et al. 2009). With our study we add to the evidence that as long as law enforcement is effective and conflicts with park management are minimized high prices for coffee alone do not lead to forest encroachment. The intensity of conflicts between communities and authorities seems strongly dependent on politics as a process. In fact, Banana et al. (2010) identified political interference as a major obstacle to forest conservation in East-Africa.

The choice of cash crop by local farmers is strongly dependent on history, national policy incentives and regional market factors. In their study of factors driving Tanzanian farmers to expand into the forest, Angelsen et al. (1999) found that apart from population growth, agricultural product prices explained most of the deforestation in their model. This was particularly the case for non-permanent crops. In southern villages on Elgon, the returns from illegal cultivation of seasonal crops inside the park seemingly outweigh the risk of eviction for some farmers. In these places, people are especially sensitive to (and encouraging for) campaigning politicians, fuelling rule defiance and conflicts in their search for local votes. In the traditionally less densely populated areas, changes in agricultural lifestyles and increasing populations lead to new demands for land (Scott

1998). Efforts to mitigate these effects should come from both from policy and park management (see also Struhsaker et al. 2005).

As in our study, a number of recent meta-analyses support the view that collaborative management can benefit local people and improve conservation outcomes (Persha et al. 2011, Porter-Bolland et al. 2011). But conflicts need to be resolved. Attempts by park management to include local politicians in public discussions about conservation with local communities may help to strengthen governance of forest use (A. Bintoora, Conservation Area Manager, 2011, personal communication). A carefully selected combination of incentives for conservation as well as disincentives for encroachment is likely to be most effective. These must be tailored and kept updated to address local contexts and can include measures for the promotion of agroforestry or conservation-related certification or fair trade schemes. The Mount Elgon Regional Conservation and Development Project (MERECP), has recently supported payments to local communities to avoid deforestation and restore forest inside the park (LVBC 2009). This adds another potential lever with which those concerned with conservation outcomes might seek to improve effectiveness. In the longer term local interventions may be inadequate if wider regional pressures and contexts do not provide sufficient support. In any case conservation outcomes require us to consider a wide range of factors operating at different scales but determining local choices.

2.6. Conclusions

The role of protected forests as providers of ecosystem services and products is threatened by deforestation and forest degradation. Mt Elgon presents an opportunity to unravel the effects of changing political, institutional and socio-economic factors on forest loss and recovery in a protected area over an extended period of time. Protected areas are spatially and socially heterogeneous (see also Nagendra et al. 2010) and population or poverty alone did not explain the patterns we observed. The motivation for people to encroach into a park is dependent on the balance of factors that originate at larger scales, such as policy, commodity prices, law enforcement and political interests. It is the *context* under which underlying *drivers* such as population, wealth and market access operate and influence local drivers, rather than these drivers *per se*, that influences the way they impact forest cover. Even when actively policed, boundaries are easily encroached when other factors allow and even encourage it. Understanding the changing contexts and multiple influences that determine which drivers impact local choices and subsequent forest cover changes in any given time and place will help identify interventions that yield better forest protection, while also supporting local needs and development.

Acknowledgements

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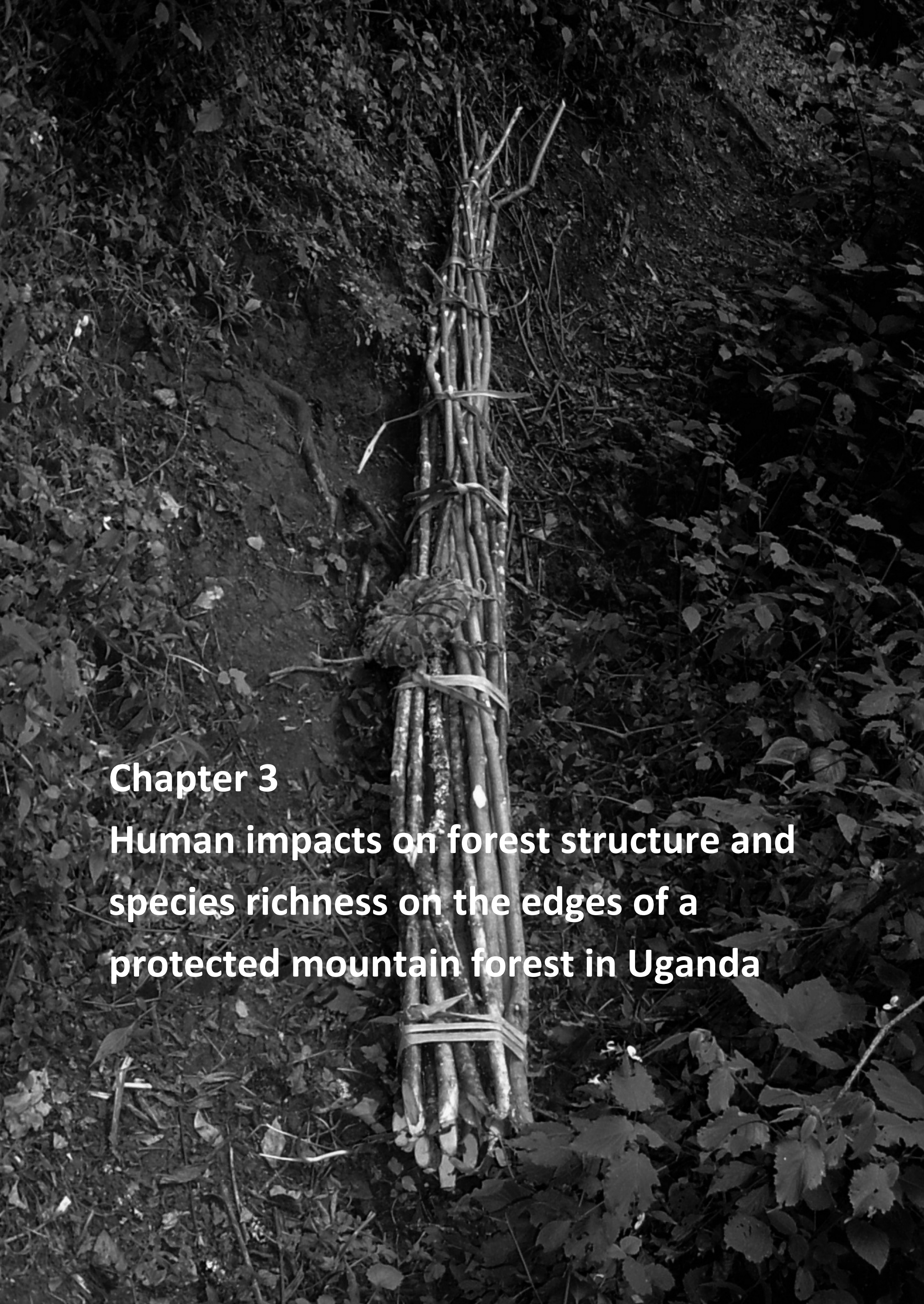

Chapter 3
Human impacts on forest structure and species richness on the edges of a protected mountain forest in Uganda

Abstract
We investigated how local scale variation in human impacts influenced forest structure and tree species richness within Mt Elgon National Park, Uganda. We assessed basal area (BA), stem density, diameter at breast height (dbh) and indicators of human activity in 343 plots in four study sites, on transects running inwards from the boundary of the park. Mt Elgon hosts the only remaining natural forest in a densely populated region (150-1000 p/km^2). All study sites suffered past encroachment for agriculture and were in various stages of recovery or renewed-clearing at the time of the study. Areas recovering from encroachment had lower mean BA (BA = 3-11 m^2/ha), dbh and often also lower stem densities than forest that had never been cleared (BA = 21-43 m^2/ha), even 35 years after abandonment and with restoration planting. Human impacts were found beyond 2 km into the park. Although most activities decreased with distance inside the boundary, their prevalence varied among sites. High coefficients of variation in BA (Cv = 0.8-1.1) and stem density (Cv = 1.0-2.2) within sites, together with the evidence of sustained human activities, suggest that forest use histories strongly influenced local forest structure. Mean BA increased with distance inside the boundary in all sites, but stem densities reflected more complex patterns. Large trees (dbh ≥ 20 cm) were most affected by former clearing for agriculture. The collection of stems used as crop-supports reduced regeneration and the density of smaller stems at one site. In another site, charcoal making was associated with the smallest mean BA and marked variability in forest structure. Grazed forest consisted of large trees with very little regeneration. On forest margins in two sites grazing, generally together with fire and tree-cutting, had eroded the forest edge and prevented regeneration. Human impacts as well as natural gradients had major impacts on species richness patterns. Several areas in intermediate states of disturbance showed higher tree species richness than either old-growth forest or more severely degraded areas. This study illustrates the fine scale variation due to local impacts within one forest.

Keywords: tropical forest; forest structure; tree diversity; human disturbance; conservation; East Africa

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3.1. Introduction

Most tropical forests, even those in protected areas, are influenced by human activity (Olupot et al. 2009, MacKenzie et al. 2012). Harvesting of forest resources to meet livelihood needs can impact forest regeneration, structure and diversity (Fashing et al. 2004, Olupot 2009), but there is scope for considerable variation with location, human activities and histories. To better manage forests for multiple local, regional and global values we need to understand human impacts and their variation at local scales. Such understanding is pertinent for forests managed for biodiversity conservation, catchment values and tourism that are increasingly considered in terms of their carbon stocks and the various benefits that they can provide to local people.

Different types and intensities of local resource extraction can lead to varying outcomes even within one forest (Thapa and Chapman 2010). For example, forest grazing leads to different impacts than cutting timber or gathering other forest products (Fashing et al. 2004, Vadjunec and Rocheleau 2009), and intensive extraction of certain highly valued species may have a greater impact on diversity than less intense forest uses (Ndangalasi et al. 2007).

Distance from settlements is often considered as a proxy for the extent of human impacts on forest (Boudreau et al. 2005), but preferred forest resources may not be evenly distributed and differ among groups of people. Environmental gradients like elevation, slope, substrate and moisture can confound results based on distance. For example elevation is known to affect tree size and species diversity (Ghazoul and Sheil 2010), but human activities are also likely to be more intensive in lower elevation forest that is easier to access than on more remote, higher elevation sites.

In densely populated landscapes remaining natural forests have generally been subjected to multiple human impacts. The resulting complexity and the challenge of defining simple cause and effect relationships may explain why these patterns have seldom been studied in detail. Yet the diversity in human activities and their impacts call for different interventions. For instance, different approaches may be required where people have long used forest as a source of medicinal products or foods, or as a location for cultural activities, than in areas where people claim forest-land for agriculture. Historical factors such as conflicts over boundaries may also influence attitudes and behaviours towards forest management (Cernea and Schmidt-Soltau 2006).

In this paper we investigate how use of the forest by communities on the edge of the Mt Elgon National Park (Uganda) has affected local forest structure and tree species richness. This is part of a linked series of studies that examine these forests and their relationship with local people. In a previous paper we have described the processes, contexts and drivers that led to localised episodes of forest loss and recovery over recent decades (Sassen et al. 2013). Here we look more closely at the nature of the resulting forests. We conducted a detailed comparative analysis of four study sites that vary in terms of the local land-use and the history of forest clearing and regeneration. We studied the variation of local activity and their ensuing impacts. We addressed the following questions: 1) How do indicators and measures of human activity vary within and among sites? and 2) How do forest structure and diversity vary with these indicators?

3.2. Site and Methods

3.2.1. Mt Elgon

Mt Elgon (4321 m) is an extinct solitary shield volcano from the Miocene on the border between Uganda and Kenya. The top is an 8 km wide crater. The slopes are generally gentle until 2800-3000 m, with characteristic steep cliffs dropping down to the plains in the north, and some steeper slopes in the south-west. The mountain's slopes are cut by river and stream valleys that run down the mountain from the caldera (Figure 3.1). A 20 km long ridge extends towards the west (Figure 3.1). Dry north-easterly and moist south-westerly winds determine the climate. Rain falls year-round but peaks in April-May and September-November. Annual precipitation is between 1500 and 2000 mm. Rainfall is higher on the southern and western slopes than on the northern and eastern slopes and most rain falls at between 2000-3000 m above sea level (Dale 1940, IUCN 2005).

Mt Elgon is an important water catchment area for several million people in the surrounding districts, for the Nile and Victoria basins as well as Lake Rudolf through the Turkwell River (IUCN 2005). A belt of bamboo and afromontane forest is found at on average between 2000 and 3000 m, followed by heathers and high elevation moorland (Dale 1940, van Heist 1994). Mt Elgon is valued for its global biodiversity values (Howard 1991). It hosts 39 endemic higher plant species as well as many species with limited distributions (for details see Davenport et al. 1996, IUCN 2005). Wildlife consists mainly of various monkeys, small ungulates and bush pig (*Potamochoerus larvatus*); rodents and birds are abundant (Davenport et al. 1996), but larger wildlife, in particular elephant (*Loxodonta africana*) and buffalo (*Syncerus caffer*), are found mainly on the Kenyan side of the mountain (van Heist 1994).

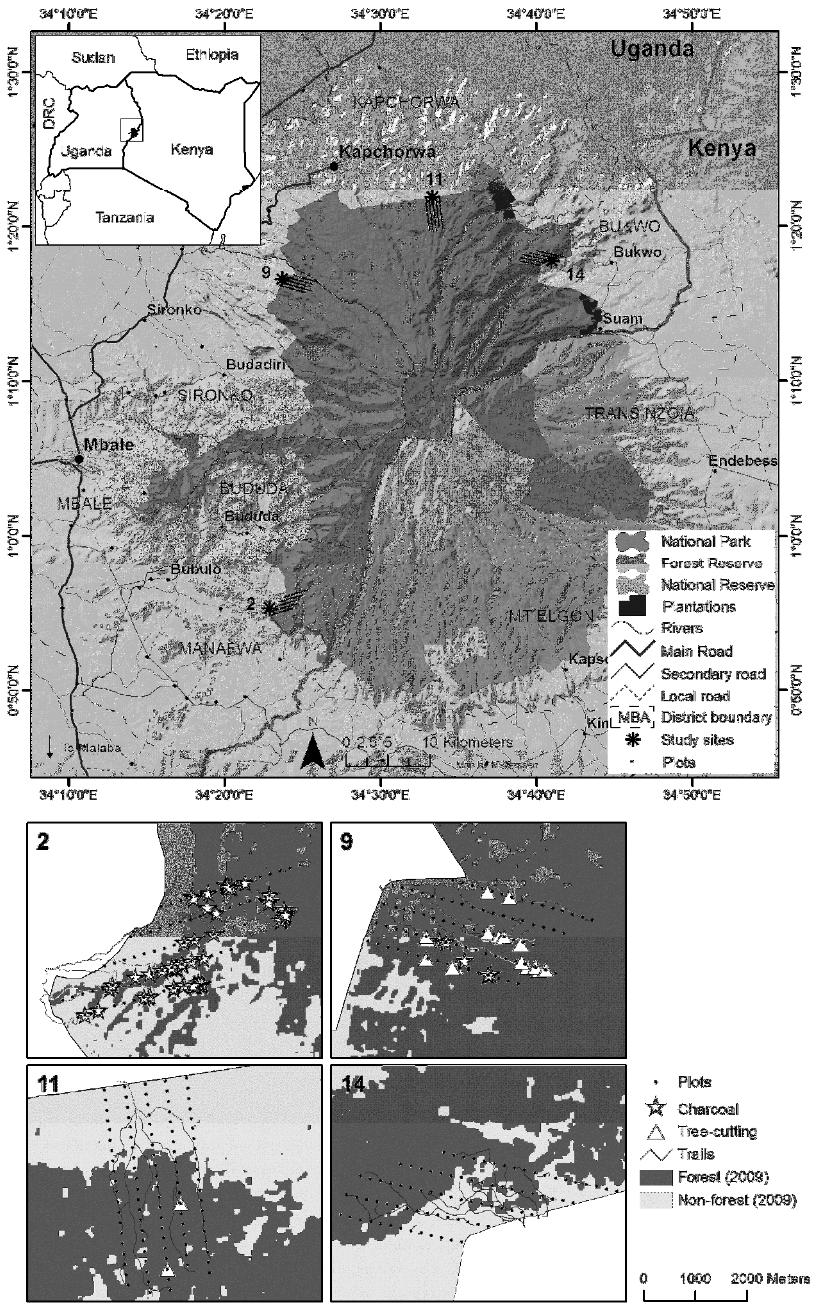

Figure 3.1: Map of Mt Elgon, Kenya/ Uganda and study sites. The small maps of the study sites show the location of the plots with a background of forest (dark) / non-forest (light) from the classification of a 2009 Landsat ETM+ image (Sassen et al. 2013). The locations of large (> 40 cm) cut trees and charcoal-burning pits encountered in the field were marked by GPS and are shown on the maps.

Mount Elgon National Park in Uganda (1120 km^2) has a long history of human influence. As long as people remember, its forests provided a broad range of products and services such as fuelwood, medicine, food, materials for construction, grazing for cattle, cultural sites (e.g. circumcision rituals, burial) and shelter against cattle raiding (Scott 1994a, Katto 2004).

The slopes of Mt Elgon in Uganda are inhabited by two ethnic groups. The Bagisu, of Bantu origin, dominate the south and south-west since around 1500 AD. Their population density reaches over 1000 people/km^2 in places (UBOS 2002c). They practice an intensive mixed agriculture dominated by coffee and banana (McMaster 1962, Kayiso 1993). Important forest products for the Bagisu include construction materials, bamboo stems and shoots and crop-supports (called "crop-stakes" from here-on) for bananas and for climbing beans (Sassen, unpublished data; Scott 1994a).

The second group is the Sabiny, a Nilo-Cushtic group of pastoralists, settled in the north and north-east from the 17th century. They lived on the edges of open grassy areas inside the forest (called "glades") on the higher slopes of Mt Elgon, until they were resettled down the mountain in the 1980s. Land for resettlement was allocated in an excision from the protected area, which was then still a forest reserve (van Heist 1994). The forest in the excised area was rapidly converted to agricultural land (Scott 1998) where people cultivate maize, potatoes, wheat and maintain pastures as cattle remain important. The Sabiny still use the forest and the glades (up to 3 km inside the park boundary) for (illegal) grazing, timber, medicine and wild foods (Scott 1994a, Norgrove 2002).

Communities living near the park are poor and suffer land shortages; nearly all land directly surrounding the park is cultivated (van Heist 1994, IUCN 2005). There are no remnant forests within 20 km around the park and people are settled up to right next to the park boundary (Sassen et al. 2013). On the western and southern slopes trees are part of the agricultural system. They are found in combination with coffee and bananas, around homesteads and in valleys planted with *Eucalyptus* woodlots. In the north, where people are more recently settled, trees outside the park are scarce, particularly nearer to the park boundary. A few isolated former forest-canopy trees remain scattered amongst the fields (Sassen et al. 2013).

Political instability from 1971 until 1986 was associated with widespread encroachment of Uganda's forest reserves (Hamilton 1985, Turyahabwe and Banana 2008) and around 30% of Mt Elgon was cleared for agriculture (Sassen et al. 2013). From 1987, forest boundaries were reinstated and restoration activities were started on the western slopes (UWA 2000). The forest on Mt Elgon was first gazetted as a reserve in 1938 and became a national park in 1993 (Scott 1998). Since the late 1990s, Uganda Wildlife Authority (UWA), which manages the park, has initiated agreements with local people that allow regulated collection of non-timber products, fuelwood and crop-stakes from restricted non-tree species (*Mimulopsis arborea* and *Vernonia* spp.) (Scott 1998, UWA 2000). Although activities such as pit-sawing declined after the establishment of the national park (Scott 1998), illegal resource extraction remained common at the time of our study. Law enforcement efforts were understaffed and overstretched but also felt that they could not always stop people from harvesting essential resources such as firewood (A. Bintoora, Conservation Area Manager, personal communication; personal observations).

3.2.2. Data collection

Four locations were selected to represent different elevation ranges and forest change histories. These locations are subsequently referred to as Sites 2, 9, 11 and 14 (see Table 3.1 for site codes and corresponding villages) – these numbers are the same as those used in Sassen et al. (2013). The communities near Sites 2 and 9 practise intensive coffee-banana based agriculture, while those near Sites 11 and 14 grow mainly maize and potatoes.

Table 3.1. Study villages, land-use and history of encroachment

	Site 2	**Site 9**	**Site 11**	**Site 14**
Villages (2011)	Bukuwa	Kinyofu/ Gibuzale	Korto/ Kamatelon	Sindet/ Kapsata
Sub-county (2011)	Bupoto	Masira	Kwosir	Kortek
Population density 2002[a]	631 p km^{-1}	712 p km^{-1}	448 p km^{-1}	374 p km^{-1}
Main cash crops[b] **(% times listed)**	Onions (34%), cabbages (34%), coffee (28%)	Coffee (69%), cabbages (35%)	Potatoes (37%), maize (28%)	Maize (98%)
Main food crops[b] **(% times listed)**	Maize (53%), banana (57%)	Banana (60%), maize (42%),	Maize (82%), potatoes (73%)	Maize (98%)
Mean number trees/ hh	33	27	3	8
Collaborative management	no	yes	no	no
Main periods of encroachment	1979-1992, 2006-2008, 2010	1979-1992	1990-2008	1985-1997, 1991-1993, 2008 (patchy)
Restoration planting 1990s	+-	++	--	--

[a] (UBOS 2002b, d, a)

[b] Based on number of people listing the crop as either their first or second cash crop (so total % > 100)

Sassen et al. (2013) assessed the patterns of forest loss and recovery around Mt Elgon, including in the four study areas. They found that people near Site 9 did not re-encroach the forest inside the park after boundaries were reinstated and forest restoration began around 1990. In the other sites however, renewed clearing and recovery took place at different times (Table 3.1). In Site 2 most regenerating vegetation and restoration planting was cleared again from 2006. Forest clearing in Site 11 started in the 1990s, when a controversy arose about the boundary of the area excised for resettlement (see also Himmelfarb 2006). In Site 14, encroachment was patchier than in the other sites (Sassen et al. 2013). At the time of our study, the parish neighbouring Site 9 had an agreement with UWA to collect resources such as fuelwood, green vegetables and medicinal plants in limited amounts, twice a week.

Fieldwork took place from November 2010 till April 2011. In each site, five parallel transects were laid out 400 m apart and as perpendicular as possible to the general orientation of the boundary while ending in relatively intact forest (Figure 3.1). The centre of the first plot on each transect was taken 50 m inside the park boundary and further plot-centres at 200 m intervals. We used a handheld GPS (Garmin 60CSx) to determine the position of the plot-centres along the transect line. We sampled 13 to 21 plots along each transect. The number depended on the travel time between each plot, which was influenced by terrain (e.g. obstacles) and vegetation (denser undergrowth required more clearing to discern the reference height of 1.30 m on tree stems). Each transect ended with at least two or three plots in areas that, according to our informants, were too far from the boundary to be used for poles and firewood. Some pit-sawing and/or charcoal burning would nevertheless still occur. We could not establish controls because any less disturbed site would not be similar, i.e. at higher elevation or less accessible. Hence we used distance effects to gauge impact intensity from assumed high to low.

We used a relascope (horizontal point sampling) approach for tree selection and direct basal area (BA) estimation (Bitterlich 1984). This method allows quick sampling of many plots, with minimal accuracy differences compared with fixed-area plots (Piqué et al. 2011). During the 360° sweep from each plot-centre, a tree was counted as “in” if it was wider than the relascope notch; borderline trees were checked with their dbh and distance from the plot-centre. The BA of each plot was calculated by multiplying the number of “in” trees with the BAF (Bitterlich 1984). Correction was later done for slope. Starting with the first plot, in every fifth plot after that we measured the diameter at breast height (dbh at 1.30 m) of each “in” tree. In these plots (referred to as “detailed-plots”), we also used a checklist to record whether branches or stems had been cut-off trees (termed "lopping"), whether the tree was alive and whether it had been planted. Sub-plots of 5 m diameter were used to measure stumps (< 1.30 m in height) and count saplings smaller than 2.5 cm dbh and between 2.6 and 5 cm, but taller than 1.30 m and seedlings shorter than 1.3 m.

We scored each plot for signs of human activity such as trails and trampling, agriculture, fire (as evidenced from charring of stems and stumps), pit-sawing, charcoal burning (pits), pole cutting (cut stems) and other signs of wood harvesting. The collection of naturally fallen firewood does not leave obvious signs, but signs of wood splitting were found as waste or piles of split stems drying away from trails. Scores were assigned on a scale of 2 (absent, present) or 3 (absent, present, severe) depending on the indicator (Table 3.3). Additionally, we recorded the location of any charcoal pits or signs of pit-sawing encountered while moving between plots (Figure 3.1).

Tree species (standing trees and recently cut or coppicing stumps) were identified by cross-referencing names given by local informants and two knowledgeable rangers (one Bagisu and one Sabiny) with available references (Hamilton 1991, Katende et al. 2000) and later at the Institute for Tropical Forest Conservation (ITFC), Uganda using photographed specimens.

3.2.3. Data analysis

Plots were classified into four categories according to encroachment history as reported by local informants: c4 = currently cleared and cultivated or grazed, c3 = cleared in the 1990s and 2000s now recovering, c2 = cleared in the 1970s and 1980s now recovering, c1 = not cultivated within living memory (called "old-growth" from here-on). In Site 11, c2 plots consisted mainly of plots in former settlement or grazing areas inside the forest from which people were relocated from halfway the 1980s.

Due to the irregular shape of the park boundary, plot position along a transect is not the same as distance inside the boundary, except in Site 11 (Figure 3.1). The actual distance to the boundary was derived for each plot using a GIS (ArcGIS 10.0). We calculated BA and density of trees, seedlings, saplings and stumps per hectare for each plot. Human activities were gauged using stump density, lopping intensity and the activity indicators. From the indicators, we calculated mean scores per activity for each site but also for distance classes into the park (Figure 3.2). Lopping intensity was calculated for each plot as the proportion of trees with signs of cutting (branches or stems).

We used two methods to calculate species richness correcting for the unequal numbers of stems per plot. We used "BiodiversityR" (Kindt and Coe 2005), based on the "vegan" package in R (R Development Core Team 2011, Oksanen et al. 2012) to calculate rarefied species richness for plots with a minimum of five stems (using Hurlbert's (1971) formulation). We also calculated Z = (species count) / log (stem count) per plot (all plots ≥ 2 stems) as a measure of species richness corrected for stem density, which allows for smaller sample sizes (Sheil et al. 1999).

Linear regression, analysis of (co-)variance (ANCOVA/ ANOVA) and non-parametric tests (Kendall's tau-b rank correlation, Kruskal-Wallis (KW), Mann-Whitney U (U)) were used to investigate the relationship among forest structure, species richness, distance inside the boundary and indicators of human activity. There were no significant interactions among the activity indicators and we did not investigate how the covariates affected interactions between factors (Yzerbyt et al. 2004). Plots in areas that were cultivated at the time of the study (in Sites 2, 11 and 14), though within the national park, lacked much natural vegetation and we therefore also analysed the data without these areas. Statistical analysis was done using SPSS version 18.0 (SPSS Inc., Chicago IL).

3.3. Results

In total we assessed 343 plots, 76 in Site 2, 84 in Site 9, 101 in Site 11 and 52 in Site 14. Elevation ranged overall between 1911 and 2877 m above sea level but ranges also differed among the four sites (Table 3.2). We recorded 2722 live stems using the relascope method, of which 593 were measured for dbh in detailed-plots (Table 3.2). In total 61 species were recorded, although 3 of these were only identified to family and 6 remained unknown. The incidence of human activity indicators per site is summarised in Table 3.2.

Table 3.2. Summary of study site characteristics

	Site 2	**Site 9**	**Site 11**	**Site 14**
All plots (detailed-plots)	76 (17)	84 (20)	101 (25)	52 (19)
Altitude range in masl	1911-2318	2152-2606	2478-2877	2238-2699
Recorded live stems	403	553	1139	627
Recorded tree species	39	32	17	30
Activity indicators:				
Plots with trails	46 (61%)	78 (93%)	67 (66%)	62 (76%)
Plots with cut stems	43 (57%)	60 (71%)	51 (50%)	35 (43%)
Plot with split wood	14 (18%)	34 (40%)	24 (24%)	11 (13%)
Plots with grazing	0 (0%)	12 (14%)	32 (32%)	26 (32%)
Plots with cultivation	19 (25%)	0 (0%)	30 (30%)	5 (6%)
Plots with fire signs	6 (8%)	7 (8%)	3 (3%)	26 (32%)
Plots with charcoal pits	12 (16%)	3 (4%)	0 (0%)	0 (0%)
Plots with pit-sawing	0 (0%)	4 (5%)	2 (2%)	0 (0%)

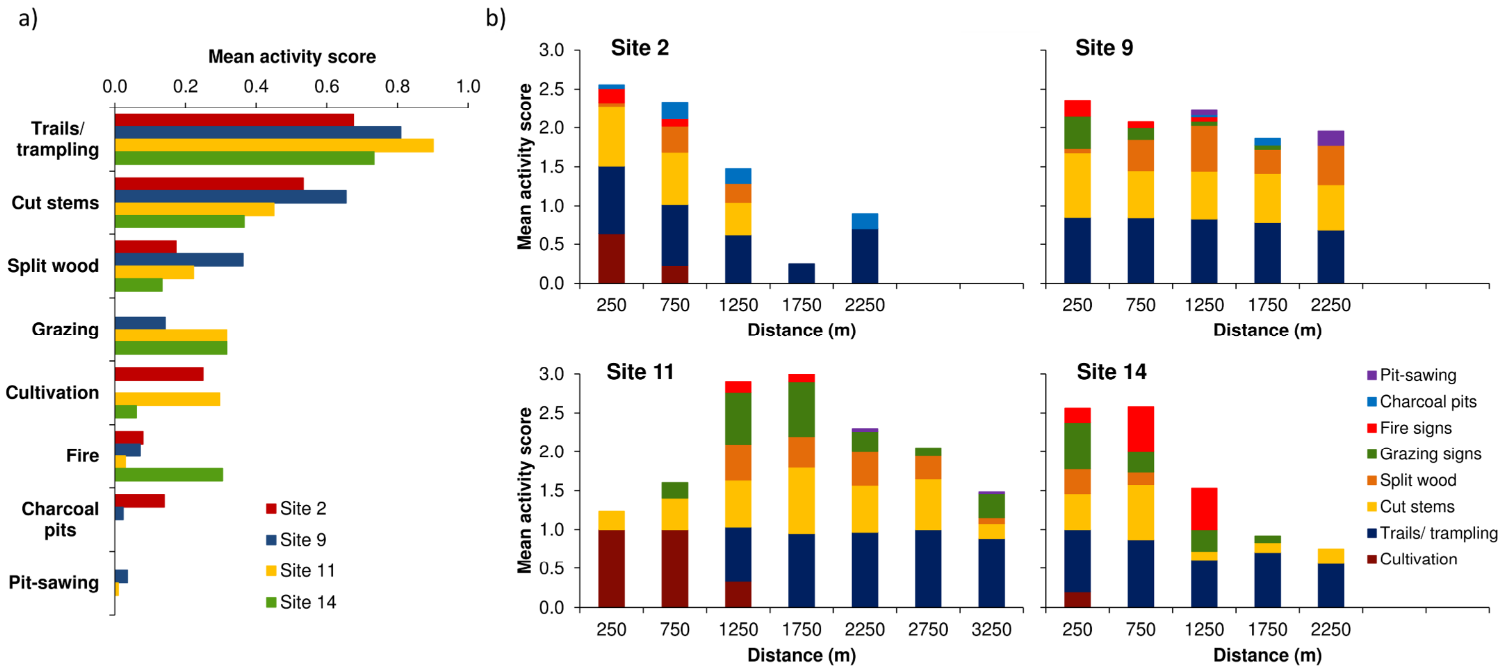

Figure 3.2. Mean scores per site for each activity indicator (a) and mean scores per site at distance class mid-points from the boundary (b). Note the different scale for Site 11 in b) as transects were longer in this site. Trails and trampling signs were only counted in non-cultivated plots. In Site 11 the boundary was straight and all cultivated plots were situated in a 1000-1200 m wide strip where cultivation or pasture dominated all other possible disturbances.

3.3.1. Disturbance

Activity indicators

Figure 3.2 summarises the scores for the human indicators per site (a) and with distance inside the boundary (b).The types and degrees of disturbance varied within and among sites (Figure 3.2.a), and indicators of human activity extended more than 2 km from the boundary into the park in all study sites (Figure 3.2.b). Site 9 scored highest on cut stems and split wood (Figure 3.2.a). In Site 11, all plots in the first 1000 m along the transects were cultivated or grazed (Figure 3.2.b).

Signs of disturbance tended to decrease with distance inside the boundary, with variations per site and activity (Table 3.3). Plots that were cultivated at the time of the study were excluded from the test as agriculture dominated all other impacts. Trails continued into the forest beyond our last plots at all sites. Plots that scored high on trampling were closer to the boundary than plots with no trails or intermediate scores (Table 3.3). Except in Site 11 where wide and well-worn trails were found up to the former settlement and grazing areas at around 3000 m from the boundary (Figure 3.3). Plots nearer the boundary tended to score higher for cut stems (Figure 3.2.b, Table 3.3). Split wood drying in the forest was widely observed in Site 9, but it decreased with distance inside the boundary in the other sites. This decrease was significant in Site 11 and 14 (Table 3.3). In Site 2 and 11, signs of fire occurred mostly on the edge between cultivated areas and forest at around 500 m and 1000 m inside the boundary respectively (Figure 3.2.b), whereas in Site 14, they were found in grassy areas scattered among degraded patches of forest or bush at varying distances up to 1500 m from the boundary (Table 3.3). In Site 2, charcoal-burning was an important activity and (old) pits were observed at almost all distances (Figure 3.1 and 3.2.b, Table 3.3). Pit-sawing tended to occur further away from the boundary, although the relationship was not significant ($p < 0.05$) (Figure 3.1 and 3.2.b and Table 3.3).

Table 3.3. Kruskall-Wallis or Mann-Whitney U tests for the relationship between distance from the boundary and activity indicator scores per plot (currently cultivated areas were omitted).

	Trails and trampling	**Cut stems**	**Wood splitting**	**Grazing**	**Fire**	**Charcoal**	**Pit sawing**
Site 2							
df	2	2	2		1	2	
KW/ U*	14.5	14.7	2.5		15*	0.06	
n	57	57	57		57	57	
p	0.001	0.001	0.286		0.001	0.973	
Site 9							
df	2	2	2	1	2	2	2
KW/ U*	7.5	7.0	4.1	171*	5.7	2.1	4.9
n	84	84	84	84	84	84	84
p	0.024	0.030	0.126	0.001	0.057	0.343	0.085
Site 11							
df	2	2	2	1	1		1
KW/ U*	13.4	16.3	13.0	316*	9*		85*
n	67	67	67	67	67		67
p	0.185	0.000	0.002	0.005	0.039		0.499
Site 14							
df	2	2	1	1	2		
KW/ U*	37.1	14.5	59*	129*	0.7		
n	61	61	61	61	61		
p	0.029	0.001	0.000	0.000	0.688		

Figure 3.3: Main trails in Site 9, 14 and 11. Note the scale in relation to the human figures in Sites 11 and 14. The photo in Site 9 is at a similar scale. The trails in Site 11 and 14 are used by cattle.

Lopping and stumps

Remnant trees in cultivated areas often showed signs of lopping: they had cut-off branches or stems. Overall, lopping decreased with distance inside the boundary (Kendall's tau-b = -0.472, n =49, p < 0.000), and was most common in Sites 9 and 11 (Figure 3.4).

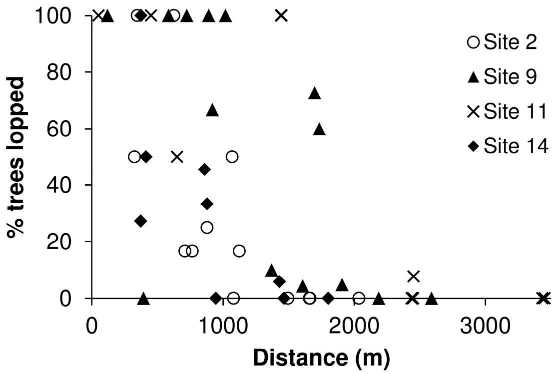

Figure 3.4. Lopping intensity (% trees with cut- off branches or stems per plot) and distance inside the boundary.

No stumps were recorded within 400 m from the boundary (Figure 3.5). In Site 9 we found at least one stump per plot in eight of our plots (40%), all more than 800 m from the boundary. This represented a mean density of 504 stumps per ha. In this site 45% of all stems had been cut, but 63% of the stumps (16) were re-sprouting. Stump density in Site 9 decreased with distance inside the boundary (Figure 3.5), but the data were too sparse for this to be statistically significant. Stems with diameters commonly used as crop-stakes (3 – 15 cm) were cut more often than larger ones: their stumps represented 49% of all recorded stems in Site 9.

In the other sites, we found only one stump in each of 2 plots in Site 2 (12%), in 1 plot in Site 11 (4%), and in 2 plots in Site 14 (11%). This represented a proportion of cut stems of 16 % in all three sites. The cut stems observed in Sites 2 and 11 were more than 15 cm in diameter. In Site 2 stumps were observed mostly in open places in the forest, associated with signs of past pit-sawing or charcoal-burning.

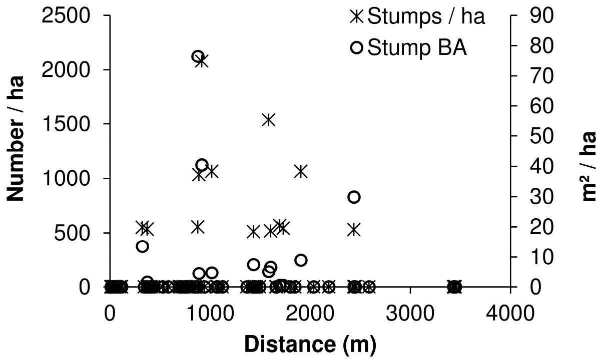

Figure 3.5. Stump density (number.ha^{-1}) and basal area (m^2.ha^{-1}) in Site 9.

3.3.2. Impacts of human activities

Impacts of human activity were assessed for BA, stem density, tree regeneration and species richness.

Basal area

Mean BA in the most intact vegetation type was different among sites (Kruskal-Wallis: KW= 50.5, n = 147, p < 0.001). Pairwise comparisons revealed that BA was significantly larger in Site 11 compared with all other sites (Table 3.4). BA per hectare increased significantly with distance inside the boundary in all sites (linear regression: Site 2: adj R^2 = 0.49, $F_{1,74}$ =71.4; Site 9: adj R^2 = 0.54, $F_{1,82}$ = 99.4; Site 11 adj R^2 = 0.70, $F_{1,99}$ = 127.7; Site 14 adj R^2 = 0.48, $F_{1,80}$ = 75.3; p < 0.001) (Figure 3.6). As can be seen in Figure 3.6, plots in cultivated areas (c4), with relatively few trees, influenced the relationship between distance inside the boundary and BA. Excluding these plots from the analyses weakened the increasing trend slightly, particularly in Site 11 (linear regression: Site 2: adj R^2 = 0.37, $F_{1,55}$ =33.8; Site 9: adj R^2 = 0.54, $F_{1,82}$ = 99.4; Site 11 adj R^2 = 0.19, $F_{1,65}$ = 16.5; Site 14 adj R^2 = 0.43, $F_{1,59}$ = 46.9; p < 0.001). It made no difference in Site 9 where there was no cultivation at the time of the study. BA in glades (Site 11), (natural) tree fall areas or bamboo patches (Site 2 and 9) further inside the park was lower than in other plots at similar distances from the boundary (Figure 3.6).

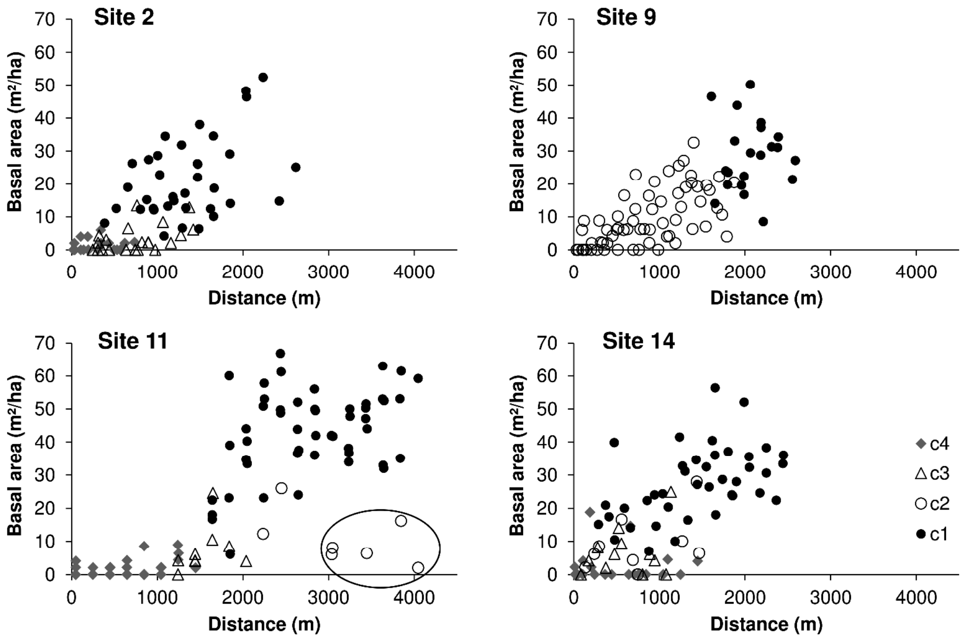

Figure 3.6. Basal area of all plots and distance inside the boundary per study site, coded for each encroachment category as: c1 = never cultivated, c2 = cleared in the 1970s and 1980s now recovering, c3 = cleared in the 1990s and 2000s now recovering, c4 = currently cleared and cultivated (2011). In Site 2, c3 plots nearer the boundary were situated on top of a wide rocky cliff that is kept "open" by fire (to harvest ground honey). In Site 11, the encircled plots were in or on the edge of traditional grazing areas inside the forest. In Site 14, grasslands, regeneration and more intact forest form a patchwork up to 1500 m from the boundary.

The proportion of explained variation increased in all sites when activity indicators were included as independent variables and distance, slope and elevation as covariates (Type II ANCOVA: Site 2: adj R^2 = 0.51, $F_{13,43}$ =5.5; Site 9: adj R^2 = 0.66, $F_{17,66}$ = 40.6; Site 11 adj R^2 = 0.77, $F_{15,54}$ = 16.7; Site 14 adj R^2 = 0.63, $F_{14,60}$ = 10.1; p < 0.001). The encroachment category had significant influence on BA per hectare in all sites (p = 0.004 in Site 2 and p < 0.001 in the other sites) (see also Table 3.4), but the contribution of activity indicators varied per site. Pit-sawing (p = 0.002) and wood splitting (p = 0.004) were significantly related to BA in Site 9. Grazing (p <0.001) contributed to explain BA whereas the influence of distance became insignificant.

In Sites 11 and 14 BA seemed to plateau-out at around 2000 m from the boundary, but in the other two sites effects continued further. Local variation of mean BA was high in Sites 2, 9 and 14, as shown by the standard deviations in Table 3.4.

Table 3.4. Characteristics of the study plots, per encroachment category. Basal area was measured in all plots, other forest structure data only in the detailed-plots. Forest structure data are given as means with standard deviation

	Site 2				Site 9			
	Never cultivated	Cleared 1970s-1980s	Cleared 1990s-2000s	Currently cultivated	Never cultivated	Cleared 1970s-1980s	Cleared 1990s-2000s	Currently cultivated
General characteristics								
Number of plots, detailed in brackets	36 (8)	0 (0)	21 (4)	19 (5)	21 (4)	63 (16)	0 (0)	0 (0)
Maximum distance to boundary (m)	2619.0		1424.1	739.0	2588.7	1873.5		
Mean altitude (masl)	2173 ± 67		2231 ± 42	1995 ± 74	2427 ± 137	2276 ± 77		
Forest structure								
BA (m^2 ha^{-1})	21 ± 12		3 ± 4	1 ± 2	29 ± 11	10 ± 8		
Stems ha^{-1} > 5cm	546 ± 1052		210 ± 244	56 ± 101	790 ± 604	433 ± 486		
Dbh of trees ≥ 5 cm (cm)	57 ± 37		21 ± 17	24 ± 25	62 ± 48	27 ± 23		
Saplings ha^{-1} 2.6-5 cm >1.3 m	212 ± 422		1508 ± 3017	0 ± 0	0 ± 0	164 ± 461		
Saplings ha^{-1} ≤2.5 cm >1.3 m	0 ± 0		2033 ± 2362	0 ± 0	0 ± 0	1465 ± 1870		
Seedlings ha^{-1} ≤ 1.3 m	619 ± 1158		134 ± 267	0 ± 0	901 ± 1802	601 ± 782		
	Site 11				**Site 14**			
	Never cultivated	Cleared 1970s-1980s	Cleared 1990s-2000s	Currently cultivated*	Never cultivated	Cleared 1970s-1980s	Cleared 1990s-2000s	Currently cultivated*
General characteristics								
Number of plots, detailed in brackets	50 (8)	7 (2)	10 (4)	34 (11)	40 (8)	9 (2)	12 (3)	21 (6)
Maximum distance to boundary (m)	4046.0	4046.7	2041.5	1449.1	2448.5	1464.9	1135.8	1447.6
Mean altitude (masl)	2807 ± 36	2829 ± 43	2748 ± 14	2587 ± 70	2556 ± 85	2480 ± 124	2434 ± 133	2438 ± 109
Forest structure								
BA (m^2 ha^{-1})	43 ± 13	11 ± 8	7 ± 7	2 ± 2	28 ± 11	9 ± 9	7 ± 7	2 ± 5
Stems ha^{-1} > 5cm	291 ± 137	79 ± 4	12 ± 5	0 ± 0	490 ± 507	526 ± 664	159 ± 248	1 ± 2
Dbh of trees ≥ 5 cm (cm)	66 ± 31	70 ± 30	83 ± 27	25 ± 71	64 ± 38	52 ± 26	48 ± 34	14 ± 35
Saplings ha^{-1} 2.6-5 cm >1.3 m	0 ± 0	0 ± 0	0 ± 0	0 ± 0	277 ± 515	0 ± 0	349 ± 302	0 ± 0
Saplings ha^{-1} ≤2.5 cm >1.3 m	128 ± 236	0 ± 0	130 ± 260	0 ± 0	1252 ± 1920	255 ± 360	519 ± 513	0 ± 0
Seedlings ha^{-1} ≤ 1.3 m	64 ± 181	0 ± 0	1298 ± 1499	0 ± 0	1674 ± 2375	255 ± 360	0 ± 0	0 ± 0

*Includes pastures kept open through burning and grazing

Tree density

The data for tree density was not normally distributed. We therefore used rank correlations to analyse the relationship between tree density and distance inside the boundary and elevation.

The density of larger trees (≥ 20 cm dbh) increased with distance inside the boundary in all sites (Site 2: Kendall's tau-b = 0.547, n =16, p = 0.004; Site 9: Kendall's tau-b = 0.526, n =20, p = 0.002; Site 11: Kendall's tau-b = 0.596, n =22, p < 0.001; Site 14: Kendall's tau-b = 0.556, n =19, p = 0.001) (Figure 3.7). The density of smaller stems (< 20 cm dbh) increased with distance only in Sites 9 and 14 (Site 9: Kendall's tau-b = 0.445, n =20, p = 0.009; Site 14: Kendall's tau-b = 0.396, n =19, p = 0.031). When cultivated areas (c4) were excluded these relationships were no longer significant. They remain valid for Site 9, which had no c4 plots.

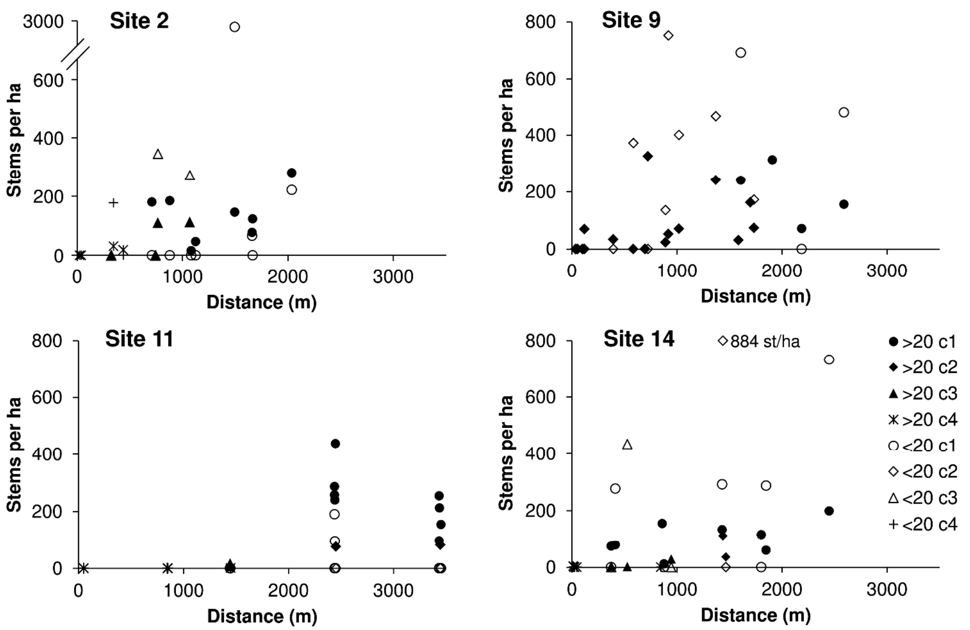

Figure 3.7: Stem densities (stems ≥ 5 cm) and distance inside the boundary per study site (detailed plots). Density of stems < 20 cm and ≥ 20 cm in diameter per plot, coded per encroachment category. The furthest plots in Site 11 were near or in traditional grazing areas.

The density of larger stems was greater in areas that had been abandoned longer ago (Mann-Whitney U or Kruskal-Wallis tests for categories c1 till c3: Site 2: U = 8, n = 12, p = 0.048; Site 9: U = 10.5, n = 20, p = 0.039; Site 11: KW = 10.3, n = 14, p = 0.006; Site 14: KW

= 13, n = 5.9, p = 0.052). In Site 11, we found more large trees in plots with signs of wood splitting than in plots without (Kruskal-Wallis: KW = 7.3, n = 14, p = 0.026). In Site 14, we found fewer large trees in plots with signs of fire than in plots without (Kruskal-Wallis: KW = 4.7, n = 13, p = 0.031). Relationships between tree density and other activity indicators were not significant.

We tested if tree density was correlated with elevation outside of currently cultivated plots and found a negative correlation with large tree density (≥ 20 cm dbh) in Site 2 and a positive correlation in Sites 9 and 11 (Site 2: Kendall's tau-b = -0.443, n =12, p = 0.046; Site 9: Kendall's tau-b = 0.352, n =20, p = 0.035; Site 11: Kendall's tau-b = 0.456, n =14, p = 0.024; Site 14: Kendall's tau-b = 0.128 n =13, p = 0.542).

Regeneration

Densities of saplings and seedlings varied strongly (Table 3.4) and were not significantly correlated with distance inside the boundary. We tested for the effect of shading by larger trees on regeneration but found no significant correlation with BA. There was least regeneration in Sites 9 and 11 (Table 3.4). In Site 9 saplings were recorded only in regenerating areas and 86% originated from coppicing stumps. In Sites 2, 11 and 14, respectively 52%, 33% and 29% of all measured saplings were coppices. The greatest density of seedlings in old-growth forest was found in Site 14 (Table 3.4).

Species richness

We recorded the largest number of species in Site 2, and the least in Site 11 (Figure 3.8). We used two methods for calculating species richness, rarefaction in plots with a minimum of 5 stems and a Z-species richness score corrected for stem numbers that included plots with at least 2 stems. Differences among sites were significant both when using rarefied (5-stem sample) richness (Kruskal-Wallis: KW = 23.1, n = 178, p < 0.001) and Z-scores (all plots ≥ 2 stems) (Kruskal-Wallis: KW = 12.1, n = 243, p = 0.007). Pairwise comparisons revealed greater species richness in Site 2 compared with Site 11 for both methods. Using rarefaction, species richness was also greater in Site 14 compared with Site 11.

<u>Rarefied species richness:</u> Plots with a minimum of 5 trees *de facto* excluded the cultivated areas (c4) in Sites 2 and 11 and retained only 2-3 plots in encroachment categories c2, c3 and c4 in all sites, except in Site 9 (Figure 3.9.a). Rarefied species richness was not significantly correlated with BA or tree density in any of the sites. In Site 2 rarefied species richness was significantly greater in old-growth forest (c1) compared with

regenerating areas (c3) (Kruskal-Wallis: KW = 4.7, n = 34, p = 0.031), but there was no correlation with elevation or distance inside the boundary. In Site 9 the only significant correlation was the increase in species richness with distance inside the boundary (Kendall's tau-b = 0.252, n = 44, p = 0.017) (Figure 3.9). In Sites 11 and 14 rarefied species richness was correlated only with elevation (Site 11: Kendall's tau-b = 0.213, n = 54, p = 0.023; Site 14: Kendall's tau-b = -0.181, n = 60, p = 0.043).

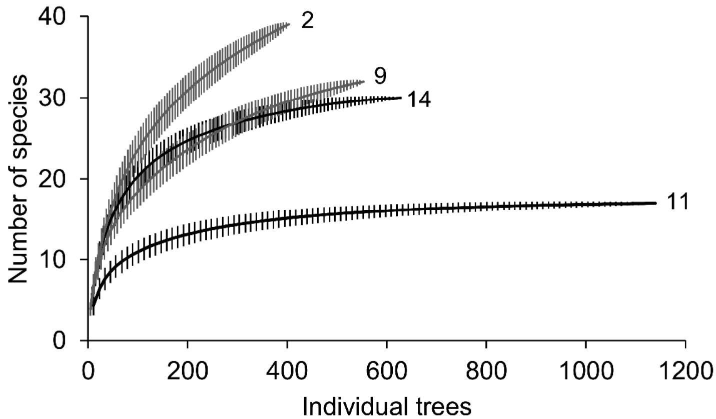

Figure 3.8. Rarefied species accumulation curves per study site (in BiodiversityR, with 100 permutations) with standard deviation.

<u>Z-scores:</u> More plots were included in the calculation of Z-scores as these require a minimum of only 2 plots (Figure 3.9.b). In Sites 9 and 14, Z-scores did not show any significant correlations with BA or tree density. In Sites 2 and 11, Z-species richness was negatively correlated with BA (Site 2: Kendall's tau-b = -0.288, n = 46, p = 0.005; Site 11: Kendall's tau-b = -0.415, n = 71, p < 0.001) and in Site 11 also with tree density (Kendall's tau-b = -0.617, n = 14, p = 0.003). Z-scores for Site 11 were significantly different between encroachment categories (Kruskal-Wallis: KW = 12.2, n = 71, p = 0.007). Pairwise comparisons revealed that Z-scores were smallest in old-growth forest (c1) and largest in regenerating areas (c3). These were located between cultivated land (c4) and old-growth forest (c1) in this site (Figure 3.9.b, Site 11). In Site 14, Z-species richness was significantly different between encroachment categories (Kruskal-Wallis: KW = 8.0, n = 60, p = 0.046), with more richness in intermediate encroachment categories (c3 and c2) than in old-growth (c1) or currently cultivated plots (c4) (Figure 3.9.b), but pairwise comparisons between categories were not significant. In Site 14 the correlations of Z-species richness with distance and with elevation were identical and negative (Kendall's tau-b = -0.180, n =

60, p = 0.043), whereas in the other sites correlations with distance or elevation were not significant.

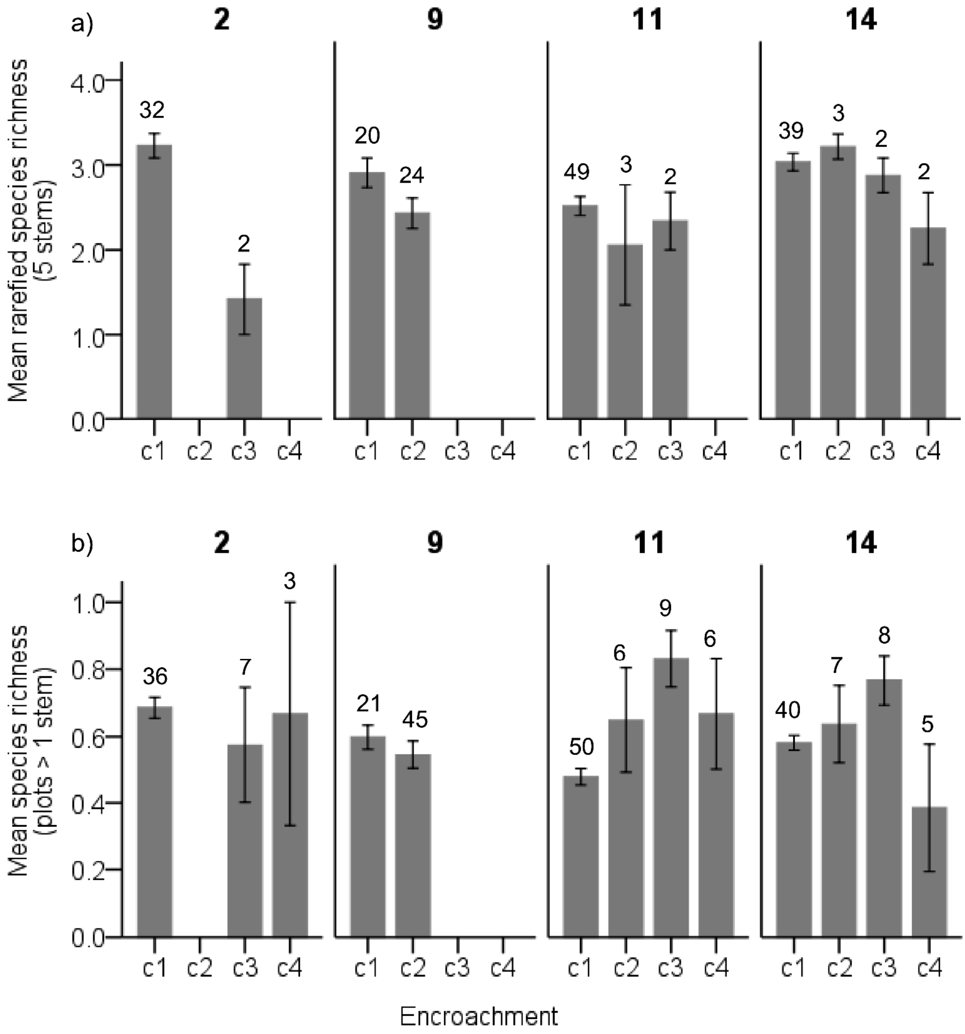

Figure 3.9. Mean species richness (with ± 1 standard error) per plot per encroachment category: c1 = never cultivated, c2 = cleared in the 1970s and 1980s now recovering, c3 = cleared in the 1990s and 2000s now recovering, c4 = currently cleared and cultivated. a) rarefied species richness (samples of 5 stems) and b) species richness in plots with more than 2 stems (Z = log(species count) / log(stem count)). n values are at the top of each bar.

3.4. Discussion

The legacies of human activities observed in this study reflected both past events and on-going processes. These showed marked local variations. Such variations have implications for management. Here we shall first discuss the variation in human activities and their impacts on forest structure. Before addressing site-specific impacts, we first discuss more general patterns and extents of human impacts. We then follow this with an evaluation of the impacts of human activities on species richness. Finally we discuss implications for management and propose a number of options to consider or further investigate that could contribute to forest conservation on Mt Elgon for both local and wider demands.

3.4.1. Activities and impacts on forest structure

Signs of human activities and their impacts remained visible over 2 km inside the park boundary, revealing a broader impact zone than the 900 – 1000 m estimated in Western Uganda by Olupot (2009). Not all signs of activity were equally prevalent in all sites or at all distances from the boundary, presumably reflecting varying local demands and practices, availability of the resource and concern to avoid detection. Some trails remained wide and well-worn far into the park. Such paths are important indicators of human activity, as was also observed on the Kenyan side of Mt Elgon, where the size of local trails was significantly correlated with stem and grass harvesting (Hitimana et al. 2010). The relationship of paths with human activities was also highlighted in a study in Kakamega, Kenya, where evidence of pole-cutting in sites where research and management trails provided access (Fashing et al. 2004).

Past episodes of cultivation within the park still affect forest structure gradients. BA and often stem densities were lower in formerly-cultivated than in old-growth forest areas, even where forest had been “recovering” for 35 years (Sites 2 and 9). In old-growth forest plots from our study we found comparable BA and stem densities to those found in previously selectively logged sites on the Kenyan side of Mt Elgon (Hitimana et al. 2004, Ongugo et al. 2008). Unfortunately there are no studies of recognisably “untouched” forest on Mt Elgon, if such sites exist, to compare with.

Structural diversity and patchiness is characteristic for montane forests in East Africa (Hamilton and Perrott 1981). This means that the high local variations in BA and stem densities that we found in our sites (as expressed in high standard deviations, see Table 3.4) could be natural. It is also plausible that the patterns seen by Hamilton and Perrot (1981) rather reflect extended human presence on most mountain forests rather than any natural pattern (Hamilton et al. 1986). In our sites, the known and inferred history of human presence, the extent and the intensity of the signs of human activity and the observed overall increase in BA (all stems) and stem density (stem size ≥ 20 cm dbh) with distance inside the boundary, suggest that human impacts played a major role.

Hamilton and Perrot (1981) also observed that, overall, tree density (stems ≥ 15 cm dbh) on Mt Elgon decreased up to about 2700 m elevation, which is the elevation spanned by our transects (except in Site 11 where they reach almost 2900 m). We found the opposite or no pattern in our study sites, which was likely related to previous disturbance: in all four sites trees with stem size ≥ 20 cm dbh were negatively affected by the concentration of previous clearing nearer to the park boundary. Trees on the lower rather than the higher slopes were therefore more affected. Logging as in Site 9 and charcoal burring as in Site 2 generally affect larger trees. Any such artificial reduction of large trees is a concern as large trees dominate forest structure and micro-climate, store considerable amounts of carbon and are important as habitat and for forest regeneration (Clark and Clark 1996).

3.4.2. Site-specific impacts

Site-specific histories and resource use patterns can help us to interpret our results. Sites 2, 11 and 14 had areas in different stages of regeneration or degradation more patchily distributed. Therefore forest structure gradients - beyond strips of current cultivation - were less clearly related to distance inside the boundary in Sites 2, 11 and 14 than in Site 9 (Figure 3.6 and 7).

In Site 9, the relationship between BA, tree density and distance inside the boundary (largest R^2, see also Figure 3.6 and 7) reflected a gradual change from bushy regeneration to more advanced regrowth to fairly intact forest away from the boundary. The dominant crops - bananas, coffee and climbing beans - near this site require crop-stakes. The area closest to the boundary in Site 9 appears to be kept in an early succession state due to continuous harvesting of coppice shoots and small trees to meet the demand for small poles. Contrary to previous observations suggesting that local forest use was concentrated in the regenerating areas (Scott 1998), our study revealed that impacts on regeneration and small stems also occurred in old-growth forest.

Resource use agreements include the monitoring of resource off-take by a local resource use committee (UWA 2000). Yet, in Site 9, the incidence of split wood, the signs of cut stems, the numbers of stumps and lopped trees indicated that this was likely not entirely effective. Local informants in this site (Site 9) confirmed that the split fuelwood and crop-stakes for banana that we found drying at further distances in the forest were for commercial purposes (local informants and community conservation ranger, personal communication).

The smallest mean BA and stem sizes in old-growth and regenerating forest were found in Site 2 (Table 3.4). The patchy structural variation of the forest was clearly seen in the high standard deviations of stems densities and BA (see Table 3.4). There were many clearings in the forest, some were likely natural, but charcoal pits were found in 16% of the plots while many others were encountered while moving between the plots (Figure 3.1). Charcoal production removes larger trees than are taken for firewood or for poles (Girard 2002) – this creates larger openings. The local importance of charcoal-burning in this area is consistent with previous findings (Scott 1994a). Conflicts between local people and park management were high in this area of the park (Site 2), which may have encouraged destructive behaviours and illegal activities (Sassen et al. 2013).

In Sites 11 and 14, cattle appeared the main control of regeneration. Cattle are important to the local economy is these sites (Table 3.1) and law enforcement has been unable to prevent grazing inside the park. In Site 11, forest structure data exhibited patterns typical for a grazed forest: open forest with little regeneration and the maintenance of grassy glades (Reed and Clokie 2000). Grazing on either side of the forest edge in combination with tree-cutting for firewood appear the main forces keeping these areas open and eroding the forest edge.

In Site 14, the combination of fire and grazing hindered regeneration in formerly cultivated areas. Fire is also used to help harvest ground honey, an important forest product. However, we found fewer signs of grazing in old-growth forest (although sometimes burn scars) in Site 14 than in Site 11 which could help explain why regeneration seemed better in this site.

Aside from grazing and fire other signs of human use in Sites 11 and 14 were less prevalent than in Sites 2 and 9, possibly because of lower demands due to lower population densities. We note that the crops grown in Sites 11 and 14 (cereals and potatoes) do not require crop-stakes – thus limiting demand for small poles. However, the human population is growing fast (2.5 - 4.3% (UBOS 2002a, b, d)) which may lead to increased demand for wood products such as poles for construction and firewood. People in Sites 11 and 14 had fewer trees planted on their land that could provide alternatives for wood resources from the park (see trees per household in Table 3.1).

3.4.3. Patterns of species richness

Our results suggest human impacts have affected tree species richness on Mt Elgon. We discuss our results in light of existing theories of natural patterns of species richness on mountains. Among our four study sites, and consistent with previous observations (Hamilton and Perrott 1981, Rahbek 1995), overall plot level tree species richness decreased with elevation. The highest mean number of tree species per plot was found in Site 2 and the lowest in Site 11 – the lowest and highest elevation sites respectively. Looking only at plots in old-growth forest Site 11 was also less diverse than the other sites (Figure 3.9).

There are multiple theories concerning natural patterns of species richness on elevational gradients and their determinants (Ghazoul and Sheil 2010). On mountains, area effects have been shown to affect species richness patterns (Romdal and Grytnes 2007, McCain 2009). Many studies indicate a so called “mid-domain effect” (MDE) in which species richness increases and then declines with increasing elevation and that can be seen as a natural consequence of species range patterns and elevation limits (Colwell and Hurtt 1994, Cardelús et al. 2006). This theory has been challenged by empirical research (e.g. Kessler et al. 2011). The relative importance of different factors such as the MDE, climatic and topographic factors likely depends on climatic histories, taxa (Grytnes and Beaman 2006, Acharya et al. 2011) and on spatial scale (Colwell et al. 2004).

As in our case, localised studies may only span a portion of the available elevation gradient. Our plots started above 1900 m and the highest elevation 2877 m was well below the treeline at around 3200 m. At our four sites, we sampled only a section of the forest -range so it is unremarkable that we did not see the humped-shaped elevational gradients of species richness either within the individual study sites or among them. Historical disturbances might in any case strongly influence these theoretical patterns. This indeed seems to be reflected by the complex site-specific patterns at each of the four study sites.

There has been much work on the impacts and influences of disturbance and disturbance regimes on the diversity of tropical forests. For example the intermediate disturbance hypothesis suggests that disturbance of old-growth forest can lead to enrichment through the addition of early successional species, but that excessive disturbance can lead to a decline (Sheil and Burslem 2003).

We discuss how patterns of species richness were affected by human activity in each site and the evidence for the intermediate disturbance hypothesis, which varied among sites.
In Site 2 there were no plots that had been regenerating for more than 20 years (c2). Because only plots with at least 2 or 5 stems were included there were few plots in encroached areas (c3), which made it difficult to reveal clear patterns (Figure 3.9). Z-species richness was negatively correlated with BA, which was strongly affected by intensive charcoal production in this site. Charcoal production led to many large openings at various distances into the park, which explains why we found no patterns with distance inside the boundary or elevation. In Site 9, there were no significant gradients. Selective harvesting may have affected species richness in the formerly encroached (c2) areas. The intermediate disturbance does not cover such selective processes (Site 2 and 9). In Site 9 replanting with a mix of native species will also have affected associated diversity patterns to an unknown degree.

We found evidence for the intermediate disturbance hypothesis In Site 11 and 14. In both sites plots in areas that were recovering or begin degraded (c2 and c3) were richer in species than old growth forest. In Site 11, rarefied species richness was slightly higher at higher elevations. This is because in this Site formerly encroached (c2 and c3) plots with at least 5 stems were found on ridges further inside forest. Z-species richness was greater in areas with less BA, as most old-growth forest with large BA in this site was dominated by few species (*Cornus volkensii, Schefflera, volkensii, Hagenia abyssinica* and *Podocarpus Milianjianius*) (Sassen, unpublished results). A previous study in a site near Site 11 also found that formerly grazed or settled locations had greater species richness than old-growth forest (Reed and Clokie 2000).
In Site 14 species richness decreased with elevation and distance inside the boundary, because there were no formerly encroached plots (c2 and c3) further into the forest as was the case in Site 11. In general terms it appeared that in these two Sites (11 and 14) human impacts, through fire, grazing and wood harvesting at intermediate levels (c2 and c3) often led to some enrichment, as might be predicted from the intermediate disturbance theory (Connell 1978, Sheil and Burslem 2003).

3.4.4. Implications for management
As is the case for many parks in the tropics, even if it were deemed ethical and necessary, it would be impossible to stop people from entering Mt Elgon National Park. Our results highlight various concerns. For instance, even in a site (Site 9) where local forest use is formally regulated and monitored, the intensity of pole harvesting raises concern over the sustainability of such activities. However, further research should investigate whether the

coppicing ability of tree species that are illegally used for poles and crop-stakes (e.g. *Neoboutonia macrocalyx*) could be harnessed to provide a sustainable legal source of such materials. In addition, alternatives need to be found outside the forest. Bamboo-cultivation is currently being promoted in some areas around the park (community conservation ranger, personal communication; personal observation). Shade-trees in coffee can be further promoted as they have shown to benefit coffee production in sub-optimal smallholder systems such as those on Mt Elgon (DaMatta 2004). Charcoal-burning (Site 2) is a commercial activity and unless brought under control will likely lead to continued forest degradation.

Grazing cattle inside the forest is traditionally important to the communities in the north and north-east. A strategy to avoid conflicts could include a system of periodic grazing, e.g. during the dry season when there is less fodder available outside the forest. Or a system where people restrict grazing to the open grassland inside the forest where this has a long history, without hampering regeneration in the surrounding forest (Reed and Clokie 2000). A more in-depth study of the phenology and life history strategies of the plants in these glades may help development of a better understanding of the effects of grazing on the longer term.

Cause and effect of human activities on species richness on Mt Elgon were harder to determine than impacts on forest structure. But in all sites, species richness was affected by past and present disturbance (see also Huang et al. 2003). The effect of fire on the forest community on Mt Elgon needs to be studied in more detail, as some communities on the edges of grasslands may be fire-dependent (van Heist 1994).

Approaches are required that balance conservation and local demands, and that can adapt interventions to local contexts. Mt Elgon has important values for local communities directly neighbouring the park and is also an important water-catchment area for more than a million people in the wider region. Most mountain forests have been able to support or recover from extended human influence over time (Taylor et al. 1999). Therefore, opportunities for creating and maintaining resource use areas in intermediary states of succession - balancing minimum conservation needs while meeting local needs - should be explored (see Hutton and Leader-Williams 2003). Particularly in areas with high population densities with strong claims on forest resources.

Despite some weaknesses, resource use agreements seemed to lead to better outcomes for forest conservation (see also Sassen et al. 2013). Developing capacity for collaborative management and monitoring, sharing responsibilities and rights between park management and local communities can help achieve better conservation outcomes, while taking into account local needs (Vermeulen and Sheil 2007). Developments relating to payments to local communities for avoided deforestation or forest restoration, such as in the context of REDD+ are currently being developed on Mt Elgon as potential win-win solutions for conservation and local development. Practical implementation of such schemes have yet to prove successful in achieving better forest protection and benefits for local people.

Options depend on local needs and preferences and their impacts on conservation (biodiversity, hydrological, carbon) and other values. We can only have informed discussions about these complex trade-offs once we better understand them – and only if we begin to look more carefully at the diversity of local consequences.

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Chapter 4

Fuelwood collection and its impacts on a protected tropical mountain forest in Uganda

Abstract

Local communities who live close to protected forests often depend on them for woodfuel. The extraction of wood for fuel can impact forest structure, function and biodiversity. Our aim was to assess the effects of fuelwood collection on the forest of Mt Elgon National Park (Uganda). Fuelwood collection is legal in areas under collaborative management and illegal, but sometimes tolerated, in areas without such arrangements. We interviewed 192 households about fuelwood use and surveyed dead wood in 81 plots inside the park. Forest was the most important source of fuelwood. People collected on average between 1.1 and 2.0 m^3 of fuelwood per capita per year. Other activities involving wood fuel extraction from the forest include commercial fuelwood harvesting and charcoal making. Quantities of dead wood were affected by fuelwood collection up to at least 1000 m inside the boundary of the park. Depletion of dead wood inside the park was greater in the sites with the highest population density. People who planted more trees on their own land perceived land outside the park to be important and valued old growth forest less as a source of fuelwood. Highly-preferred tree species were most depleted, particularly when they were also valued timber trees, such as *Prunus africana*, *Popocarpus milianjianus*, *Allophylus abyssinicus* and *Olea* spp.. Locally dominant species such as *Hagenia abyssinica*, *Neoboutonia macrocalyx*, *Cornus volkensii* and the seasonal shrub *Vernonia* spp. were less affected. Impacts varied among sites depending on the history of encroachment and locally-specific forest uses, e.g. harvesting of trees for poles or use of the forest land for grazing. Allowing the collection of dead wood is double-edged as it also creates opportunities for other activities that can damage the forest. Demand for fuelwood is likely to grow and our study indicates that the forest may become more degraded as a result. Our results demonstrate that pressure on forests for fuel has negative consequences for both people who depend on the forest and for conservation. Further research into the local ecological and cultural contexts and perceptions concerning losses and benefits may help devise more sustainable management options and successful alternative sources of fuel.

Keywords: Forest conservation, fuelwood, human impacts, tree species, Mt Elgon, Uganda

Sassen, M., Sheil, D. and Giller, K.E. (submitted) Fuelwood collection and its impacts on a protected tropical mountain forest in Uganda

4.1. Introduction

Fuelwood is the main source of energy for cooking and heating in large parts of the world (Parikka 2004, FAO 2010). Most fuelwood comes from bush and fallow lands, but wood extraction for fuel from forests is still important where people have few alternatives (Arnold et al. 2003). Small land-holdings and high population densities increase people's dependence on protected areas for wood (Naughton-Treves et al. 2007, Hartter et al. 2011). The extraction of wood for fuel by collecting dead wood or by harvesting trees or their branches, can impact forest structure, function and biodiversity (Ndangalasi et al. 2007). Woody debris plays an important role in forest ecosystems, in nutrient cycling processes and as habitat for a diversity of fauna, plants, decomposers and other organisms (Duplessis 1995, Christensen et al. 2009). Active harvesting may lead to forest degradation and local forest loss (Geist and Lambin 2002, Arnold et al. 2003). Preferences for certain tree or shrub species may affect species composition. The factors affecting the importance of forests as sources of fuelwood for local communities are poorly understood, and the effect such activities have on forest conservation are uncertain.

The management of tropical forested protected areas needs to consider both the needs of the surrounding population and the impacts of any forest use. New attitudes to forest management call for more devolved approaches to conservation, that allow access and use of forest resources by local communities living in the vicinity of protected areas, in exchange for improved forest protection (Vermeulen and Sheil 2007). But giving people access to protected areas can be double-edged.

In Uganda, pressure on protected forests increases due to a combination of population growth, demands for land and expanding industrial and domestic consumption of wood fuels, including charcoal. Remnant natural forests outside reserves or national parks are rapidly decreasing (Naughton-Treves et al. 2007). The poorest and most vulnerable rural households especially rely on forest resources for energy, food and medicine (McSweeney 2004, Vinceti et al. 2008, Powell et al. 2011, Wan et al. 2011). More than 85% of households in Uganda use fuelwood as the main cooking fuel, 98% if charcoal is included (UBOS 2006). The implications of growing demands for food and energy on forest resources are not well known at local scales where variable contexts may lead to different outcomes. Understanding the importance of forests as sources of fuelwood for local communities and the effects of fuelwood collection on forest conservation values can help design management options that better balance livelihood needs and forest conservation goals.

In this paper we study the patterns and effects of fuelwood extraction on the edges of a humid protected mountain forest in eastern Uganda. We investigate the characteristics and the effects of fuelwood collection and other activities on the availability and distribution of dead wood in four sites of Mt Elgon's forest, in relation to agricultural encroachment, distance from the park boundary, forest structure and local preferences for fuelwood species. We expected that many people would depend on the park for fuelwood, although people with alternative fuel sources less so. We hypothesized that preferred species would be most depleted and this would impact fuelwood use. This study is the third in a series of linked studies that examine these forests and their relationship with local people. In a first paper we described the contexts and drivers that led to local variation in forest loss and recovery over recent decades (Sassen et al. 2013). A second paper examined the nature of the resulting forests under different patterns of local use (Sassen and Sheil 2013).

We first investigated the differences in fuelwood use and the role of the availability of alternatives sources of fuel among contrasting sites situated along the northern and western boundaries of Mt Elgon National Park. We then quantified the volumes of dead wood in 81 plots in the four study sites and their relation with encroachment, distance inside the boundary and forest structure. Species specific impacts of fuelwood collection were also considered. We compared impacts on preferred and used species as reported in 192 household interviews, whereby dead wood from preferred species for fuelwood was expected to be relatively more depleted at greater distances from the boundary compared with that of less preferred or actually used species.

4.2. Study area

Mt Elgon is located on the border between Uganda and Kenya. It is a large extinct volcano (4321 m) with an 8 km wide crater and generally gentle slopes until 2800-3000 m down from the crater-rim. Below this, slopes are steeper the south-west while characteristic sheer cliffs drop down to the plains in the north (Figure 4.1). Annual precipitation falls year round and is between 1500 and 2000 mm but it peaks in April-May and September-November. Rainfall is higher on the southern and western slopes than on the northern and eastern slopes (Dale 1940, IUCN 2005). Mt Elgon is an important water catchment area for several million people in the surrounding districts and for important areas such as the Nile and Victoria basins (IUCN 2005). The mountain is covered with a belt of bamboo and afromontane forest at on average between 2000 and 3000 m, followed by heathers and high altitude moorland (Dale 1940, van Heist 1994). The forests above 2000 m and the higher altitude vegetation host biodiversity characteristic of the Afromontane Region, with a number of species endemic to Mt Elgon (for details see Davenport et al. 1996, IUCN 2005).

Mt Elgon's volcanic soils are fertile and in the south and south-west they support an intensive mixed coffee and banana based agriculture (Kayiso 1993, ILRI 2007). Coffee (*Coffea arabica*) is the main cash crop and is traditionally grown in combination with bananas and multi-purpose shade-trees, both indigenous and exotic species. *Eucalyptus* woodlots are often planted in stream valleys. People have been settled and cultivating these slopes since around 1500 AD. In the north and northeast, agriculture is practiced on larger plots of maize, potatoes, wheat and pasture (ILRI 2007). In this area, people have settled and started practicing agriculture more recently. From the 1980s they were resettled down from the higher slopes of the mountain and from the insecure lower plains to the North. Planting trees is not part of local culture in the north and northeast and trees are therefore rare on farmland, especially nearer the forest edge (Scott 1998).

Uganda's protected forests were widely encroached during the period of political instability that lasted from 1971 until 1986 (Hamilton 1985, Turyahabwe and Banana 2008). Since 1987, forest restoration activities were started in the worst affected areas on the western slopes (UWA 2000), with mixed success. In later years new forest clearing took place in different areas of the park (Sassen et al. 2013). When Mt Elgon was gazetted a national park in 1993, local communities lost all legal access rights (Scott 1998). Since the late 1990s park management has initiated agreements with local communities living next to the park (at parish level) that allow regulated collection of a limited number of non-timber products, fallen dead wood for fuel and the stems from a limited number of shrub species (e.g. *Vernonia* spp.) to support crops like bananas and climbing beans (Scott 1998, UWA 2000). Illegal activities include cattle grazing, tree-cutting, charcoal burning and hunting. Whether or not a community living next to the park has entered into such an agreement depends strongly on the level of conflict with UWA about park boundaries and access for cattle grazing (Sassen et al. 2013). In areas without agreements some uses, including dead wood collection, are sometimes tolerated on an *ad-hoc* basis in an effort by local rangers to minimize conflicts. Dependence on forest products remains important and illegal resource extraction common (Scott 1994a, Norgrove 2002, Katto 2004). This is unlikely to decrease in the near future as population densities continue to grow and increase local demands for wood. No natural forests remain within 20 km around the protected area (Sassen et al. 2013). In 2002, human population densities in the parishes surrounding Mt Elgon ranged from 150 p/km^2 in the north to more than 1000 p/km^2 in the west. Average annual population growth rates ranged between 2.5% and 4.3% (UBOS 2002a, b, d).

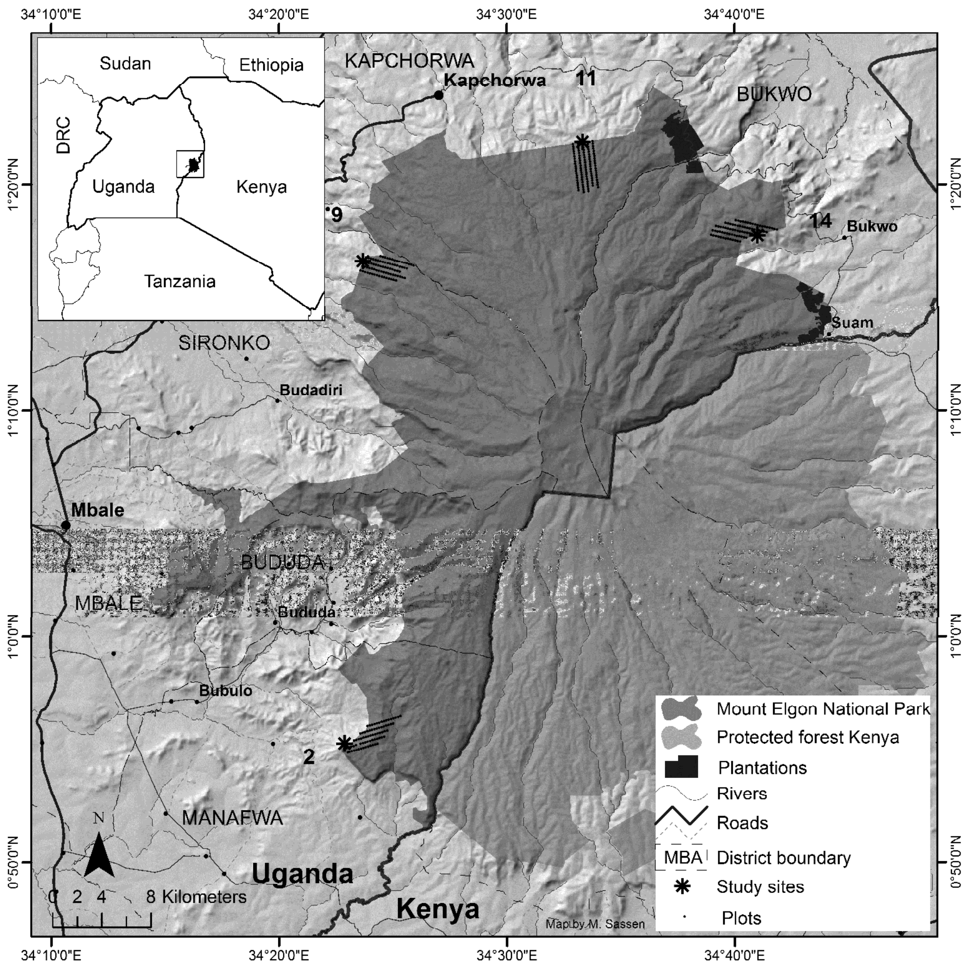


Figure 4.1. Map of Mt Elgon with the study sites (administrative division boundaries valid in 2010).

4.3. Methods

4.3.1. Field data

We collected data in four sites on the edge of Mt Elgon National Park, to represent different elevations, forest types and forest cover change histories. Forest cover change on Mt Elgon is strongly related to its history of agricultural encroachment and regrowth renewed clearing and recovery took place at different times (Sassen et al. 2013). In Site 9 the forest was not re-encroached after people were evicted from the park in the early

1990s, whereas in Site 2 most of the regenerating vegetation and restoration planting was cleared again from 2006. In Site 11 and forest clearing for agriculture started in the 1990s, and was intensified by a conflict about the boundary of the area excised for resettlement. In Site 14, encroachment started in the 1980s but it took place in patches rather than as a 'front' from the boundary inwards. Each site corresponded to a sample village (Sassen and Sheil 2013).

In each site, we first laid out one transect approximately perpendicular to the general orientation of the boundary, and then two parallel transects 400 m apart on each side (Figure 4.1). We measured dead wood volumes on plots 50 m, 850 m, 1850 m, 2850 m and when possible 3850 m along the transects (Figure 4.2). Due to irregular shape of the boundary, this translated to different actual distances from the boundary. Actual distance to the boundary for each plot was calculated using a GIS. We used a handheld GPS (Garmin 60CSx) to determine plot position and orientation along the transect line. On each transect line we measured between 3 and 4 plots (15-21 per site, 81 in total), depending on travel time between plots which was influenced by terrain (e.g. obstacles) and vegetation.

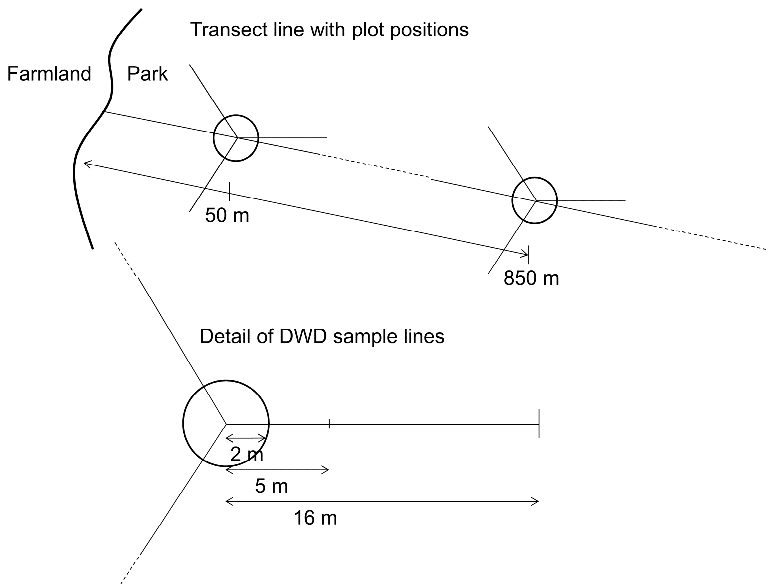


Figure 4.2. Transect line with plot positions and detail of DWD plot. The centre of each plot is also the point from which the relascope sweeps were held.

We adapted methods for assessing woody debris from Harmon and Sexton (1996). We established three 16 m sampling lines from the centre of each plot to record dead wood using a line intercept method (diameter at intercept, decay class for larger pieces, count for smaller ones). The first line was oriented East and the other two at 135° and 225° anticlockwise (Figure 4.2). We recorded woody debris lying or hanging <2 m from the ground. For each coarse woody debris (≥5.1 cm diam.) crossing the sampling lines we measured the diameter at line intercept and recorded the decay class as follows: class 1: solid wood, recently fallen, with bark still intact, cannot push a nail into the wood by hand; class 2: solid wood with > 50% bark still intact, can push a nail into the wood by hand to a maximum of 0.5 cm; class 3: less-solid wood, especially the outer layer, but with deeper layers still hard, bark <50% intact, a nail can be pushed into the wood by hand more than 0.5 cm; rotten: soft, rotten wood, no bark, a metal nail can be pushed into the wood easily or it collapses when stepped on. We tallied smaller pieces (1-5 cm) on sampling line segments of 5 m long (starting from the centre). Only non-rotten pieces of wood were counted.

In an assessment of forest structure we recorded the basal area of standing live and dead trees (taller than 1.30 m at dbh) in 343 plots using angle count sampling, and measured stem density in 81 plots (Sassen and Sheil 2013). The same plot-centres were used for the assessment of standing trees and dead wood. In each plot we recorded terrain, vegetation cover, signs of disturbance and history of encroachment. With the help of local informants, we classified each plot into one of the following categories: c1 = not cultivated within living memory (also called 'old-growth forest'), c2 = cleared in the 1970s and 1980s but now recovering, c3 = cleared in the 1990s and 2000s but now recovering, c4 = currently cleared and cultivated or grazed (2011).

Species were identified by local informants and two knowledgeable rangers (one of Bagisu and one of Sabiny ethnic background). We cross-referenced the names of standing trees with available references (Hamilton 1991, Katende et al. 2000). Photographs of unknown species were taken to the Institute for Tropical Forest Conservation (ITFC), Uganda, for identification. Surrounding trees aided identification of woody debris that were harder to recognize, but decay hindered identification.

4.3.2. Data on firewood collection and use

In each site, we conducted semi-structured interviews with households selected within the sample village and the neighbouring village that most corresponded to the boundary section intersected by the transects (Table 4.1). The households were selected by randomly drawing names from a list established with the help of village leaders. Respondents provided information on standard household characteristics, land ownership, frequency, quantities and location of fuelwood collection, preferred and used species, perceived changes in the availability of fuelwood and numbers and species of trees and large shrubs on their own land. We considered different sources of fuelwood: "Old forest" was defined as forest that was never cleared for cultivation within living memory but still accessed for other uses such as fuelwood, crop stakes, medicine and vegetables and to reach bamboo or grazing areas deeper into the forest. "Formerly encroached forest" included areas that had been cleared at some point in the past (from the 1970s onwards) and that were in various stages of recovery at the time of this study. "Own land" was defined as land owned, rented or otherwise occupied by people. In sites with on-going encroachment, this sometimes included land inside the official boundary of the park. Fuelwood from the market was usually purchased within the parish or neighbouring parishes, collected from either the forest or from planted trees (*Eucalyptus* spp.). The Uganda Wildlife Authority defines a back- or headload as a bundle that people can carry on their backs or heads in one haul. We use headload as a standard term from here-on. We asked people to estimate the number of headloads they collected per week and per source area. We measured 22 loads of fuelwood carried by people coming out of the forest in Site 9 on resource collection days. In the other sites this was more difficult because people did not have a resource use agreement with UWA and therefore fuelwood collection was formally prohibited.

4.3.3. Data analysis

We used a conservative approach to estimate the volumes of fuelwood collected per household per year. We did not include fuelwood reportedly collected from people's own land because we observed that people tended to collect pieces or bundles on a more *ad hoc* basis from there. Reported loads of fuelwood from the forest and from markets were more likely to be consistent in size with the headloads we measured. We also did not include fuelwood bought from markets to avoid double counting with wood that people collected from the forest and then sold. It was not possible to get figures for quantities sold. People were more willing to say that they bought fuelwood than that they collected fuelwood for sale, as this was illegal. For the conversion of the volume of a bundle of fuelwood to a solid volume measure, we used a conservative average conversion measure of 0.37 (FAO 1983).

The relative score for preferred or used species consisted of the sum of the scores (inverse of rank) that a species received from each respondent divided by the total score for all species in that site. We compared the lists of the five most preferred and the five most used species in each site, on the premise that discrepancies between the lists indicate depletion due to overharvesting or difficulties of access.

We calculated the volumes of dead wood following Harmon and Sexton (1996), and the volume of standing dead trees using dbh and height and a form factor of 0.5. We explored the correlations between volumes of dead wood and encroachment, distance into the park and measures of forest structure. We also compared the volumes of dead wood found for preferred and used species. We calculated the relative basal area of live trees of preferred and used species to assess differences in the impacts of harvesting (for fuelwood but also other timber uses) on preferred species and on the species people reportedly actually used. Data analysis was carried-out using SPSS version 18.0 (SPSS Inc., Chicago IL).

Table 4.1. Characteristics of the study sites.

	Site 2	**Site 9**	**Site 11**	**Site 14**
Village (2011)	Bukuwa	Kinyofu/ Gibuzale	Korto/ Kamatelon	Sindet/ Kapsata
Sub-county (2011)	Bupoto	Masira	Kwosir	Kortek
Population density 2002[a]	631 p km^{-1}	712 p km^{-1}	448 p km^{-1}	374 p km^{-1}
Mean household size	4.7	5.2	5.6	5.8
HH interviewed (% of total)	53 (77%)	45 (45%)	51 (63%)	43 (66%)
Perennial crop (%[b])	Coffee (28%) banana (57%)	Coffee (69%) banana (60%)	Only seasonal	Only seasonal
Resource use agreement	no	yes	no	no
Plots	17	20	25	19
Elevation plots in masl	1911-2318	2152-2606	2478-2877	2238-2699

[a] (UBOS 2002b, d, a)
[b] % of interviewed people listing the crop as either their first or second crop (so total % > 100)

4.4. Results

4.4.1. Fuelwood collection and use

People generally collected fuelwood once or twice a week (all sources combined - Figure 4.3). The park - old growth forest and regenerating areas combined – supplied the largest quantities, followed by markets (Table 4.2). In Site 14 none of the interviewed households reported collecting any fuelwood from regenerating areas, although two of them had first ranked these areas second as a source of fuelwood.

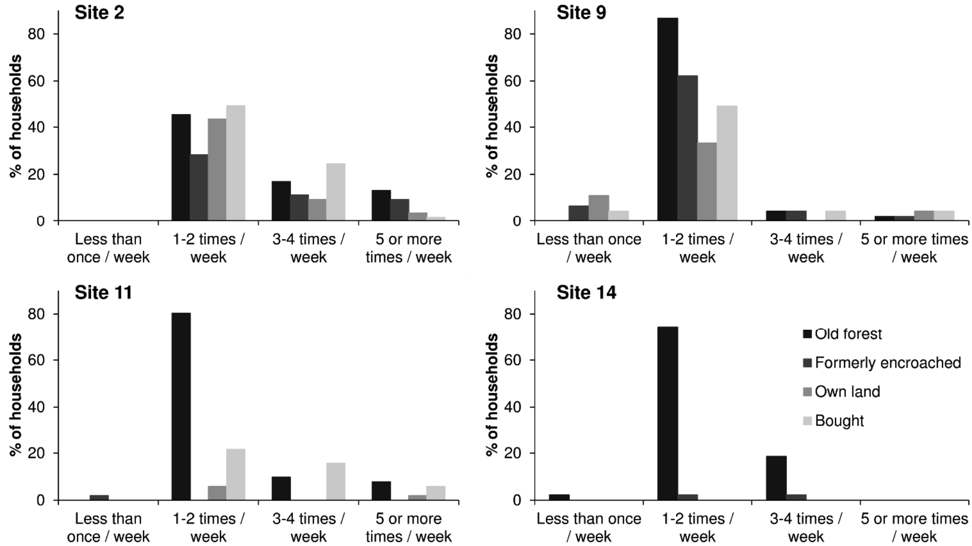


Figure 4.3. Weekly frequency at which people collect fuelwood from different sources, expressed as the percentage of households per site that reported that frequency for that source.

Table 4.2. Average estimated number of head-loads of fuelwood used per household per week from different sources, and use.

	Source				Volume		Use (% HH)			
Site	Old forest	Form. encr.*	Own land	Market	Per HH $m^3.y^{-1}$	Per capita $m^3.y^{-1}$	Cook	Heat	Brew	Sale
2	2.2	1.4	0.9	1.6	9.5 (±9.5)	2.0	100	0	4	13
9	1.8	1.4	0.6	1.0	8.4 (±3.7)	1.6	100	0	11	7
11	2.4	0.0	0.1	1.6	6.3 (±4.4)	1.1	100	2	6	6
14	3.2	0.0	0.0	0.0	8.3 (±4.9)	1.4	100	0	0	5

*Formerly encroached land inside the park

The average volume of a measured headload of fuelwood was 0.14 m^3 (N = 25), which translated into an average solid volume of 0.05 m^3 (± 0.003, 96% confidence interval) per headload. Taking household sizes into consideration implies that people collected on average between 1.1 and 2.0 m^3 of solid fuelwood per capita per year (Table 4.2). The number of bundles that people reportedly collected per week was not significantly correlated with the size of the household (Kendalls tau-b = -0.12, p = 0.825, n = 192) or the area of land owned (Kendalls tau-b = 0.19, p = 0.754, n = 177). Larger households tended to own more land and households in the Sabiny-dominated sites (Sites 11 and 14) were larger than those in the Bagisu-dominated sites (Sites 2 and 9), but the area of land owned did not differ between tribes (data not presented). In all households the primary use of fuelwood was cooking (Table 4.2). Some sale of fuelwood was reported in all sites.

4.4.2. Sources of fuelwood

In all four sites local respondents considered forest – in particular old growth forest –the most important source of fuelwood (Table 4.3). In Site 2, 85% of the respondents also reported buying fuelwood, at least occasionally. In Site 9, 69% and in Site 11, 43% of people sometimes bought fuelwood, whereas all fuelwood in Site 14 reportedly came from the forest. In Site 2 and 9, respectively 60% and 51% of the respondents collected fuelwood from their own land (Table 4.3).

Table 4.3. Ranks given to different sources of fuelwood per site (% households).

	Old forest		**Form. encr.***		**Own land**			**Market**			
Site	**1st**	**2nd**	**1st**	**2nd**	**1st**	**2nd**	**3rd**	**1st**	**2nd**	**3rd**	**4th**
2	74	2	4	45	13	21	26	9	13	38	25
9	80	16	18	51	2	16	33	2	11	29	22
11	98	0	0	0	2	8	0	0	43	8	0
14	93	0	5	0	0	0	0	0	0	0	0

*Formerly encroached land inside the park

The 192 households we interviewed reported a total 51 (39 native) species of trees and shrubs on their own land. The mean number of stems per household was least in Site 11 (less than 3 stems/HH) and highest is Site 2 (more than 33 stems/HH or 31 stems/HH when not counting tree-like shrubs such as *Ricinus communis* and *Vernonia spp*.) (Table 4.4). Households in Site 2 and 9 reported a greater variety of species and more trees on average than in Sites 11 and 14. Exotic species were the most common species at all sites, but were especially dominant at Sites 11 and 14 (Table 4.4).

Overall, the importance – in rank, frequency and for quantities – that households gave their own land as a source of fuelwood was positively correlated with the number of trees they had on their own land and the amount of land they owned. The correlation results for the number of trees were respectively for rank, frequency and quantities: Kendall's tau-b = 0.465, 0.415 and 0.444, $n = 192$, $p < 0.001$. The correlations for the area of land were respectively for rank, frequency and quantities: Kendall's tau-b = 0.289, 0.247 and 0.248 $n = 177$, $p < 0.001$. The correlations between the density of trees on people's land and the importance of various sources of fuelwood were similar (Appendix 4.A). There were variations within sites which are reported in Appendix 4.A.

Table 4.4. Mean number of species per site, percentage of households (HH) with trees on their land (minimum 1 stem), mean number of trees per household (bold in table) and maximum number of stems listed by one household. For the five most frequently reported species per household: percentage of households listing the species, mean number of stems and range. All can be used for fuelwood.

	Species	Proportion of households (%)	Mean number stems per HH	Maximum number of stems
Site 2	**All (3.98 ± 2.54 species)**	**87**	**33.25**	**178**
n=53	*Eucalyptus* sp.	51	16.98	100
	Markhamia platycalyx	16	5.21	50
	Cordia africana	6	1.83	12
	Persea americana	5	1.74	18
	Vernonia auriculifera	4	1.21	40
Site 9	**All (4.51 ± 2.46 species)**	**93**	**27.29**	**155**
n=45	*Eucalyptus* sp.	47	13.11	100
	Persea americana	15	4.07	20
	Markhamia platycalyx	8	2.22	30
	Eriobotrya japonica	6	1.78	10
	Ehretia cymosa	4	1.13	20
Site 11	**All (0.65 ± 0.90 species)**	**43**	**2.96**	**40**
n=51	*Eucalyptus* sp.	53	1.57	40
	Allophylus abyssinicus	17	0.49	10
	Cornus volkensii	10	0.29	10
	Grevillea robusta	6	0.18	6
	Dombeya goetzenii	4	0.12	4
Site 14[a]	**All (0.56 ± 0.63 species)**	**49**	**8.16**	**70**
n=46	*Eucalyptus* sp.	95	7.79	70
	Grevillea robusta	3	0.21	5
	Ekebergia capensis	1	0.09	4
	Persea americana	1	0.07	3

[a] In Site 14 only four species were reported

Combining all sites, the importance – in terms of frequency and for quantity – of old growth forest was significantly negatively correlated with the number of trees people had on their own land, although the correlation was weak. The correlation results were respectively for rank, frequency and quantities: Kendall's tau-b = -0.095 with p =0.120, -0.125 with p =0.035 and -0.135 with p =0.017, n = 192 (details per site in Appendix 4.A). The importance of formerly encroached forest was also positively correlated with the number of trees that households report they had on their own land although the correlation was less strong than for "own land" (respectively for rank, frequency and quantities: Kendall's tau-b = 0.240, 0.295 and 0.286, n = 192, p < 0.001 for all). Households with more land had more trees on that land (Kendall's tau-b = 0.233, n = 177, p < 0.001), although not always in terms of density (Kendall's tau-b = -0.050, n = 169, p = 0.400).

4.4.3. Woody debris

Quantities of woody debris

In all sites large woody debris (LWD) (> 5.1 cm) made up most of the woody debris volume (Figure 4.4). Mean volumes per hectare of dead wood were smallest in Site 11. More standing dead trees were found in Sites 11 and 14 compared with Sites 2 and 9 (≤ 2) (Figure 4.4).

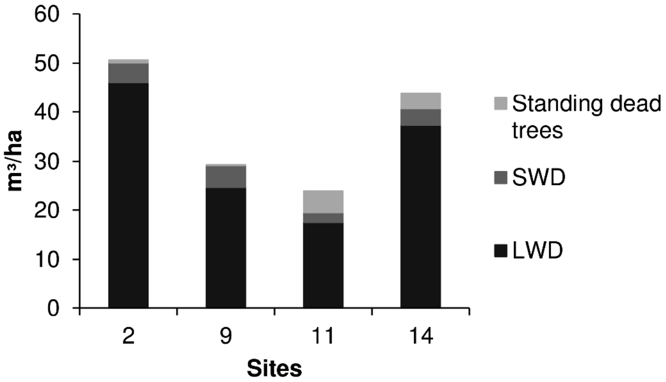


Figure 4.4. Mean volume of dead wood per ha per site for small woody debris (SWD), large woody debris (LWD) and standing dead trees (> 1.30 m). Stumps (LWD) and standing dead trees were measured in fixed area plots.

The largest volumes of dead wood occurred in old-growth forest, followed by long-recovered areas and the smallest volumes in the most recently-encroached lands (Figure 4.5). Woody debris that originated from human activities varied between 1 and 18% of all recorded LWD. The highest proportion of debris that showed signs of manual cutting was found in Site 9 (Table 4.5).

Volumes of dead wood per plot generally increased with greater distance into the park (Figure 4.6). Total dead wood volume per plot was positively correlated with distance inside the park boundary, live tree basal area and tree density in all sites (Kendall's tau-b = 0.449, 0.520 and 0.425 respectively for distance, BA and stem density, n = 81, p < 0.001 for all sites combined). Overall, the volume of woody debris was neither correlated with slope (Kendalls tau-b = 0.009, p = 0.913, n = 80) nor elevation (Kendalls tau-b = 0.103, p = 0.184, n = 81) although there were variations among sites (details in Appendix 4.B).

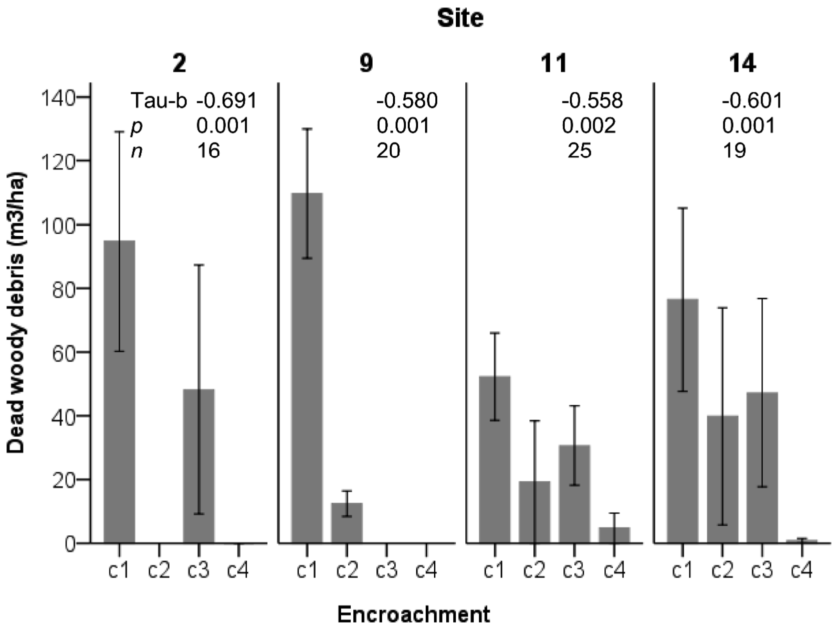


Figure 4.5. Mean total dead wood volume (± 1 standard error) per site and per encroachment category: c1 = never cultivated, c2 = cleared in the 1970s and 1980s now recovering, c3 = cleared in the 1990s and 2000s now recovering, c4 = currently cleared and cultivated (2011). Kendall's tau-b correlation results are given for each site.

Table 4.5. Percentages of rotten wood and large woody debris (LWD) that was considered suitable for fuelwood per decay class and proportion cut by people.

	Mean volume per decay class ($m^3.ha^{-1}$) and %				LWD suitable as fuelwood per decay class (% of volume found)			LWD cut by people (% of pieces found)
Site	**1**	**2**	**3**	**Rotten**	**All**	**1 and 2**	**3**	
2	22 (48%)	4 (8%)	10 (21%)	10 (22%)	64[a]	96	48	1
9	6 (25%)	7 (30%)	9 (34%)	3 (11%)	87	98	98	18
11	4 (23%)	4 (24%)	6 (33%)	3 (20%)	54	91	33	12
14	15 (40%)	10 (26%)	8 (22%)	4 (11%)	87	99	96	5

[a] In Site 14 class 2 included some burnt LWD

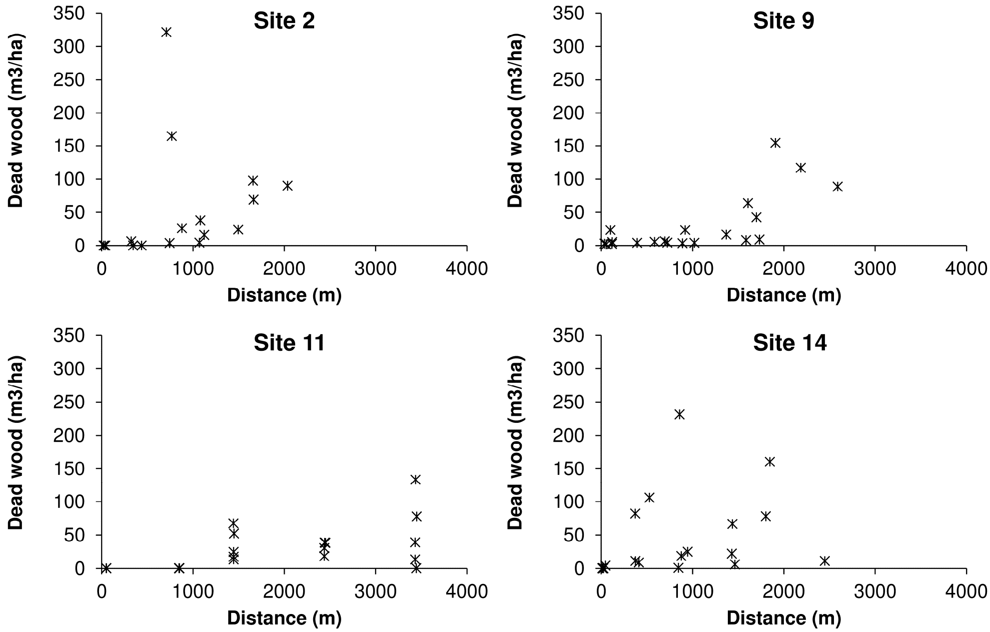


Figure 4.6. Volume of dead wood per ha and distance from the boundary in the four study sites. Note: In Site 3 two large fallen *Aningeria* spp. in resulted in high values in two plots at around 800 m from the boundary.

Suitability of woody debris for fuel

Around 25% of woody debris in Sites 9 and 11 showed no signs of decay. In Sites 2 and 14 this was 40% or more (Table 4.5). The proportion of dead wood suitable to use as fuelwood according to local informants in the field varied between 54 and 87%, with the rest being too rotten (Table 4.5). We found the smallest proportion of rotten wood in Sites 9 and 14, where people were also the least selective in terms of fuelwood quality: almost all pieces of wood in decay class 3 were still considered usable as fuelwood (Table 4.5). In Site 11 only one third of wood in decay class 3 was considered suitable as fuelwood.

4.4.3. Impacts of fuelwood collection on preferred species

Preferred and used species

The number of different species listed in the 'top 5 preferred species' by the different households ranged from 39 in Site 2 (N=53) to 15 in Site 14 (N=43). Sites 9 (26 species; N=45) and 11 (17 species; N=51) were intermediate.

Certain species were consistently given high ranks by most households in a site, such as *Prunus africana* (listed by 64%, 89%, 84% and 79% of all households in Sites 2, 9, 11 and 14 respectively), *Cornus volkensii* and *Olea chrysophylla* (both listed by 76 % of the households in Site 11) and *Allophylus abyssinicus* (listed by 69% and 93% of households in Sites 11 and 14 respectively). In Sites 2 and 9, respectively, at least 57% and 67% of the households listed each of the overall top two preferred species while the other three species were listed by between 32-44% of the households. In Sites 11 and 14 the species in the top five preferred species were listed by at least 55% and the top 3 by at least 76% of the households. Not all listed species were forest species (Table 4.6). For example in Site 2, *Eucalyptus* sp. was listed among the top five of preferred species by 36% of the households (detail in Appendix 4.D). In Sites 11 and 14 the list of the five most preferred and the five most used species had more names in common than in Sites 2 and 9 (Table 4.6).

Table 4.6. Relative scores of preferred and used species from household interviews and relative volumes of dead wood for the species found in the forest survey (unidentified species omitted, see notes).

Site	Preferred species	Score (%)	% of total BA	Used species	Score (%)	% of total BA
2	*Prunus africana*	16.6	1.5	*Eucalyptus* sp.	20.2	
	Aningeria spp.	13.3	6.1	*Vernonia auriculifera*	11.2	
	Eucalyptus sp.	9.5	0.0	*Markhamia platycalyx*	10.6	0.2
	Croton spp.	8.1	0.2	*Cordia africana*	9.3	0.0
	Vernonia auriculifera	7.3		Maize stems/cobs	6.3	
9	*Prunus africana*	21.5	1.5	*Vernonia auriculifera*	27.9	
	Podocarpus milianjianus	16.7	1.4	*Hagenia abyssinica*	12.6	20.5
	Allophylus abyssinicus	9.3	4.0	*Neoboutonia macrocalyx*	12.3	27.1
	Hagenia abyssinica	8.3	20.5	*Maesa lanceolata*	11.6	4.8
	Olea welwitschii	6.8	0.5	*Mimulopsis arborea*	10.6	
11	*Cornus volkensii*	18.2	59.7	*Cornus volkensii*	17.7	59.7
	Olea chrysophylla	16.8	0.9	*Olea chrysophylla*	15.3	0.9
	Prunus africana	16.0	5.2	*Allophylus abyssinicus*	15.0	0.9
	Allophylus abyssinicus	15.4	0.9	*Prunus africana*	15.0	5.2
	Podocarpus milianjianus	11.4	10.1	*Podocarpus milianjianus*	10.5	10.1
14	*Prunus africana*	18.5	2.8	*Vernonia* spp.	28.3	
	Allophylus abyssinicus	17.8	26.1	*Solanum* sp.	22.7	
	Vernonia sp.	13.3		*Prunus africana*	10.9	2.8
	Croton spp.	12.9	2.7	*Allophylus abyssinicus*	9.5	26.1
	Ekebergia capensis	11.4	6.9	*Croton* spp.	6.5	2.7

Table 4.6. (continued)

Dead wood found	% Vol/ha	% of total BA	Trees found	% of total BA
Aningeria spp.	60	2.9	*Neoboutonia macrocalyx*	27.53
Neoboutonia macrocalyx	16	27.5	*Macaranga kilimandscharica*	17.37
Syzygium guineense	8	4.1	*Tabernaemontana holstii*	12.80
Macaranga kilimandscharica	6	17.4	*Syzygium guineense*	4.11
Mimulopsis arborea	5[1]		*Strombosia schefflerii*	3.31
Syzygium guineense	27[2]	6.6	*Neoboutonia macrocalyx*	26.85
Prunus africana	18	1.5	*Hagenia abyssinica*	20.31
Podocarpus milianjianus	17	1.4	*Macaranga kilimandscharica*	8.92
Neoboutonia macrocalyx	15	26.9	*Syzygium guineense*	6.64
Hagenia abyssinica	10	20.3	*Schefflera volkensii*	5.24
Cornus volkensii	44	59.7	*Cornus volkensii*	59.69
Allophylus abyssinicus	14[3]	0.9	*Podocarpus milianjianus*	10.12
Prunus africana	13	5.2	*Schefflera volkensii*	7.61
Dombeya goetzenii	10	1.6	*Hagenia abyssinica*	6.01
Rapanea melanoploeos	7	3.2	*Prunus africana*	5.16
Allophylus abyssinicus	37	26.1	*Allophylus abyssinicus*	26.05
Olea welwitschii	25	0.8	*Neoboutonia macrocalyx*	19.45
Neoboutonia macrocalyx	16	19.4	*Hagenia abyssinica*	7.45
Croton spp.	9[4]	2.8	*Ekebergia capensis*	6.94
Rapanea melanophloeos	4	2.0	*Schefflera volkensii*	6.46

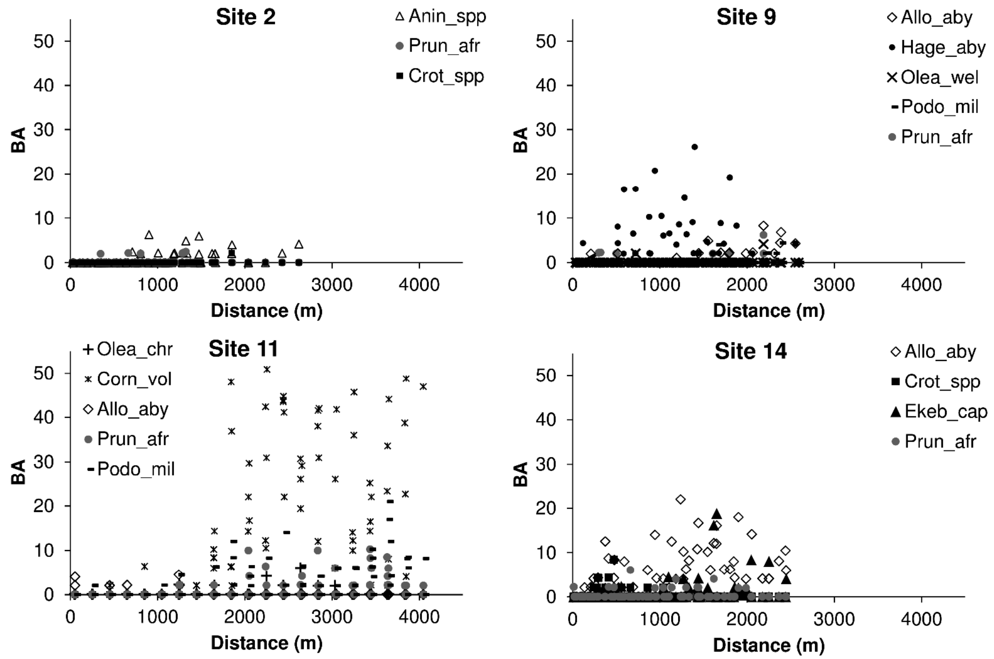
[1] Unidentified species made up 5% of the volume/ha
[2] Unidentified species made up 43% of the volume/ha, but 85% of that was one unidentified log.
[3] Unidentified species made up 19% of the volume/ha
[4] Unidentified species made up 6% of the volume/ha

Basal area and dead wood of preferred and used species

The basal area of the five most highly preferred and used forest species decreased with distance from inside the park towards the boundary (Figure 4.7). Certain species were almost completely depleted within the distance range of our study, e.g. *P. africana* and *Croton* spp in Site 2. Many highly preferred forest species that had small actual (Figure 4.7) and relative (Table 4.6) basal area were not in the top five of most used species in that site, except for example *O. chrysophylla* and *A. abyssinicus* in Site 11. The basal area of locally dominant species that were also highly preferred or used - such as *H. abyssinica, Neoboutonia macrocalyx* (Site 9), *C. volkensii* (Site 11) and *A. abyssinicus* (Site 14) - was generally relatively high, although they were all depleted near the boundary (Figure 4.7). Less dominant species had relatively low basal areas, including some preferred species such as *A. abyssinicus* and *O. chrysophylla* in Site 11 and *Croton spp*. in Site 14, but also others such as *Markhamia platycalyx* in Site 2 and *Olea welwitschii* in Site 9 (Table 4.6, Figure 4.7a).

Woody debris remained unidentified in 5%, 8%, 9% and 4% of the cases in Sites 2, 9, 11 and 14 respectively (see volume equivalents under Table 4.6). The tree species for which we found the largest quantities of dead wood were similar to those for which we found the highest relative basal areas in most sites (Table 4.6). Dead wood was most abundant for the dominant species in their respective sites (Table 4.6) and completely absent for others, such as for the preferred forest species *P. africana* and *Croton* spp. in Site 2. For others, such as *O. chrysophylla* in Site 11, we found only stumps (volumes of dead wood per species with distance into the park in Appendix 4.C).

a)



b)

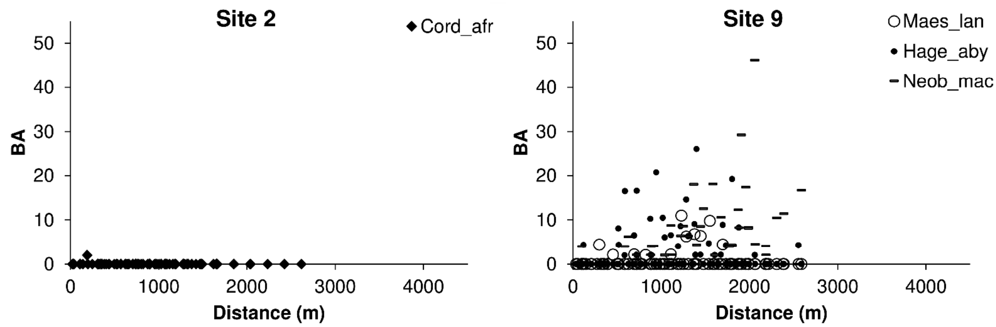


Figure 4.7. Basal area (BA) of a) preferred and b) used species for fuelwood with distance from the boundary. Preferred and used species are the same in Site 11 and 14, except *Ekebergia capensis* in Site 14 but no dead wood was found for that species. Key to species abbreviations: Anin_spp: *Aningeria* spp., Prun_afr: *Prunus africana,* Crot_spp: *Croton* spp., Allo_aby: *Allophylus abyssinicus,* Hage_aby: *Hagenia abyssinica,* Olea_wel: *Olea welwitschii,* Podo_mil: *Podocarpus milianjianus,* Olea_chr: *Olea chrysophylla,* Corn_vol: *Cornus volkensii,* Ekeb_cap: *Ekebergia capensis,* Maes_lan: *Maesa lanceolata,* Neob_mac: *Neoboutonia macrocalyx.*

4.5. Discussion

First, we discuss the importance of the forest inside Mt Elgon National Park for neighbouring communities as a source of fuelwood. We then reflect on how fuelwood collection has affected the availability of woody debris and the tree species composition in the four sites. We considered availability, location and preferences. Finally, we consider the likely implications of future demands for fuel and discuss options to address these impacts as well as needs for further research.

4.5.1. Fuelwood collection and use

Fuelwood collection on Mt Elgon appears to be more intense than in other areas studied in the region. The volume of 0.05 m^3 per headload we observed was greater than the standard 0.03 m^3 used to assess fuelwood consumption in other studies in the region (Banana and Turiho-habwe 1999). We calculated yearly quantities that were almost double the 4.5 m^3 per household or the 0.6-0.7 m^3 per capita found near Budongo forest in Uganda (see Table 4.3). In Rwanda, Ndayambaje and Mohren (2011) report a per capita consumption of 0.91 m^3 y^{-1}, while in a moist semi-deciduous forest in Ghana Osei (1993) reports a per capita use of between 1 and 1.2 m^3 per year. If we used the standard size of 0.03 m^3, then fuelwood collection (3.8-5.7 $m^3 y^{-1}$ per household) would be more consistent with the averages reported in Budongo (Banana and Turiho-habwe 1999). Other studies examined fuelwood *consumed* by households whereas we investigated the fuelwood *collected*. It is common for people, women mostly, to sell stocked fuelwood *ad hoc*, when cash is needed, which may partly explain the relatively high values we found on Mt Elgon compared with other studies. Other factors may contribute to the intense use of fuelwood on Mt Elgon. Cooking beans and bananas, which are a staple in Sites 2 and 9, require more fuel than maize and potatoes for example. Cooking times are extended at higher elevations, such as in our study sites (> 1900 masl). People did not boil water for drinking as the quality of water from natural springs was generally good (local informants, personal communication and personal experience), but they did heat water for bathing because it was often cold (< 10°C at night, especially in Sites 11 and 14). There have been past attempts to introducing fuel-saving stoves but they were used by none of the households that we stayed in (14 in total).

Other uses of wood for fuel that contribute to high wood consumption were unlikely to be included in the quantities reported by our respondents, e.g. commercial harvesting and charcoal making. Because commercial fuelwood collection is illegal, selling was likely under-reported (see Table 4.3). Commercial brick- and charcoal-making contribute to high wood consumption in other Ugandan forests (Naughton-Treves et al. 2007). In our study neither activity was mentioned. We did not observe brick-making in our sites, but found charcoal pits and evidence of commercial fuelwood harvesting in Sites 2 and 9 inside the park. Charcoal was already noted near Site 2 in the 1990s and has negatively impacted forest structure in this area (Sassen and Sheil 2013).

While more than half of our respondents also used fuelwood from elsewhere, the forest – which on Mt Elgon means the national park – was the most important source of fuelwood in terms of both its perceived importance and the quantities reported. Areas with the highest density of trees outside the park (on people's land) also had the most dense human populations, explaining why the park remains important to meet their fuelwood needs. The density of trees on people's land (outside the park) reflects local history. In densely-populated areas, such as Sites 2 and 9 that have been settled and cultivated since around 1500 AD, households had native trees as part of their intensive coffee-banana system and households with sufficient land had woodlots of exotic species (*Eucalyptus* spp.). In the less-densely populated areas such as Sites 11 and 14 that were settled only during the 20th century, there were few trees except for scattered forest individuals left after clearing land (Table 4.4). These relics were progressively being felled (personal observation). Here, the people were formerly pastoralists and have no culture of tree planting. Also tenure insecurity related to conflicts about the boundary of the area excised for resettlement and the allocation of land likely contributed to the fact that people planted few trees (Himmelfarb 2006).

Households with more trees on their own land tended to value their land as a source of fuelwood more and used old-growth forest less than those with fewer trees. For example in Sites 2 and 9, households with more trees on their own land ranked their own land highest (Table 4.3 and 4.4). In Site 2, despite the importance given to old growth forest, people reported using mainly species that grew on their own land (Table 4 and 6). In Site 9 people with more trees on their own land also ranked old-growth forest as a less important source of firewood (Table 4.3 and 4.4). In Sites 11 and 14, forest was the main source of fuelwood regardless of the number of trees on people's own land or the area of land owned, because there were few trees on their land in these sites (Appendix 4.A and Tables 2-4). Our results suggest that people forage less far for fuel if there are closer-by alternatives.

4.5.2. The quantity of fuelwood in the park

Fuelwood collection reduced the amount of dead wood in the forest, in particular in accessible areas close to the boundary. In addition, site-specific encroachment histories and other forest uses impacted local forest structure (Sassen and Sheil 2013), which in turn influenced dead wood availability at varying distances into the park. For example, in Sites 11 and 14 which are less impacted, previous encroachment had been less intense than in Sites 2 and 9 (Sassen et al. 2013). Other forest uses in Site 11 and 14 were mainly related to cattle-grazing which does not involve tree harvesting. In Site 9, the impacts of fuelwood collection were added to those of intense collection of stems for other uses, such as supports for banana and climbing beans (Sassen and Sheil 2013).The altitudinal range of our plots was too narrow for elevation to influence rates of decomposition markedly (Table 4.1), suggesting that removal was the main reason that increasing volumes of dead wood were found further into the park.

The selectivity of fuelwood collectors was influenced by the quantity and quality of dead wood available in combination with species preference. For example, in Site 9 people were less selective because of a shortage of fuelwood, especially in the formerly-encroached areas near the boundary (Figure 4.5). In Site 14, on the other hand, preferred species were still available and people collected these before others even when more decayed as they were often hardwood species.

4.5.3. Impacts of fuelwood collection on preferred woody species

The interviews as well as the field survey provided evidence that certain tree species in the forest were overexploited for fuelwood and other uses. Highly-preferred species were not necessarily the ones people actually used the most, which indicates a shortage of the preferred species. When queried about reasons for not using highly-preferred species, our informants always mentioned their depletion in accessible areas. According to them this was because they were also valued and harvested for timber (data not presented).

Results from the field survey confirmed findings from the interviews. The volumes of dead wood and the basal area of the most highly-preferred and used species were all smaller near the park boundary. Some species were particularly preferred and affected, especially slow growing hardwood species that are also valued for timber, e.g. *P. Africana*, *P. milianjianus*, *Aningeria* spp., *O. chrysophyla and O welwitschii* (Scott 1994a, Hitimana et al. 2010). Often they still occurred further inside the park (> 2000 m), but because they had become difficult to access our respondents did not rank them so highly in terms of actual use. For example in Site 9, *P. africana* was highly preferred but not highly used. Even though its woody debris represented 18% of all dead wood recorded in this site, stems were found mostly further from the boundary (Figure 4.7).

The degree of depletion of highly-preferred or used species was affected by site specific species composition. Stems of much preferred or used, but locally dominant, species such as *C.volkensii*, *A.abyssinicus*, *N. macrocalyx* and *H.abyssinica* (Table 4.6) were more abundant closer to the boundary than less dominant species, sometimes despite the fact that they also had other uses. For example, pioneer species such as *H. abyssinica* and *N. macrocalyx* that dominate the older regenerating areas of Site 9, were also much used as crop supports and poles. Other pioneers include *Vernonia* spp. which is a fast growing tree-like seasonal shrub that grows in degraded areas just inside the park boundary.

In places with less dense population and less historical degradation, the lack of alternatives may lead to increased pressure on preferred species in the future. In Site 11 and 14 most preferred species were still heavily used, meaning they were not yet as severely affected as in the other study sites. However, certain species that were both highly preferred and used - such as *O. chrysophyla* and *A. abyssinicus* in Site 11 and *E. capensis* in Site 14 - had small relative (Table 4.6) and absolute (Figure 4.7) basal area, suggesting they may become depleted if current use continues.

Community composition varied with elevation independently of human impacts, and care is required not to confuse natural distribution effects with human impacts. For example *A. abyssinicus* in Site 11 may have a restricted lower altitudinal range close to that of the boundary of the park in this area. *P. milianjianus* in Site 9 seemed depleted close to the boundary but may also have been restricted to higher elevation areas within this site. Respondents in Sites 2 and 9 listed a higher number of species than respondents in Sites 11 and 14, which is likely because species richness was higher in Sites 2 and 9 (Sassen and Sheil 2013).

4.5.4. Future developments

Fuelwood collection is important for local people on Mt Elgon. They lack sufficient alternatives and this dependence affects the forest. Fuelwood collection and other forest uses impact forest structure (Sassen and Sheil 2013), species composition and the availability of woody debris (this study). This in turn affects forest functioning and its ability to provide important resources for local populations in the long run.

As population increases, demand for fuelwood is likely to grow. Our study indicates that this may lead to further forest degradation, both in intensity and extent. On Mt Elgon new roads are constructed and access is improved (Sassen et al. 2013). On each of our visits between 2009 and 2011, we observed new buildings being constructed in the small trading centres along the roads or paths leading to the park. The main town of Mbale (Figure 4.1) is expanding (Mbale District 2007) which will lead to an increase in commercial fuelwood harvesting and an increasing demand for charcoal (Girard 2002, Bensel 2008, Zulu and Richardson 2012). Commercial fuelwood extraction and charcoal production can lead to much more severe impacts on the forest than dead wood collection because they entire trees are removed (Mwampamba 2007, Sassen and Sheil 2013).

Allowing people access to the forest is double edged. Currently some communities around the park are given legal access to the park to collect dead wood and non-timber resources. In areas without formal agreements (e.g. Sites 2, 11 and 14) local rangers sometimes tolerate the collection of dead wood to avoid conflicts (Community conservation ranger, personal communication). On the one hand, granting people formal or informal access aids relations between the park and local people, and may help curb agricultural encroachment (Sassen et al. 2013). But the park management lacks the means to enforce the rules of the agreements and local forest user committees are unable or unwilling to impose them. Thus, illegal activities abound, including cutting of whole trees. In previous assessments there were indications that human uses had less impact on forest cover and structure in sites with a collaborative management agreement compared with sites where there was no agreement (Sassen and Sheil 2013, Sassen et al. 2013). But we also found most intense commercial fuelwood harvesting in a site where an agreement with park authorities gave people access twice a week (Sassen and Sheil 2013). We also observed that people from neighbouring parishes entered the forest on each other’s allocated resource collection days. Completely banning fuelwood collection from the park is unrealistic as many people rely on it at least party. Even if off-take was better monitored and found to be unsustainable, it will be impossible to stop people from entering the park. But dead wood collection and tree cutting should be viewed differently and there is a need for a system that regulates or controls the cutting of whole trees which at the same time allows the collection of dead wood.

No one wins by losing the forest. There is a need to investigate ways in which local people can be empowered to have more ownership and control over the forest and how this can lead to more effective forest management. An important insight is that forest degradation is more likely where people view forests as an open-access resource rather than a common-pool resource (Ostrom 1999). The resource use agreements on Mt Elgon are an attempt to give people more ownership over resources in exchange for forest protection, but the degree of human impacts indicates that they are inadequate (Sassen and Sheil 2013). However, in sites with the highest pressure (Sites 2 and 9), people realise the impacts of firewood collection and other uses such as timber harvesting on the provision of these resources, even if they do not always act accordingly. Research has shown that such realisation is an important condition for the development of sustainable local rules (Ostrom 1999). Ostrom and others have found that congruence of local ecological and cultural contexts and perceived benefits and costs affect the success and sustainability of common-pool management arrangements (Ostrom and Hess 2010). They also developed a set of design principles for sustainable common-pool resource management regimes (Ostrom 1999), which could guide successful forest management on Mt Elgon. As an example, it may be possible to determine different rules with local communities regarding the use of slow-growing old growth forest species that are also valued for timber, trees with multiple uses and fast-growing pioneer species, if the local communities had a clear stake in the outcomes. Research on the productivity of tree species used for fuelwood by local communities could help inform such decisions (Top et al. 2004).

In addition to more inclusive management regimes, it is important to investigate options for alternative sources of fuel. Access to alternatives can make a difference in the importance people attribute to these alternatives and to the park as a source of fuelwood. Even when trees are not planted to provide fuelwood, they are often highly valued as sources of fuel (Kindt et al. 2004, Arnold et al. 2006, Ndayambaje and Mohren 2011). For instance, in Site 2, *M. platycalyx* and *C. Africana*, which are commonly planted as shade trees for coffee, were also amongst the most used species for fuelwood. People with more land planted more trees but not in proportion to their land area. Identifying the right incentives may therefore help increase tree density outside the park in particular in areas where few trees are planted traditionally. Although people with little or no land may have fewer options to plant trees, the most densely populated areas on Mt Elgon were also the ones with the highest tree density outside the park. Forest use for fuelwood or other uses, varies among local communities which leads to different impacts within one protected area. There is clearly a need to look for locally appropriate options, incentives and alternatives that balance the needs of local livelihoods and forest conservation.

4.6. Conclusions

Fuelwood demand around Mt Elgon has intense but varying impacts over a large area of the park. Key old growth forest species were most affected furthest into the park, whereas there were less severe impacts on pioneer and locally dominant species. Yet all tree species that people used for fuelwood were negatively affected on the edges of the park. Similar impacts are likely in other forests surrounded by dense human populations with limited access to alternative sources of fuel. Our results demonstrate the strong impacts of fuelwood extraction in combination with other uses, which will have future consequences for both forest dependent people and conservation. Pressure on forests for fuel is likely to increase and needs more attention.

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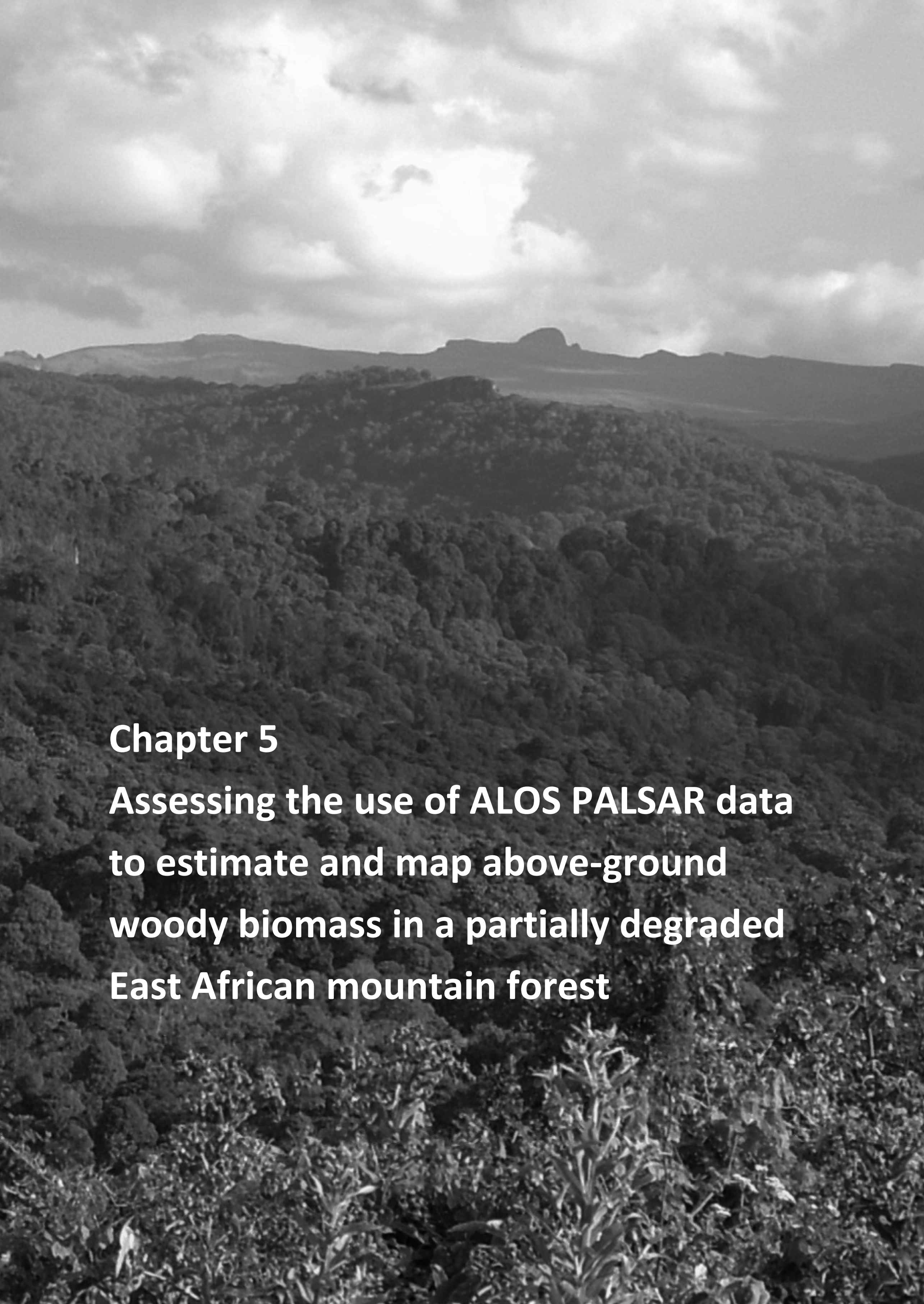

Chapter 5
Assessing the use of ALOS PALSAR data to estimate and map above-ground woody biomass in a partially degraded East African mountain forest

Sassen, M., Quiñones, M.J., Sheil, D., Mitchard, E. and van der Linden, J.F. (in preparation)

5.1. Introduction

Biomass from tropical forests plays an important role in sequestering carbon to offset climate change (DeFries et al. 2002, Pan et al. 2011). However the magnitude of the amount of carbon stored in the biomass of tropical forests is not well known, in particular in Sub-Saharan Africa (Glenday 2006, Houghton and Hackler 2006). The distribution of carbon in forests is affected by past and present disturbance, including the history of deforestation, degradation and regeneration. Degradation is often overlooked because it is more difficult to detect using remote sensing than deforestation (Putz and Redford 2010). Nevertheless, many tropical forests are degraded to some extent and are composed of many intermediate vegetation cover classes that can store significant amounts of carbon (Mitchard et al. 2012, Ryan et al. 2012). These degraded and secondary forests are often also important for local livelihoods and conservation (Wright 2005, Chazdon et al. 2009). Understanding the impact of disturbance on carbon stocks, and their rate of recovery following such disturbance, is critical in light of planned implementation of the Reduced Emissions from Deforestation and Forest Degradation (REDD+) policy mechanism under the United Nations Framework Convention on Climate Change (UNFCCC).

Local communities living near tropical forests are often highly dependent on forest resources for their livelihoods. In densely populated areas where people experience land scarcity, conflicts about the use of forest land and resources are common (Balmford et al. 2001). In principle, REDD+ or other payments for environmental services schemes could potentially provide alternative incomes and lower the dependence of local communities on the forest. However, REDD+ schemes that include local livelihoods and conservation objectives require cost effective measures of carbon stocks with known uncertainties.

Degradation processes are often gradual and take place on a small scale and are therefore difficult to detect remotely (GOFC-GOLD 2009). Small-scale forest degradation monitoring requires extensive repeated field measurements of local biomass extraction for e.g. cultivation, timber and fuelwood extraction. These measurements are rarely carried-out and unrealistic in resource-constrained conservation areas in the tropics. Optical remote sensing has limited capability to detect degradation in tropical forest because it sees only the top of the canopy. However, small scale and local degradation can happen without the canopy cover changing significantly (Mitchard et al. 2012, Ryan et al. 2012). In time series, variations in optical signatures can also be related to atmospheric effects, clouds and cloud shadows rather than to forest degradation (Saatchi et al. 2001). However, radar data offers low cost, regular monitoring possibilities (Mitchard et al. 2009) that do not suffer from cloud effects, which are often a limitation over tropical forests (Saatchi et al. 2001).

Research on the use of radar imagery for above ground biomass (AGB) estimation dates from the 1990s and was mostly focussed on the assessment of airborne radar data for biomass estimations (Dobson et al. 1992, Le Toan et al. 1992, Ranson and Sun 1994, Imhoff 1995, Kasischke et al. 1995, Rignot et al. 1995, Hoekman and Quiriones 2000), though attempts were also made using the L-band JAXA satellite JERS-1 (Santos et al. 2002). More recently remote sensing radar images have been used to consistently map and monitor tropical forest (Saatchi et al. 2007, Quiñones et al. 2008, 2009, Hoekman et al. 2010, Quiñones and Hoekman 2011). L-band ALOS PALSAR Fine beam dual polarisation (FBD) space-borne radar imagery was found to be well correlated with above ground woody biomass in different tropical forests including several African forest landscapes (Mitchard et al. 2009, Morel et al. 2011, Ryan et al. 2012). Nevertheless such correlations are better in flat terrain, whereas in complex mountain ecosystems with forests growing on steep hills this relation is expected to be affected by geometric effects like fore-shortening, layover or radar shadow and by radiometric effects where slopes facing the sensor appear brighter and slopes facing away appear darker.

The forest on Mt Elgon, Uganda is characterised by steep slopes and has a long history of deforestation and forest degradation (Sassen et al. 2013). Much effort has been put into designing alternative conservation strategies and into the search for mechanisms to support management and ecosystem recovery. In the 1990s a foreign-funded carbon offset project funded restoration planting in formerly encroached areas of the park (UWA 2000). More recently, the Mount Elgon Regional Ecosystems Conservation Project (MERECP) has initiated pilot REDD+ type projects in a number of locations around and inside the protected area (LVBC 2009). This type of project could benefit from consistent remote sensing observations and space borne biomass estimations to provide carbon levels, detect deforestation and degradation and monitor forest dynamics. The transparency provided by a consistent radar based monitoring system would help safeguard the financial benefits of local REDD+ projects to local communities.

In this study we explore whether radar remote sensing can be used for above ground biomass (carbon) mapping in a complex, degraded mountainous forest such as on Mt Elgon. We first use existing allometric equations to calculate biomass from forest plot data. We then investigate the relationship between ALOS PALSAR Fine-Beam Dual (FBD) mode and the estimated biomass derived from the field measurements. We investigate three different approaches for describing the relationship between estimated field biomass and radar backscatter and evaluate the fit between backscatter and ground measured AGB for these different approaches. As other factors than biomass may affect backscatter, we discuss the possibility of adding additional explanatory variables such as elevation, slope and aspect for the different approaches.

Our goal was to investigate whether biomass estimations obtained using a relatively simple and low-cost angle-count method based on a relascope, were related to radar backscatter and if that relationship could be used for biomass mapping and with which accuracy. We also explore the possible use of an alternative method for mountainous ecosystems by performing a vegetation-structural classification of the radar data as a possible alternative for forest biomass, forest deforestation and forest loss monitoring. We expected that patterns of vegetation structure would be reflected in the biomass maps that result from applying the biomass-backscatter equations.

5.2. Methods

5.2.1. Study site

Mt Elgon (4321m) is an extinct solitary volcano on the border between Uganda and Kenya. The slopes of Mt Elgon generally average less than 4 degrees, but there are characteristic natural terraces cut by sheer cliffs in the north, and steep slopes in the south and south-west. The climate is determined by dry north-easterly and moist south-westerly winds. July-August and December-February are relatively dry, although rain falls in all months. Protected areas cover approximately 1120 km^2 in Uganda and 1400 km^2 in Kenya (Figure 5.1) and no natural forest remains within at least 20km outside their boundaries. Estimated annual precipitation is between 1500 and 2500 mm (IUCN 2005). More rain falls on the western and south-western slopes and most falls in the forest zone, mid-slope at between 2000-3000 m elevation (m.a.s.l.) (Dale 1940, IUCN 2005).

Mt Elgon is an important water catchment area for several million people in the surrounding districts and has significant biodiversity values (Davenport et al. 1996, IUCN 2005). Historically and at the time of this study, the forest on Mt Elgon are also an important source of agricultural land, timber, fuel wood and other forest resources for local communities (Scott 1994a, Sassen and Sheil 2013). On the Ugandan side, large scale deforestation took place in the 1970s and 1980s with subsequent recovery after 1993, when a national park was established to protect the forest and the higher altitude moorlands. Since then regeneration, renewed encroachment and local forest use have led to various degrees of recovery and degradation in different places inside the park (Sassen et al. 2013).

Mount Elgon has a history of conservation and development projects (since the early 1990s) that aim to support alternative livelihood options for neighbouring communities with mixed results (UWA 2000, LVBC 2009) .More recent pilot REDD+ initiatives aim to build on this in areas outside the park while also providing incentives to local groups to restore and protect the forest inside the park (LVBC 2009).

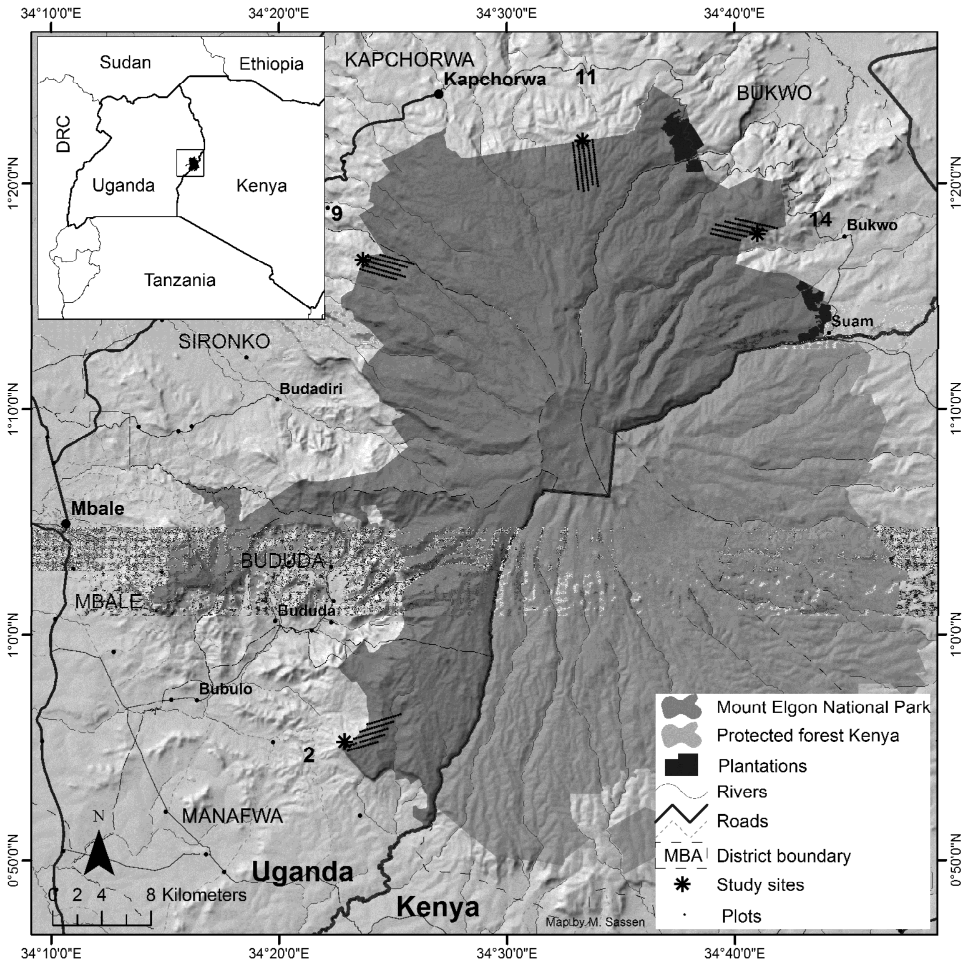

Figure 5.1. Map of Mt Elgon, Uganda/ Kenya (administrative divisions were valid in 2010).

5.2.2. Field data

Field data were collected between November 2010 and April 2011 (dry season) in 343 plots in 4 sites (Figure 5.1). In each site, plots were established along five parallel transects 400 m apart, pointing into the interior of the park (Figure 5.1). The centre of the first plot on each transect was located 50 m inside the park boundary and further plot-centres at 200 m intervals. We sampled 13 to 21 plots along each transect, locating them using a handheld GPS (Garmin 60CSx). The number of plots depended on accessibility. In each plot we used an angle-count or relascope method to directly estimate tree basal area (BA). In 77 of the plots we also measured the diameter at 1.3 m (dbh) of all trees counted in by the relascope, estimated tree height when possible and identified tree species.

Height data was missing for 36 out of 632 measured live stems because the canopy was not always visible to reliably estimate tree height. We used the measurements from 541 trees that were alive and unbroken to estimate missing heights using species- and site specific power-law regression equations (3 cases). If insufficient ($n \leq 5$ and model not significant at $p < 0.05$) site-specific data were available then species-specific equations for all study sites combined were used (8 cases). If there were insufficient stems of the species in all study sites combined ($n \leq 5$ and model not significant at $p < 0.05$), then a site specific equation based-on all stems in that site was used (3 cases).

Because the location of our plots was determined by their distance from the boundary, we did not necessarily select homogenous areas for basal area measurements, or only areas that were forested. In fact many of our plots close to the boundary were heavily disturbed. In these plots, there were sometimes no trees within range of the relascope and they were then recorded as empty (“zero-value plots”).

5.2.3. Above Ground Biomass (AGB) in field plots

We estimated calculated above ground biomass (AGB in Mg/ha) in 77 plots with dbh and tree-height data (Eq. 1) but also using only dbh (Eq. 2) using the pan-tropical stem-by-stem allometric equations for moist forest based developed by Chave et al. (2005). These equations have been found suitable in various other African tropical forests (Mitchard et al. 2009, Mitchard et al. 2011, Ryan et al. 2012).

$$AGB = exp[-2.977+\ln(\rho D^2 H)] \qquad \text{(Eq. 1)}$$

$$AGB = \rho \times \exp[-1.499 + 2.148\ln(D) + 0.207(\ln(D))^2 - 0.0281(\ln(D))^3] \qquad \text{(Eq. 2)}$$

Where ρ is the wood specific gravity (g/ cm^3), D is the dbh (cm) measured at 1.3 m and H is the estimated tree height (m) (when available). We used Wood Specific Gravity (WSG) data from the Global Wood density Database (GWDD) (Chave et al. 2009). When multiple values existed for one species they were averaged. When there was no value for the species, we used the average of values for species of the same genus in Africa (n=11 of total dataset of 63 species). If no information for the genus was available then we calculated and used the average from all other trees species in our study (n=10).

Because we used a plotless relascope method to select the trees, each stem represented a number of stems per hectare. We therefore divided the AGB per stem by its virtual plot area to obtain the AGB represented by each stem. Summing these per plot resulted in the estimated AGB/ha for each plot. Our AGB measures did not include vegetation other than trees. We then developed regression equations for the relation between measured BA and calculated AGB in the 77 measured plots (see Figure 5.2) to estimate the AGB/ha from plot basal area (m^2/ha) for plots with only direct BA estimations (n = 266).

5.2.4.Radar data

The radar imagery used was from the Phased Array L-band Synthetic Aperture Radar sensor aboard the Advanced Land Observing Satellite (ALOS PALSAR), acquired by the Japanese Space agency JAXA and distributed by the European Space Agency (ESA). We acquired two scenes on each date to cover Mt Elgon. The scenes were captured on 08/09/2007 and 01/08/2010 in the Fine-Beam Dual (FBD) mode: Horizontal-send Horizontal-receive (HH) and cross-polarised, Horizontal send Vertical-receive (HV). L-band SAR imagery is known for its ability to penetrate the forest canopy making it sensitive to forest biomass (Almeida et al. 2005, Fransson et al. 2007, Mitchard et al. 2009). The images were processed using standard approaches available in the image processing software ENVI 4.6.1. (ITT Systems) and automated scripts developed by SarVision. ALOS PALSAR standard FBD images were processed at 20 m. resolution. Radar data processing and corrections included radiometric absolute calibration, geocoding and geometric and radiometric terrain corrections that allow for a partial correction of the radar signature for slope. The resulting *Gamma naught* backscatter values are scaled to a decibel scale (dB) (for details see Appendix 5.A).

The geometric and radiometric terrain corrections were attempted using the 90 m resolution Shuttle Radar Topography Mission (SRTM) Digital Elevation Model (DEM) processed by the CGIAR Consortium for Spatial Information (http://srtm.csi.cgiar.org/). Unfortunately no higher resolution DEM was available, so it was likely that significant terrain artefacts would remain. Layover and shadow effects could not be corrected though as no data was collected in these areas (see black edges on Figures 5.5 and 5.6, representing steep cliffs).

5.2.5. Backscatter data extraction

We extracted the average backscatter values for each plot averaged from circular areas of 52-pixels (representing an area of 2.08 ha) around the GPS position of each plot-centre. Sampling areas for radar data processed at 20m resolution need to have more than 144 looks to be considered free of speckle (calculation based on the number of looks, see Hoekman and Quiriones 2000). ALOS has four looks per original pixel, so we needed extraction areas of at least 30 pixels. We derived aspect and slope data from the DEM (but see 5.4.2.) and investigated the effect of removing plots on steep slopes. Elevation was measured in the field using a handheld GPS (Garmin 60CSx).

5.2.6. Backscatter relationship with AGB

Contrasting points of view on how to best fit backscatter to AGBgm led to three different approaches to describe the relationship between backscatter and AGBgm. The contrasting points of view were:

1) including versus excluding plots with zero AGBgm values in the analysis.

2) explaining AGBgm from backscatter data versus (the other way around) explaining backscatter data from AGBgm (see also Ryan et al. 2012).

The first contrast is related to the fact that for most methods the ground based estimation of AGB is systematically underestimating actual biomass (e.g. they only measure trees with a minimum diameter and not shrubs). This means that plots that are recorded as bare can in reality be vegetated. On the one hand one could leave out the “zero-value” plots because they don’t add predictive power to the relationship between AGB and backscatter. On the other hand one can argue that the goal of finding a relationship between AGBgm and backscatter is to be able to predict AGB from space. From space you cannot recognize plots that would get a “zero value” for AGBgm when measured in the field and therefore all plots should be included in the analysis.

The second contrast - regarding which of the variables (AGBgm or backscatter) is the explanatory and which is the explained - is a matter of personal preference. Intuitively one would define AGBgm as the explanatory and backscatter as the explained variable, since backscatter is thought to depend on AGB. The other way around though, one could argue that backscatter is only partly the result of AGB (but also of terrain, soil moisture,

vegetation structure, etc.) and that the field measured AGB is (just like backscatter) an estimate of the real AGB.
The two contrasts meet in a mathematical practicality. Namely, having backscatter as the explained variable does not allow for the inclusion of AGBgm "zero values" when fitting log-based models, since this would result in having to take the logarithm of zero. This leaves three possible approaches to investigating the relationship between backscatter and AGBgm:
1) AGBgm as the explained variable and AGBgm "zero values" included.
2) AGBgm as the explained variable and AGBgm "zero values" excluded.
3) backscatter as the explained variable and AGBgm "zero values" excluded.

Besides having backscatter or AGBgm as explanatory variable we investigated the effect on the overall fit of the additional explanatory variables, slope, aspect and elevation. Aspect was described by two variables; one North-South variable and one East-West variable. All additional explanatory variables were first plotted to AGBgm one by one to investigate a possible relationship. Based on this preliminary investigation, equations were constructed to fit this relationship. All variables were fitted to AGBgm by starting off with a simple linear relationship and building in more complexity towards the relationship found through the preliminary investigation.

Aboveground biomass is commonly estimated by converting radar backscatter data through a reduced major axis (RMA) regression (Mitchard et al. 2012, Ryan et al. 2012): "RMA regression minimizes the errors on both axes (rather than just on the y-axis as in normal regression), which is appropriate because there are errors in both data sets and the observer controls neither […]." (Ryan et al. 2012). However, one major drawback of RMA regression is that, as the slope departs from ±1, the RMA slope estimate is increasingly biased and the confidence interval includes the true value less and less often (http://cran.r-project.org/web/packages/lmodel2/vignettes/mod2user.pdf). Since part of the models that we wanted to compare with each other would have RMA slopes strongly deviating from ±1, we could not use RMA regression. Also, ordinary least square (OLS) regression could not be applied as our data were not normally distributed (and could not be transformed to a normal distribution) and contained outliers. Therefore we used nonlinear robust regression (MATLAB 7.5.0). Robust regression is specifically designed to deal with outliers and non-normally distributed data.

Since our data is highly skewed and in some of our approaches zero-inflated, we expected difficulties in the evaluation of residual plots. Therefore, we fitted a parabolic function through the residuals in order to more easily assess whether the models describing the relationship between AGBgm and backscatter were biased. Furthermore, Q-Q plots were constructed to assess whether the predicted values linearly described the observed values. All analysis was performed using Matlab 7.5.0.

5.2.7. Vegetation classification

We used a pixel-based combined unsupervised/ supervised classification algorithm. First, an unsupervised classification procedure created 30 classes, which for complex tropical forest ecosystems has been found to best describe the variation in the image, based on the Bayesian Information Criterion (for details see Hoekman et al. 2010) (further details in Appendix A). The resulting classification was used to evaluate vegetation patterns based on ancillary information (van Heist 1994, KWS et al. 2001) and expert knowledge (field observations). Based on this evaluation, 25 classes were retained and training data extracted using polygons over the radar images. The training data was then used in a supervised classification procedure. Classes were identified and labelled using a combination of ancillary data and expert knowledge of the area. A Geographical Information System (GIS) was created using the different available map layers and the radar ALOS PALSAR images to aid interpretation of the images. Finally, some of the classes among the 25 were merged based on interpretation of the radar backscatter values in relation to known vegetation structure. Formal validation was not attempted within the scope of this study but will be included in a later version of this paper.

5.3. Results

5.3.1. AGB from field data

The AGB calculated for the plots with measured dbh and height varied between 0 and 731 Mg/ha. Ranges were higher when using the equation with only dbh (Figure 5.2.b, data in Appendix 5.B). We calculated linear regression equations between BA and AGBgm in both cases (Figure 5.2.a. and 6.2.b.).

For each measured plot we derived two different estimates of AGB: one resulting from the model including dbh and height (Figure 5.2.a) and the other from the model with only dbh (Figure 5.2.b).

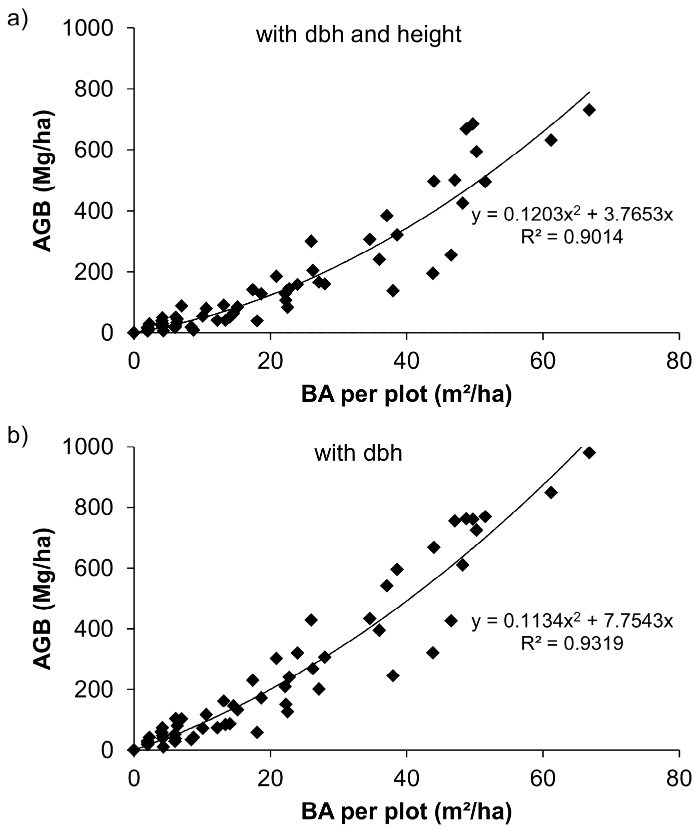

Figure 5.2. Relationship between basal area (BA) per plot and AGB calculated from height and dbh (a) and only from dbh (b).

5.3.2. Plot-level AGB backscatter relationships

To further reduce the effects of terrain (see section 5.2.4 on terrain corrections) on the relationship between AGB and backscatter, we excluded all plots with slope angle greater than 30 degrees as radar extractions for these plots show an extremely high heterogeneity (standard deviation larger than 1.5 dB) compared to plots on less steep terrain. This difference in heterogeneity could not be corrected for and could not be explained by the heterogeneity in the ground based observations of AGB from these plots and was therefore treated as a measuring error. Removing plots with slopes above 10 degrees would have resulted in an even cleaner signal, but would have left only 74 plots for analysis (out of 343). Two more plots were excluded from the analysis, because they were situated on a cliff top next to a steep drop, causing their backscatter values to be unrealistically high.

AGBgm estimated using the equation that included dbh and height (Figure 5.2.a) versus HV backscatter, consistently gave a better fit than AGB estimated without height and BA. AGB (any model) or BA versus HH backscatter performed less well than HV. Therefore only relationships between AGB estimated using dbh and height (hereafter referred to as AGBgm) and HV are presented.

Significant relationships between AGBgm and HV backscatter were found for all three approaches (Figure 5.3, Table 5.1). The approach where AGBgm was explained by backscatter and where zero values were included (Figure 5.3.a) showed a higher correlation than the two approaches without zeros (Figure 5.3.d, g). In our case most zero AGBgm values were correlated with a range of backscatter values where the best fit curve would be close to the x-axis even if there were no zeros. The difference between the zero and the predicted value on the best fit curve is therefore smaller than the average difference between the measured values and their predicted values on the curve, leading to a better correlation than when zeros are not included.

Intuitively one would favour a higher correlation over a lower one, since a high correlation is beneficial for a more accurate prediction of biomass by backscatter (also reflected by a narrower confidence interval). In this case though, the left part of the best fit curve (Figure 5.3.a) is flattening out to horizontal while the right part is getting steeper, therewith decreasing the discriminative power of backscatter over AGBgm, i.e. virtually all backscatter measurements less than -17 will result in a prediction of zero ABG, while a minute difference in backscatter above -13 can result in an error of 100 Mg AGB/ ha. Therefore in our case, there is a trade-off between accuracy (Figure 5.3.a) of the relationship and discriminative power (Figure 5.3.b) of backscatter over AGB. Nevertheless, the residuals plot shows virtually no bias and the Q-Q plot suggests a linear relationship between the observed and predicted values. Observed values in the Q-Q plot peak around 800 Mg/ha, while predicted values reach 400 Mg/ha; this large difference is caused by the large spread in AGBgm measurements around their predicted values. The saturation of the radar backscatter response above 150-200Mg/ha is well illustrated in all models presented in Figure 5.3: changes in response above this point therefore do not have much relevance. The differences between the fits below this saturation point are however still significant.

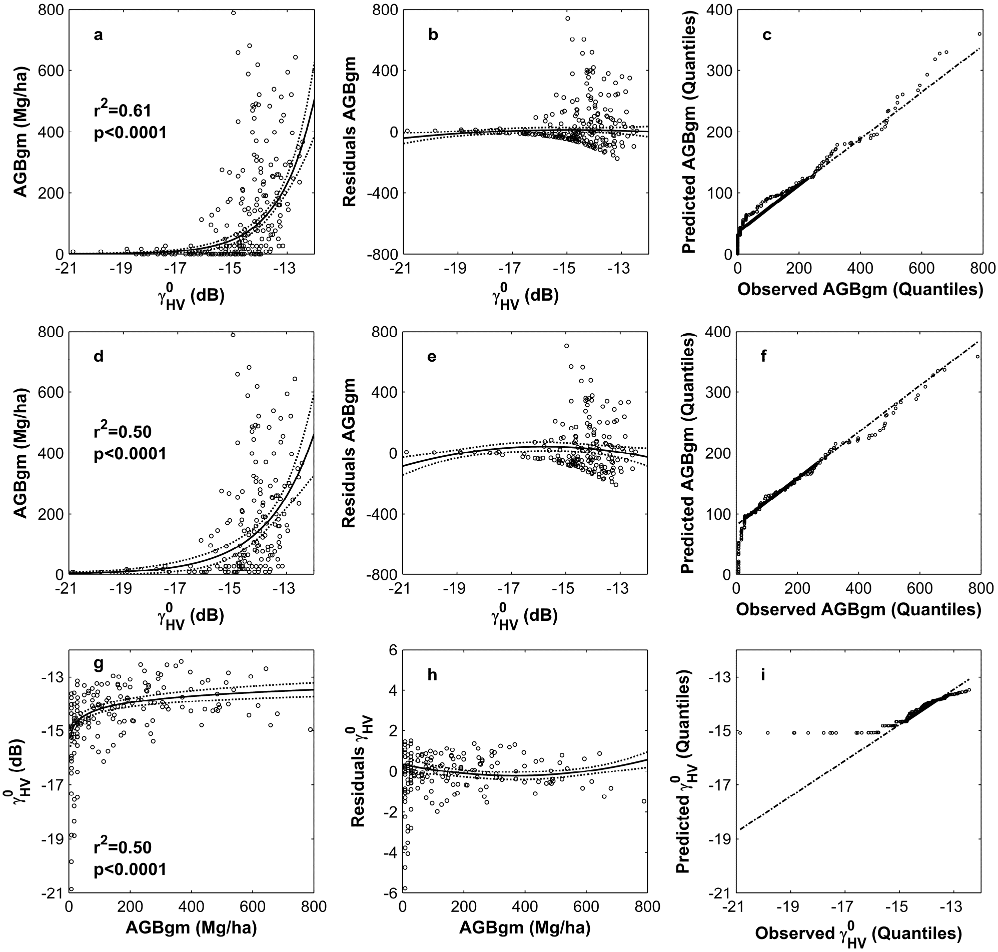

Figure 5.3. AGBgm (Mg/ha) plotted against HV backscatter (a,d). HV backscatter plotted against AGBgm (Mg/ha) (g). Residual plots with fitted parabolic functions (second column). Q-Q plots for observed vs. predicted values (third column). Columns 1 & 2, solid line is robust fit nonlinear regression, dotted lines are 95% confidence interval. Column 3, solid line is between 25th and 75th quantile, dotted line is extrapolation of solid line to assess fit for the outer quarters.

There is no difference between the correlations of the second (direct) and the third (indirect) approach, since they are based on the same data (Table 5.1). The predictive power (Table 5.1) though is a manifold higher in the direct estimation (Figure 5.3.d) compared to the indirect estimation (Figure 5.3.g). This can be clearly seen in the Q-Q plot (Figure 5.3.i), where predictions only start at -15 and quickly saturate around -13.5. This difference in predictive power can be explained by the fact that in the direct approach the error in AGB is minimised, whereas in the direct approach the error in the backscatter is minimised. The residuals plots of the direct en indirect approaches both without zeros show a backwards mirrored bias of each other that can be explained by the fact that their equations can be rearranged between them.

Table 5.1. Regression statistics for the relationship between radar backscatter and field measured aboveground woody biomass (Mg/ha) for the three different approaches. Where $\hat{b}_1$ & $\hat{b}_2$ are parameter estimates ± 95% confidence interval, r^2 is Spearman's rank correlation coefficient en p is the probability of having 0 correlation.

	Direct, Zeros included	**Direct Zeros, excluded**	**Indirect, Zeros excluded**
n	211	169	169
Model	$\mathrm{AGBgm} = e^{\left[\frac{\gamma^{\circ}_{HV}+\hat{b}_1}{\hat{b}_2}\right]}$	$\mathrm{AGBgm} = e^{\left[\frac{\gamma^{\circ}_{HV}+\hat{b}_1}{\hat{b}_2}\right]}$	$\gamma^{\circ}_{HV} = \hat{b}_1 + \hat{b}_2 \ln(\mathrm{AGBgm})$
$\hat{\mathbf{b}}_1$	19.92 ± 1.31	22.69 ± 2.81	-15.82 ± 0.47
$\hat{\mathbf{b}}_2$	1.27 ± 0.25	1.74 ± 0.53	0.35 ± 0.10
r2	0.61	0.50	0.50
p	<0.0001	<0.0001	<0.0001
RMSE*	88.91	125.61	0.87

* in Mg/ha

Amongst the additional explanatory variables, slope, aspect and elevation, only elevation significantly contributed to the improvement of the correlation between backscatter and AGBgm (direct approach with zeros included) (Figure 5.4). The fitted relationship was of the form:

$$AGBgm = e^{\left[\frac{\gamma^{\circ}_{HV}+\hat{b}_1}{\hat{b}_2}\right]} + \hat{b}_3 El + \hat{b}_4 El^2$$

Where γ°_{HV} is HV backscatter and El is elevation; coefficients (± 95% confidence intervals): $\hat{b}_1 = 29.15 \pm 11.12$, $\hat{b}_2 = 2.87 \pm 1.84$, $\hat{b}_3 = -0{,}27 \pm 0.10$, $\hat{b}_4 = 0.0001 \pm 0.00003$; $r^2 = 0.68$; p<0.0001; $RMSE = 89.56\ Mg/ha$.

Compared to the model without elevation, the Spearman's correlation coefficient increased from $r^2 = 0.61$ to $r^2 = 0.68$. Although the fitted model does not show bias in the residuals plot, one can see in the Q-Q plot that the fitted relationship does not describe the observed values linearly. Moreover the fitted relationship shows a "double band" in the predicted values. This is caused by plots at high elevation with a lower than expected AGBgm (and therefore a lower than expected HV backscatter value).

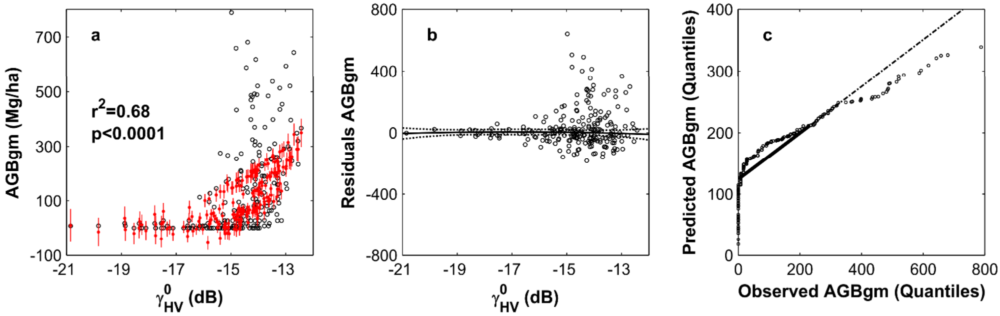

Figure 5.4. (a) AGBgm (Mg/ha) plotted against HV backscatter, red dots with lines are predicted values with 95% confidence intervals. (b) Residuals plot with fitted parabolic functions, solid line is robust fit nonlinear regression and dotted lines are 95% confidence interval. (c) Q-Q plots for observed vs. predicted values, solid line is between 25th and 75th quantile, dotted line is extrapolation of solid line to assess fit for the outer quarters.

5.3.3. Mapping biomass

We used the equations developed through all three approaches (Table 5.2) to invert the radar HV images of 2010 into biomass maps (Figure 5.5.a, b, c). We did not have plots outside the protected area so the low backscatter values of the northern "bare" areas are poorly represented in the relationship between AGB and backscatter – leading to a narrow range of backscatter values (5 dB) (Table 5.2)

Table 5.2: Equations used to calculate biomass maps for Mt Elgon (Figure 5.5)

Explored options to map biomass from AGB	**Equation**
AGB estimations based-on in-situ data measured on Mt Elgon. Approach: Direct, zeros included	$AGB = e^{\left[\frac{\gamma_{HV}^{0}+19.92}{1.27}\right]}$ (Figure 5.4a)
AGB estimations based-on in-situ data measured on Mt Elgon. Approach: Direct, zeros excluded	$AGB = e^{\left[\frac{\gamma_{HV}^{0}+22.69}{1.74}\right]}$ (Figure 5.4b)
AGB estimations based-on in-situ data measured on Mt Elgon. Approach: Indirect, zeros excluded	$AGB = e^{\left[\frac{\gamma_{HV}^{0}+15.82}{0.35}\right]}$ (Figure 5.4c)

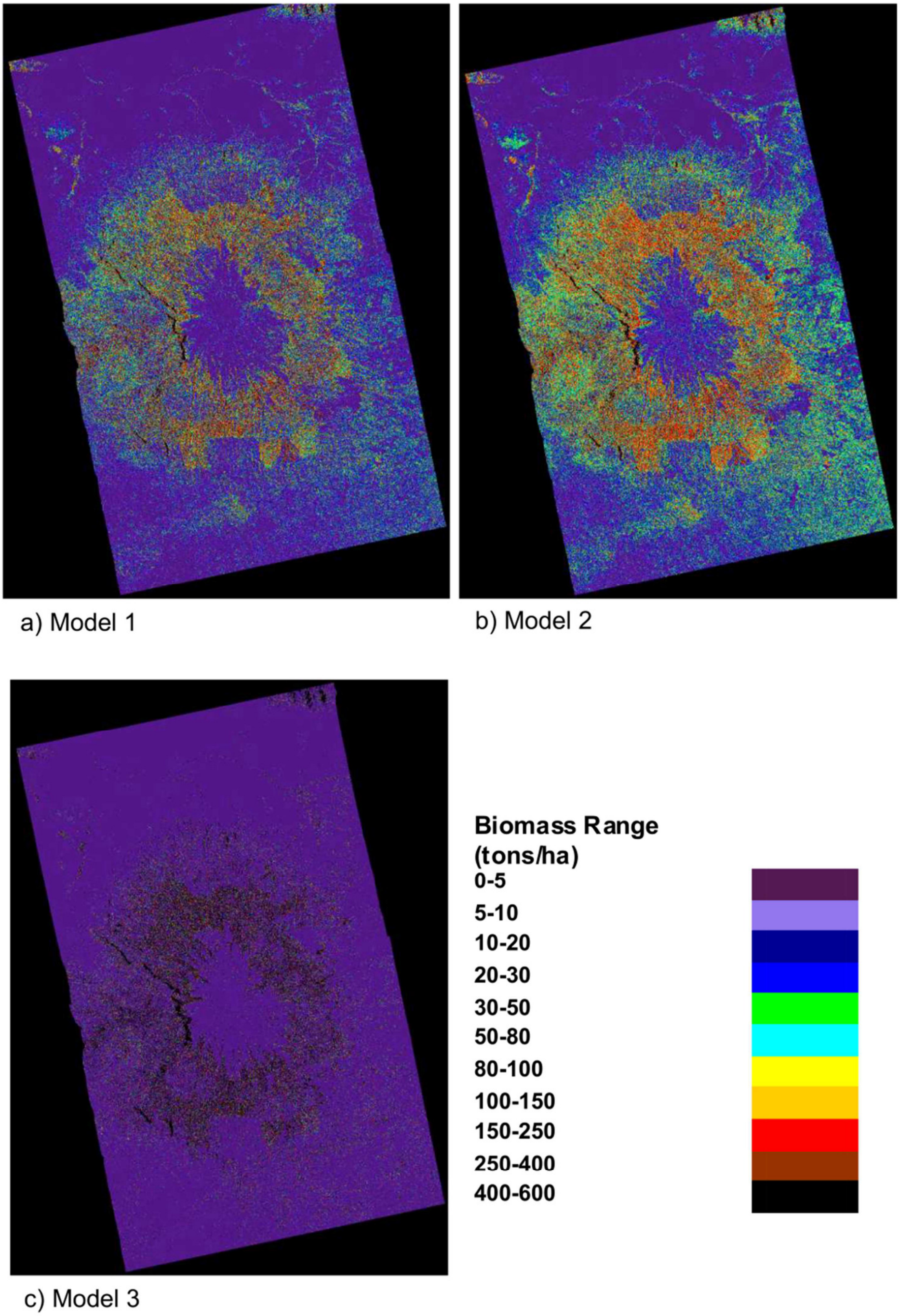

Figure 5.5. Above ground biomass maps of Mt Elgon, Uganda for 2010, using three equations based on the data from this study.

5.3.4. Vegetation classification

The classification procedure yielded 14 vegetation classes of stable and changing vegetation (between 2007 and 2010), 1 water class and a class corresponding to slopes that were too steep to correct for and therefore have no data. Many bare and grassland areas in the lower-lying areas to the north of the mountain showed variation due to fluctuating regimes of flooding and drying. These classes were merged. Most change was apparent in the agricultural areas in the lower laying areas around the mountain and in plantation areas on the Eastern slopes (Figure 5.6). The classification shows vegetation structure varying from dense mountain forest with patches of bamboo and woodlands, to different densities of bushland, grassland and “bare” areas in the north.

These vegetation classes are distinct in their levels of biomass and because of the interaction between radar backscatter and aspects of vegetation structure we expected to see these patterns back in the results of the biomass maps (see 5.3.3.).

5.4. Discussion and conclusions

5.4.1. AGB estimations based on angle-count methods

It is known that basal area is strongly correlated with AGB, as both AGB and basal area depend on tree diameter (Phillips et al. 1998, Kronseder et al. 2012) but our finding that BA measured by a relascope is strongly related to field-measured BA is an important development. Compared to having fewer more expensive fixed area plots, plotless methods can work to directly estimate AGB in a large number of plots at a relatively low cost: it is relatively fast and a relascope can be made easily using local material or even, after calibration, one’s thumb. The sampling design should however be adapted for the purpose of biomass mapping and monitoring (see 5.4.2).

There are substantial differences when using the pan-tropical equations for AGB developed by Chave et al. (2005) using dbh and height or only dbh, as is known from the literature (Feldpausch et al. 2012). Results for our plots show that AGB calculated using dbh and height produces systematically lower values than AGB calculated using only dbh (details in Appendix B). This has potentially important consequences for the economic valuation of Mt Elgon in terms of carbon stocks and therefore for the benefits that can be derived from avoided deforestation and degradation or even reforestation under REDD+. Developing local allometric equations relating basal area and AGB would help reduce uncertainties, but the general consensus from the literature is that the use of the Chave equation with height - as is possible here because we measured tree height for a subset of trees - produces more accurate results (Chave et al. 2005, Feldpausch et al. 2012).

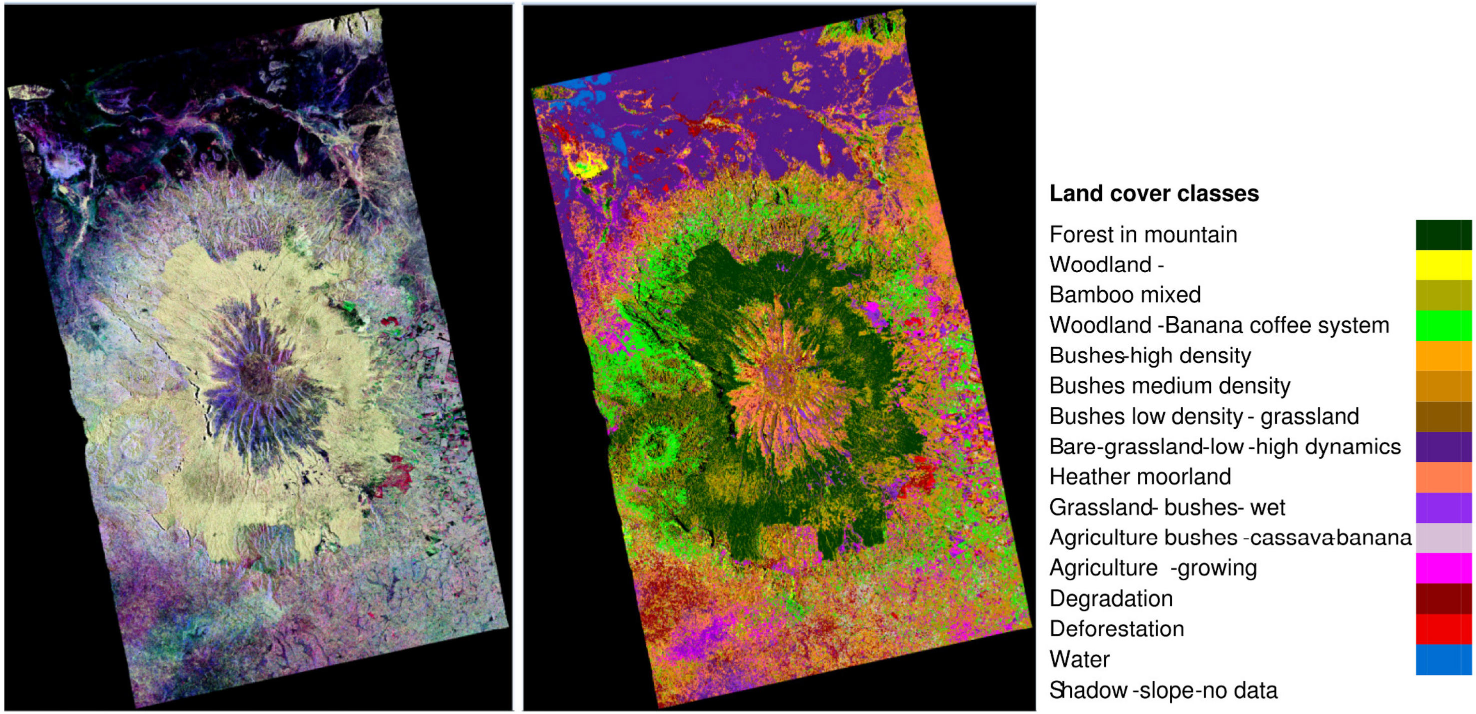

Figure 5.6. Composite of HV2007 (red) – HV2010 (green) – HH2010 (blue) and classified vegetation map of Mt Elgon 2007-2010.

5.4.2. Exploring the use of ALOS PALSAR data for biomass mapping in a complex east African forest

AGBgm estimated using the equation that included height versus HV backscatter, consistently gave a better fit than any of the other models. This is consistent with previous studies (Mitchard et al. 2009, Mitchard et al. 2011) and the study where the equations were derived, which found that errors were reduced by about half using the equations with height (Chave et al. 2005).

All three approaches found a strong relationship between HV backscatter and AGB in the 0-~150 or 200 Mg/ha range (Table 5.1). In this study the approach that used backscatter as the explanatory variable produced maps (Figure 5.5.a,b) with a higher contrast than the approach using backscatter as the explained variable. Given the relatively high uncertainty on plot-level AGB due to the use of the relascope, it was appropriate to try and minimize the error on AGB instead of on backscatter. The first model, including the zero AGBgm values, gave the better fit and thus better represented the data, which was shown by a higher r^2 and in the plots, even though it is less sensitive as shown in the map (Figure 5.5.b).

The radar backscatter saturates at higher biomass values (150-200 Mg/ha), limiting the possibilities to estimate biomass values above this threshold (Mitchard et al. 2011). This can be seen in Figures 5.3 and 5.5 where above this threshold we did not see any variation in backscatter, whereas AGB among plots still varied (Figure 5.3).

We were expecting slope, aspect and elevation to affect the relationship between AGBgm and backscatter. Despite the fact that the backscatter images where corrected for the physical terrain we were still expecting an effect of slope and aspect caused by differences in plant growth due to environmental conditions related to the terrain. But we may not have had sufficient plots to be able to detect this relationship, as it is probably weak and confounded by errors on the backscatter and AGBgm. Our results indicate that elevation (obtained through a Digital Elevation Model) potentially can contribute to a better prediction of AGB. This was not explored here as an equation including elevation would have only been applicable for a very limited spatial extent, only within elevation ranges covered by our field plots, and not over the whole landscape.

It is clear that our original terrain correction algorithms were not fully successful, as terrain influences on backscatter are clearly visible in the images. It appears that our terrain correction procedure was good at correcting for geometry shifts, but did not sufficiently correct for differences in brightness. The radiometric correction of the terrain corrected effects on slopes up to 15%. When slopes are very steep then radar has well know limitations (Ahmed et al. 2013). We need to create a local terrain slope layer, e.g. using a 15 m DEM for the study site and use this either to correct the radar data for backscatter values independently, or include as a layer in biomass prediction. We now used only a global DEM with 90 m resolution.

The biomass maps resulting from our derived equations of AGB against backscatter were not showing the patterns we had expected (Figure 5.6). Though they distinguished areas with broadly higher and lower biomass they did not show the forest structure patterns suggested from the classification image which encompasses all the variation in the radar data (Figure 5.6). Tree density and height are important aspects affecting biomass estimations per plot, but we had insufficient number of plots with data on tree density to investigate the relationship between backscatter and tree density.

We explored the use of a biomass - backscatter equation developed for another region (Eq. 3 for Guyana) where the same calibration procedures as in this study were applied but that was based on a backscatter - biomass relationship with a much broader range of values (13 dB). The result shows a higher differentiation between biomass levels (Figure 5.7).

$$AGB = e^{\left[\frac{\gamma^0_{HV}+26.836}{2.40244}\right]} \qquad \text{(Eq.3)}$$

The equation developed for Guyana produced patterns closer to what we had expected from the structural classification compared to our own results (Figure 5.5, 5.6 and 5.7). But, as expected, a plot of AGB values estimated from our plots using the Guyana equation compared to the AGB values we measured in the field showed strong bias (data not presented).

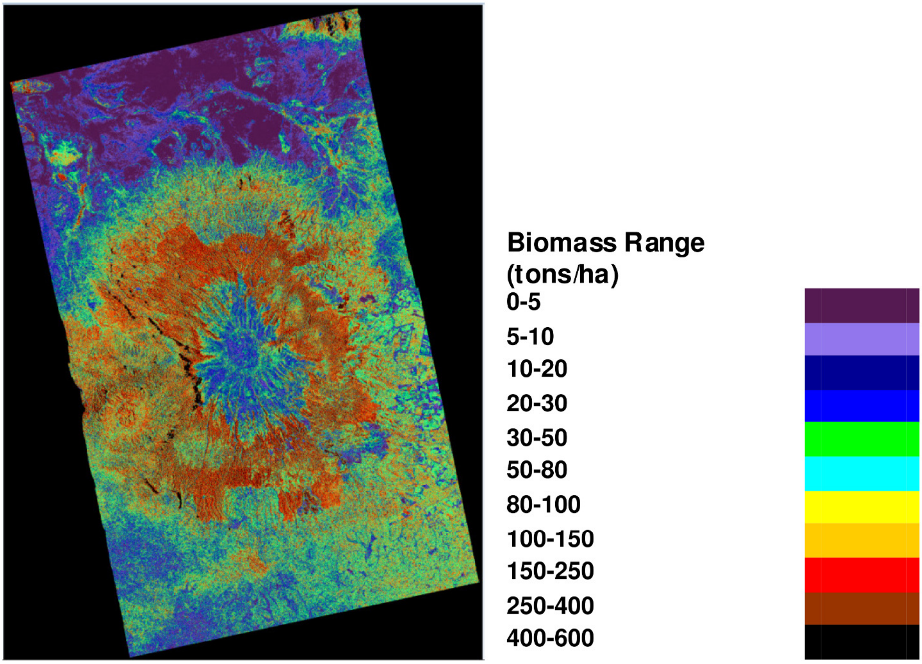

Figure 5.7: Above ground biomass map of Mt Elgon, Uganda for 2010, using the equation developed for tropical rainforest in Guyana (Quinones 2009).

Our sampling design was intended to assess the impacts of gradients of human use on the forest community (Chapter 3 and 4). We did not seek the greatest variation among (groups of) plots, neither did we try to keep forest structure within plots as homogenous as possible. The range in backscatter values among our plots was lower compared to that of other studies. We used relatively large areas (> 2 ha) to average pixel values for each plot, and it is possible that our plots did not adequately represent the forest of the surrounding 2 ha. It is likely that because we only sampled in a limited area of the study site, we covered only a comparatively small range of vegetation types (Figure 5.1).

Collecting data in more homogenous plots, to reduce variation in the pixels, with clearer differences in vegetation cover among the plots (instead of the gradient now covered) could perhaps contribute to reduce the error in the relationships with AGB. Smaller backscatter extraction areas should also be tested, although the issue of speckle from the radar signal would then increase. A wider range of vegetation types should also be sampled. Some unexplained variation in biomass was likely due to terrain and geolocation errors. The influence of using PALSAR data in an area with such steep terrain and corrected with a 90 m DEM should be assessed.

REDD+ projects would benefit from cost-effective consistent remote sensing observations and space borne biomass estimations to provide carbon levels, detect deforestation and degradation and monitor forest dynamics. A radar based monitoring system could contribute to this, but, as we show, there are still challenges to resolve. The results obtained by using different equations, based on different ranges in ground data-related backscatter values suggests that the relationship between AGB and backscatter is very sensitive to the type and range of ground data used. We therefore did not attempt to calculate a total amount of carbon for the area.

The models that we found with our data are valid only for the range of our data: the area around the mountain in Figure 5.5 shows little or no variation because we did not have any ground data for these areas. Radar technology is still in development however and perhaps new projects, such as the ESA mission using a polarimetric, interferometric P-band synthetic aperture radar will be able to overcome some of the challenges of ALOS PALSAR data [9]. However there is clearly good signal in this radar dataset, and possibly with a higher resolution DEM (for example the 10 m DEM's that are available from TanDEM-X) would have assisted in correcting the PALSAR data and improved the regressions.

In conclusion, we have shown that simple methods to estimate biomass in the field, such as with a relascope, can provide consistent data, and that ALOS PALSAR radar data can provide realistic classifications of the different vegetation types in a complex tropical mountain mosaic, differentiating well between different vegetation structural types. However, attempts to use the radar data in combination with field estimated AGB data to produce a biomass map had only limited success, probably due to limitations in the location and size of our field plots, and to remaining issues of terrain correction in our dataset. It may be that certain portions of this image had such large terrain artefacts that no correction is possible: this is a limitation in using radar data in mountainous areas. Using a radar-AGB calibration equation from Guyana appeared to produce a more realistic looking biomass map, but ultimately a higher resolution DEM is necessary to perform sufficiently good terrain corrections.

[9] http://www.esa.int/For_Media/Press_Releases/ESA_s_next_Earth_Explorer_satellite

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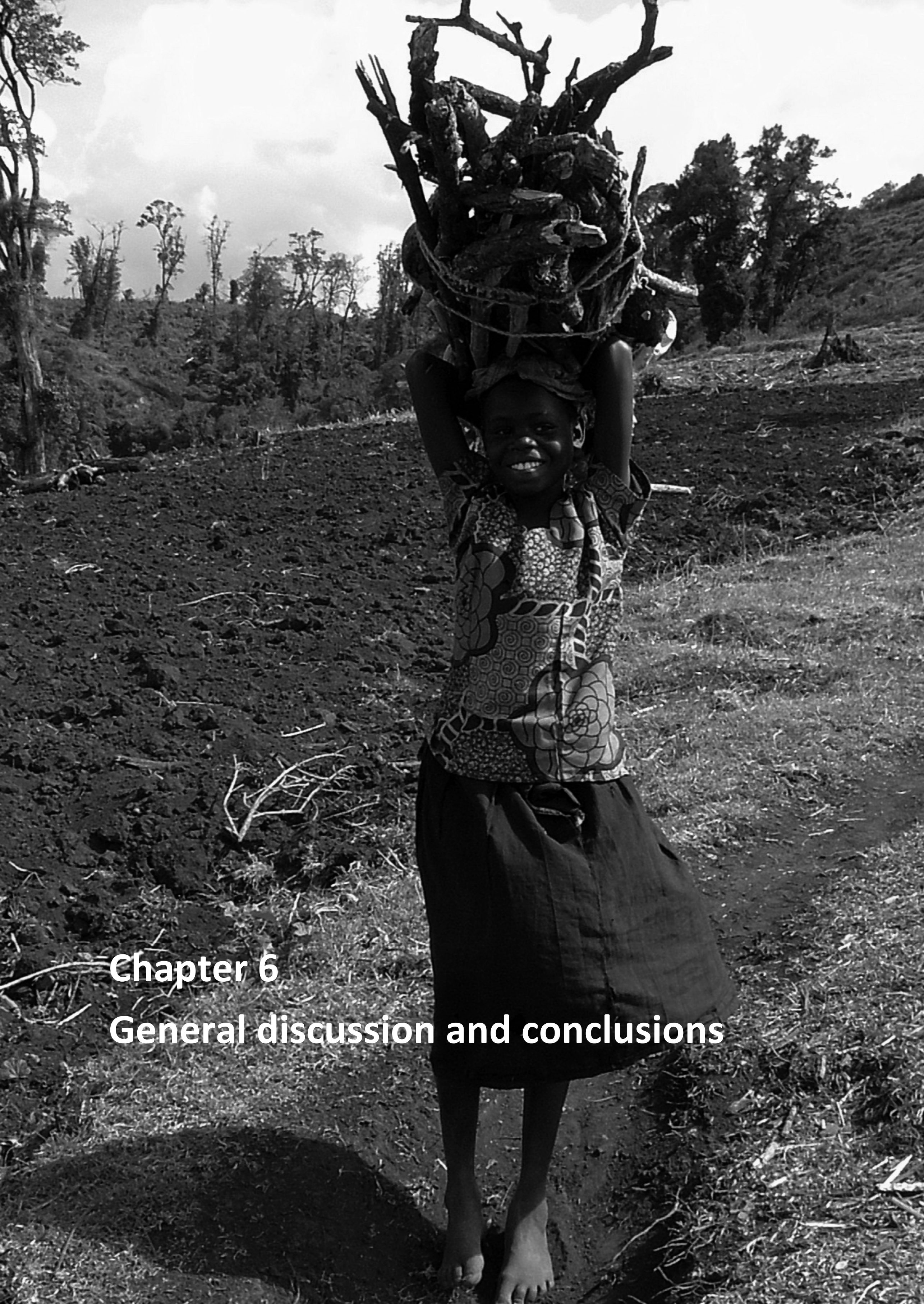

Chapter 6
General discussion and conclusions

The main goals of this study were to investigate how conflicting goals by different actors led to various outcomes for the forest and people on Mt Elgon, Uganda under different historical contexts and to explore lessons for the wider debate on conservation and development. Below I summarize the results before discussing their implications.

Mt Elgon did not experience only forest loss over the past 35 years. Locally, there were areas of recovery (Chapter 2). By analysing local variations I found that it is the *context* (e.g. law enforcement, collaborative management, political interference) under which drivers such as population, wealth, market access and commodity prices operate, rather than the drivers *per se*, that influences impacts on forest cover. Agricultural expansion on Mt Elgon cannot simply be linked to population and poverty or other individual drivers. This means that conservation and development interventions need to address local factors while recognizing influences operating at national and global levels. At the site level, local forest uses strongly influenced forest structure (Chapter 3). The type of resources collected and the impacts thereof varied according to the land use system outside the park (and sometimes inside). Human impacts also affected tree species richness. I show that generalisations about trade-offs between local uses and conservation are confounded by location specific characteristics. In the specific case of fuelwood collection, demand for fuelwood and the availability of alternatives on people's own land varied amongst the study sites and influenced fuelwood collection form the park. Dead wood was depleted on the edge of the park, particularly near the most densely populated sites (Chapter 4). Species that were highly preferred and used as fuelwood were depleted with possible impacts on tree biodiversity. Allowing the collection of fuelwood or other resources through collaborative management agreements creates opportunities for more destructive activities such as tree cutting for timber or charcoal (Chapter 3). On the other hand it helps relations between local people and park staff and is therefore a basis for further negotiation or improvement of collaborative management arrangements (see section 6.1.). Incentives to plant alternative sources of fuel on people's own land outside the park can help support more effective common pool management arrangements inside the park by helping to reduce the perceived importance of the forest as a source of fuel (Chapter 4). A relatively new approach attempting to reconcile local livelihood improvement and conservation on Mt Elgon involves PES schemes based on REDD+. I found that above ground biomass is very high in some areas of Mt Elgon, reaching above 800Mg/ ha (Chapter 5). Simple angle-count based methods can provide a cost effective method for AGB estimation, but the sampling design needs to be adapted to the purpose of biomass mapping. Topography and local degradation affected the production of a biomass map and further work is needed to address these challenges (Chapter 5).

The findings in Chapters 2, 3, and 4 constitute the first D and E of the DEED (Describe and Explain) framework presented in the introduction. Below, I explore (Explore) how the results of this study support theories on human-environment interactions that go beyond single factor relationships. In section 6.1. I discuss the importance of local motivations and how attitudes and decisions are shaped by changing contexts and conservation strategies. I then discuss generally recognised drivers of forest change from global theories on agricultural expansion and population (section 6.2.1.), to theories about the role of wealth and poverty (section 6.2.2.) and that of markets and prices (section 6.2.3.). For each of these I discuss how they in themselves do not explain forest change at the local scale. Rather, their importance is the result of the interaction of factors at different scales that determine the contexts under which local people make decisions on forest use. Finally, after considering the fine scale variation of impacts of local access and forest use (section 6.3.), I discuss possible options and implications for the design (Design) of more locally adapted and ecologically and socially sustainable management arrangements on Mt Elgon and elsewhere (section 6.4.). I conclude with recommendations for conservation actors at various levels on the need to recognise local variations in motivations, impacts and required attention.

6.1. Local decision-making and human-environment interactions (why local people do what they do)

Ultimately human-environment interactions are about people making decisions. In this study, I found that people's motivations are influenced by options available in a particular context, not necessarily by the drivers or factors themselves (Chapter 2 and 4). My findings support the theoretical framework proposed by VanWey et al. (2005) that various interacting factors determine the context within which actors make decisions and then mediate these decisions to lead to various environmental outcomes (see also Figure 1.1.): "Actors are decision makers trying to improve their well-being by choosing among productive options that appear available to them or when necessary, inventing new options". This study showed that this is true at multiple interacting scales. There is a need to understand how processes at different levels interact (VanWey et al. 2005) This study on Mt Elgon gave empirical evidence of how such interactions work and their complexities. I show the importance of the interplay and feedback between context, drivers and of differing priorities among groups of people and how this leads to varying outcomes for forest conservation (see also Leach 2008, Hersperger et al. 2010, van Noordwijk et al. 2011).

Since its first gazettment as a forest reserve by colonial powers, local people on Mt Elgon progressively lost access to forest land and resources (Turyahabwe and Banana 2008). Before the breakdown in law enforcement under Idi Amin's rule, the forest reserve boundaries appeared relatively intact (Chapter 2). There was however already a history of excision and de-gazettment under colonial rule, particularly in the southern area (Scott 1998). Also, there were fewer roads all around the mountain, urban markets were likely smaller and demand for crops and forest resources such as fuelwood and timber was likely lower. I show that when law enforcement was reinstated from 1993, this played an important role in maintaining park boundaries but was only successful over the longer term in certain areas (Chapter 2). I discuss some reasons for this under section 6.2.3.

However, I also found indications that boundaries were more resilient in areas with collaborative management or/and low conflicts (Chapter 2) (Persha et al. 2011, Porter-Bolland et al. 2011). The attitude of local communities towards the park and its management seemed crucial in determining the level of conflict and the adoption of collaborative management, although the cause and effect relationship was not always clear and would merit further investigation (see also Struhsaker et al. 2005). In Burma, Allendorf et al. (2006) found that positive attitudes towards protected areas were related to perceived benefits from conservation and from managing the area, from extraction, and lower conflicts with park staff. On Mt Elgon, I observed different types of attitudes (from south to north-east):

- Rejection of the boundary-line, agricultural encroachment, violence, back-and-forth of clearing and regrowth before and after elections, degraded forest due to charcoal burning (Chapter 2, 3, 5) (Figure 6.1).
- No agricultural encroachment but high and sometimes illegal forest use under collaborative management. Forest cover relatively well maintained or kept in an early succession state (Chapters 2, 3 and 4) (Figure 6.2).
- Rejection of the boundary-line set by resettlement, encroachment in the contested area but beyond that mainly through grazing and firewood cutting and not so much violence. Forest cover relatively intact inside the claimed boundary but apparently gradually shifting back. Forest structure affected by grazing (Chapter 2 and 3) (Figure 6.3).
- Boundary-line recognized but more recent degradation on the edges (Chapter 2) (Figure 6.4)

Figure 6.1. Fires on the edge of a formerly encroached (1990s), then recovered and now (2010) re-encroached forest area in the south. The burning is to clear land for agriculture. The cultivated area shown in this picture is inside the official boundary and under formal dispute in the Courts (photo 2010).

Figure 6.2. Edge between cultivated area outside the park boundary and regeneration from former encroachment (1990s) in the north-west. Tree regeneration is slowed-down because of small-stem harvesting (photo 2010).

Figure 6.3. Formerly cleared forest edge in the north (after 2001), now grazed by cattle (photo 2011).

Figure 6.4. Almost intact forest edge in the north-east. A white boundary pillar is just visible towards the right of the picture (photo 2010).

Different histories and a combination of the factors and contexts seem to have influenced these variations (see also section 6.2.). For example, in the north corruption in land allocation processes related to resettlement (even recent) and boundary demarcation fosters resentment and powerlessness among poor people on the forest edge. Local people reported that this influenced their behaviour. For example in a study site in the east, local people told me that when the park was established in 1993, they lost their sense of ownership over the forest and people started encroaching the edges soon thereafter (see also Scott 1998). In the south people said that UWA could try and stop them from getting the resources they needed but that they would still go and “sneak in at night or when the rangers are not aware” (anonymous, personal communication) (see also Norgrove and Hulme 2006).

Conflicts between forest management and local communities over the use of forest resources tend to be bad for conservation. Resolving the conflicts needs political will, because it seems that on Mt Elgon a large part of the conflict is due to political manipulation (local people, personal communication and various UWA staff, from ranger to Conservation Area Manager and Banana et al. 2010). Many studies focus on the negative perceptions local people hold against conservation areas, but during this study, most people in community and individual formal and informal interviews also spoke out about the positive aspects of the park (unpublished data). Even in areas with conflicts and encroachment some stated that the park was important to protect the forest from encroachment and that without it they would have cleared much more of it. They considered the forest important for water provision and climate regulation that benefit agriculture but also for cultural values (e.g. “the forest provides rainfall for farming, traditional medicine and a home for animals”, “I was born there”, “circumcision ceremonies used to be done in the forest”, “it is the home of our forefathers” (unpublished data from an exercise where people were asked to rate the relative importance of the forest compared to other lands). Conservation management should work towards fostering and building upon such positive values.

6.2. Drivers of forest change, the importance of local histories, contexts and linking scales

6.2.1. Agricultural expansion and population

Globally, conversion to crop- and pastureland is the main direct cause of deforestation (Kaimowitz and Angelsen 1998). In single-factor theories of drivers of deforestation, agricultural expansion is directly caused by population or poverty (Malthus 1873, Ehrlich 1968, Carr 2004). On Mt Elgon I found that agricultural encroachment is indeed an important source of forest loss, aided by grazing in some areas (Chapter 2 and 3). Although clearing of native forest to establish plantation areas that were never fully stocked also played an important role in some places (Petursson et al. 2012). Population densities around the park were high (150p/km^2 to more than 1000 p/km^2) and growing fast (2.5-4.3% per year) (UBOS 2002a, b, d). Yet, I found no simple direct relationship between population (growth) and deforestation on Mt Elgon. Population drove deforestation only at a certain time and under certain conditions: when there were no boundaries to expansion In fact, when conditions were favourable, forest recovery took place near some of the most densely populated areas on Mt Elgon (Chapter 2). I discuss possible reasons below.

According to Boserup (1965), an increase in population density leading to land scarcity stimulates agricultural intensification. The Bagisu on the western slopes may well have experienced such an evolution. They have been settled for centuries and have slowly moved up the mountain and cleared the forest around them as population grew: the fertile volcanic soils were able to sustain and thus likely also facilitated the continued increase in population over time. When the forest reserve was established by the colonial government in the 1920s their sedentary agricultural traditions (and high population) made it possible for them to adapt and intensify agricultural production on existing land, likely helped by the success of coffee as a cash crop (see below), which was introduced in 1910 (Bunker 1987). However, in the northern area of Mt Elgon - dominated by the formerly pastoral Sabiny - such intensification has not (yet) taken place. VanWey et al. (2005) propose that “Some groups may be faced with such rapid changes in population and scarcity of resources that they cannot adjust fast enough”. On Mt Elgon, it is possible that the increase in population and change in land use systems in the north has been so rapid (since the 1980s) that people have not been able to adapt fast enough and create a land use system that is more diversified in terms of addressing nutritional and energy needs (less variation in crops, little firewood, in particular closer to the forest edge). Or perhaps they still manage to access sufficient forest resources to compensate for this and therefore have insufficient motivations to intensify. Further research is required to understand this better and how such insight can help inform better management.

6.2.2. Poverty or wealth and deforestation

The connection between wealth or poverty and deforestation is complex. Some argue it is wealth that enables and motivates people to clear the forest, while according to others poverty leads to dependence on forest resources and thus to deforestation (Moran and Ostrom 2005). On Elgon, wealth seems to have motivated people to both clear the forest and leave it to regenerate inside the protected area, depending on the context provided by other factors (Chapter 2). Wealth enabled people to intensify agriculture at times when expansion was restricted due to institutional barriers: while there was a forest reserve before the 1970s and a national park after 1993. However, wealth also enabled them to rapidly expand when boundaries broke down in between these periods (Chapter 2). I found that from the 2000s new forces came-in and confounded these earlier patterns, such as increased market access for seasonal crops, and conflicts with park management (see section 6.2.3).

6.2.3. The role of markets and market access

Access to markets and crop prices are seen as major underlying causes of land use change leading to deforestation (Geist and Lambin 2002). But here too context and history influence people's motivations and decisions. On Mt Elgon, market liberalization enabled coffee farmers to take advantage of booming world prices. But contrary to other studies (O'Brien and Kinnaird 2003, Gaveau et al. 2009) these price increases and the associated increase in wealth did not always lead to exacerbated deforestation (see section 6.2.2). Access to markets plays a role here too: even if people have the means to invest in expanding agriculture, they may not do so because of a lack of physical access to markets. This may also explain the success of coffee on Mt Elgon: access was historically difficult and coffee is relatively easy to transport on foot in the amounts produced by smallholder farmers, especially compared to other important crops such as maize or bananas. In fact, various studies have found that the types of crops people plant are "strongly determined by transportation costs and that changes in external market value or cost of transport are reflected relatively rapidly in the spatial allocation of agricultural activities" (VanWey et al. 2005 citing Muller 1973, O'Kelly and Bryan 1996 and reflecting von Thünen's theory (1826)). Angelsen et al. (1999) also found that prices for seasonal crops in particular were associated with deforestation. We see three situations related to these theories on Mt Elgon:

- On the western slopes where access was still difficult at the time of this study, largely due to the terrain and climate, coffee remains relatively more important than other potential cash crops (Table 1 in Chapter 3). In their choice of a cash crop, this situation makes it more sensible for people to focus on a crop that is relatively easy to transport to markets downhill than on crops with a lower profit to volume ratio such as bananas. Here transportation costs determine people's focus on coffee (see VanWey et al. 2005). Other cash crops such as cabbages fall somewhere in the middle in this balance of cost versus profit in terms of transport and increase access may mean increased pressure to grow these crops, especially if coffee prices go down again.

- In the south coffee-based villages diversified away from coffee when they gained better access to markets (2000s). Relatively bulky cash crops such as onions and cabbages, but also maize, fetched high prices due to shortages elsewhere (maize in Kenya and Sudan, see Chapter 2). Terrain is less steep and roads are better which reduces transport costs so people choose to focus on these high return seasonal crops (see VanWey et al. 2005) (Figure 6.5.). As the returns from these relatively short-season crops likely compensated sufficiently for the risk of eviction by the park management (Chapter 2), renewed forest clearing took place (Angelsen et al. 1999).
- In the more recently settled North, coffee was not introduced on the higher slopes because this was still forest until the early 1980s but people were encouraged to grow maize and potatoes (which people already grew traditionally on small scale in the glades inside the forest) and later wheat as food and cash crops. New roads that have increased access to markets in Uganda and Kenya since the 1990s, reducing transport costs and therefore confirming people's choice of crops (see VanWey et al. 2005). At the same time high maize prices in Kenya (in 2008-2009) made these profitable crops, which led to some forest clearing inside the park (Angelsen et al. 1999) (Chapter 2).

6.3. Human impacts and conservation on Mt Elgon

I found that impacts of local use inside the park extended far into the forest (> 2 km). Impacts of the harvesting of stems, fuelwood collection and charcoal production on forest structure were important (Chapter 2 and 3), especially nearer the boundary. Gaps in the canopy due to timber cutting or charcoal making created further inside the boundary can likely recover relatively rapidly if these activities are stopped because they are surrounded by a more intact matrix (Chazdon 2003), but in formerly encroached and re-encroached areas this will likely take more time as dense shrubs and ferns often dominate. These also burn easily killing any tree regeneration. Similarly, in intensively grazed areas, soils may be impacted and regeneration may be slower (Reed and Clokie 2000). Some historical modifications from human use on Mt Elgon very likely contributed to its biodiversity values, such in the semi-natural glades that are found inside the forest and the moorlands above the treeline (Reed and Clokie 2000). Also, some of the plant communities on the higher moorland may be dependent on fire (Wesche et al. 2000).

Resource use agreements and low conflict did not necessarily lead to sustainable local forest use: I found that impacts of local uses in the forest were high even in the study site with a resource use agreement and low conflict with park management. Resource use agreements include the monitoring of resource off-take by a local resource use committee but this was apparently not entirely effective (Chapter 3). Only one of the research sites studied in more detail had a collaborative management agreement which prevented us from testing statistically any differences between areas with and without agreements in terms of impacts of local use. It is important to further investigate the functioning of existing resource agreements on Mt Elgon and the motivations of local people to commit (or not) to the rights and obligations that they involve (Ostrom 1999) (see also section 6.4.4.).

Depletion of fuelwood and other wood resources is a concern in forest that is surrounded by a dense population (Chapter 3 and 4). Enforcing boundaries to prevent people from accessing for example fuelwood is almost impossible in practice, although impacts were high. Likewise, the harvesting of stems to serve as stakes for crops such as beans, banana and even coffee bushes with fruit-heavy branches had impacts on forest structure and composition. Species harvested for this purpose (e.g. *Neoboutonia macrocalyx* or shrubs such as *Vernonia spp.*) either coppice or regenerate easily (Katende et al. 2000). A study on the sustainability levels for the harvest of coppices of fast growing pioneer species in regeneration areas near the boundary could help find ways to accommodate local preferences and contribute to fostering good relations with the park. This study indicates that in order to design sustainable management strategies, it is important to understand the impacts of local forest use on the conservation values that the park is meant to preserve.

6.4. Future options, balancing conservation and development

In the previous sections, I analysed and discussed the drivers of forest change and impacts of local uses on the forest community under various interacting contexts and local factors. I now explore a number of future options and scenarios for more ecologically and socially sustainable management on Mt Elgon.

6.4.1. Conservation and development

Designing conservation management requires making choices between conservation values such as biodiversity, water catchment or other ecosystem services and other values, such as the priorities of local communities. Especially in places of high population density, which often have highly fertile soils (that were able to support these populations in the first place), the pressure on remaining forest areas is high (Naughton-Treves et al. 2007). Mt Elgon provides a prominent example. In such places it is impossible to achieve effective conservation based purely on strict law enforcement and exclude local people completely.

The ability of biodiversity conservation to contribute to poverty alleviation and development is disputed (Adams et al. 2004). But most will agree that conservation should not make people poorer, both from an ethical point of view but also practically as increasing the number of poor will lead to an increase in pressure on resources (Hutton and Leader-Williams 2003, Minteer and Miller 2011). Yet, by definition (Dudley 2008) protected areas entail restricted or at least controlled access to land and resources, which then affects forest dependent neighbouring communities in one way or the other. This is also the case on Mt Elgon, as several studies found (Norgrove 2002, Katto 2004, Namugwanya 2004, Gosalamang et al. 2008). The impact of local income generating activities through tourism or employment as rangers, compensation for conservation schemes, etc. is often limited (Agrawal and Redford 2006, Brooks et al. 2006). This is the case even where communities benefit from tourism activities, such as in forest parks harbouring charismatic species like gorillas, let alone in a protected area such as Mt Elgon that sees only about 5000 visitors per year.

6.4.1. Alternative resources for land: Intensification of agriculture outside the park?

According to the Borlaug hypothesis, agricultural intensification (due to technological change, such as during the green revolution) will lead to decreased demand for land and thus decreased deforestation; whereas others argue that increased yields and higher profits will lead to greater incentives for expansion (review in Angelsen and Kaimowitz 2001). But contexts matter. For example, in Ecuador, coffee cultivation in a context of labour constraints has led to less deforestation on farms. People continue to grow coffee even if it does “not provide the highest immediate income. Coffee has, however, a guaranteed market and low transportation costs and is important for farmers’ long-term income security” (Angelsen and Kaimowitz 2001). On Mt Elgon I found indications for both scenarios in the densely populated coffee-producing areas of the west and south-west (enough labour but little land), depending on contexts. When conflicts were low, collaborative management agreements were in place and the park boundaries accepted, people seemed to intensify production on their own land or find alternative sources of income (not studied here). In areas with conflicts and increased market access the risks ran by expanding into the park were apparently compensated by the potential returns (see section 6.2.3) (Figures 6.1 and 6.5).

Figure 6.5. Maize cultivation and fields prepared for onions near the park edge in the south of Mt Elgon (photo 2009).

Figure 6.6. Landslide on the edge of a formerly encroached and now grazed area of the park in the north (photo 2011).

In the north of Mt Elgon, coffee was not grown on the higher slopes close to the park where land was more abundant and the population less dense than in the west and south-west (Chapter 2). People used oxen ploughing because they have easier terrain, which makes it practical to grow seasonal crops such as potatoes, maize, wheat with very few trees in and around their fields. Little inputs were used and people probably preferred to save on labour than on land (Angelsen and Kaimowitz 2001). Erosion on the seasonally bare slopes was clearly visible (Figure 6.6) whilst more trees or permanent crops on people's land may help address both this and provide alternative sources of wood. The conditions and incentives that would encourage farmers in the north of Mt Elgon to intensify whilst integrating more trees into their land-use system should be explored.

6.4.2. Alternative resources for forest products: agroforestry?

The importance of integrating improved livelihoods and biodiversity conservation goals in a larger landscape was shown particularly in with Chapters 3 and 4, where the importance of alternative resources outside the park became apparent. Trees on farms, such as in agroforestry systems, help maintain or restore the environment for agriculture (and can be seen as a remnant of shifting cultivation practices where regenerating forest vegetation helped restore soil fertility. Trees on farms also provide numerous products: food, fuel, construction materials, fodder, mulch etc. and help risk management (e.g. trees as a source of income in case of crop failure) (Arnold and Dewees 1997). Historically, access to traditional sources of tree-based resources has decreased because of exclusion of local uses from protected areas or because of physical changes (degradation, deforestation). Farmers then started to protect, plant and manage the resources they found important on their own land. Expanding markets for tree products (fuelwood and others) has accelerated this (Arnold and Dewees 1997). In many places this meant that the density of planted trees increased over time as a general trend (Tiifen et al. 1993, Fairhead and Leach 1996, Arnold and Dewees 1997).

Patterns of trees on farms vary with agro ecological, economic and other contexts. It only makes sense when the land and capital to do so are available and when the trees provide higher (perceived) benefits than alternative uses of land and capital. In Kenya it was found that older households keep more trees as woodlots, because they need less labour than crops, but younger households tend to replace trees with crops such as tea of coffee as these are more lucrative. Having trees on land can also free up labour so that people can have jobs elsewhere (Arnold and Dewees 1997). On Mt Elgon shade trees in the coffee-systems complement or contribute to crop outputs: they do not compete with them (Figure 6.7). When possible, existing systems should be built upon and sufficient incentives created for people to maintain them (Soini 2006).

Figure 6.7. Agroforestry system on the western slopes of Mt Elgon, with coffee, trees, and banana (photo 2009).

6.4.3. Likely future economic change and alternative sources of income?

The growth in the demand for certified products, in particular coffee, provides opportunities for farmers to earn a premium on their product. Mt Elgon had/has a strong reputation for good quality Arabica coffee that could be built upon. The potential for marketing as a fair-trade "specialty" coffee is high and already exploited by some buyers (Ponte and Kawuma 2003). Certification for forest-conservation-friendly coffee should be explored for Mt Elgon as a way to encourage and support farmers to invest in sustainable coffee production and make the risks of cultivating seasonal crops inside the park less worthwhile (Chapter 2).

Growing accessibility followed by the development of trade-centres creates opportunities for increased wealth through other trade such as shops and restaurants etc. I observed change just within a year coming back to the four sites: the villages had more shops and the nearest "trade centre" had grown. But growing trade-centres also can lead to higher demand for wood resources from businesses such as the numerous tea and chapatti places as well as small restaurants. This likely also leads to increased demand for charcoal instead of fuelwood by business people with more cash (Angelsen and Wunder 2003, Arnold et al. 2003). Increased rural-urban migration of people in search of new opportunities outside agriculture may help alleviate pressure on the forest from neighbouring communities, but this may be compensated by increased demands for wood products from urban areas (DeFries et al. 2010).

Alternative sources of income can also come from payments for environmental services such as in the context of REDD+ schemes (Leimona et al. 2009). On Mt Elgon the MERECP project is piloting a number of REDD+ related initiatives that involve incentives for tree planting outside the park as well as restoration and protection inside the protected area (LVBC 2009). For such schemes to work on Mt Elgon and in light of findings of this study, they will have to provide enough incentive to compensate for (perceived) losses from cropping either on people's own land or in the park. This is likely not a purely economic balance as people also value forest and trees for other things than tree products or the land that they stand (see section 6.1.). Carbon-related projects that involve local communities around forests are common pool resource situations that are embedded in wider, national and international contexts; through country-level REDD policies and international carbon markets. Outcomes will therefore depend on the interactions of these contexts with local factors - as shown in Chapters 2, 3 and 4 - and how these are addressed within such projects.

On Mt Elgon, benefits from the MERECP project depend on whether one belongs to a local organisation (a registered community based organisation) that has been selected to participate in the project, and so it excludes a large section of the park neighbours (Nel and Hill 2013). It is not clear how others will benefit in the long run, which represents a risk to the long term success of this scheme: it has been shown that successful common property resource management systems need to be based on inclusive and adaptive institutions that are locally relevant, both socially and ecologically, and take into account local drivers for land use change (Reynolds 2012, Shames et al. 2012). Like other common pool resource management situations, carbon-related projects also need to ensure that benefits to local communities are maintained on the longer term and that they are not captured by the elite or the state (Leimona et al. 2009, Pollini 2009, Persha et al. 2011, Nel and Hill 2013).

6.4.4. Conditions for management with better outcomes for forest and people

Powerful economic and political forces at different levels drive people's actions. Conservation must work with local people to find a realistic balance between conservation goals and local needs and demands for resources that support their livelihoods (Kaimowitz and Sheil 2007). There is a need to assess realistic conservation values against the needs of local people. This may mean accepting "lower standards" than would be dictated by traditional conservation of untouched “habitats”, but possibly sufficient to maintain essential ecosystems services and biodiversity (see section 1.1.2.), while meeting local needs (in terms of resources and empowerment) sufficiently to help foster more positive attitudes towards protected areas. Giving people access to the forest for a limited number of resources can help foster such positive attitudes, although this depends on people’s perception of the costs versus the benefits they derive from such arrangements (Ostrom and Nagendra 2006). There also lies a challenge in that conservation actors and local communities typically operate under different time horizons (long versus shorter). A middle-ground needs to be found.

There is a need to experiment with and learn from a wider range of institutional arrangements and interventions (Game et al. 2013). External conservation organisation and funders need to support such experimentation. In the global debate on drivers of forest change and in discussions about devolved forest management, local use of forest resources and payments for environmental services to local communities, models leading to “simplified institutional prescriptions” do not reflect complex local social and ecological realities (Ostrom and Cox 2010). Conservation actors must build strategies based on fostering existing positive local attitudes for conservation and the perceived benefits of retaining forest. The incentives for people to create alternative resources on their own lands, for example, need to be further studied and built upon. Congruence of local ecological and cultural contexts, together with perceived benefits and costs affect the success and sustainability of natural resource management systems more than the particular form of ownership (Ostrom and Nagendra 2006). Policies and institutions should be therefore built in a dynamic process of inclusive decision-making, implementation and evaluation. Further investigation is required into the local conditions and incentives that would support the development of sustainable common-pool resource institutions on Mt Elgon and elsewhere, based for example on the *Design Principles* defined by Ostrom (1999).

This study shows that law enforcement is important to avoid deforestation, and perhaps limit degradation, but it should go hand-in-hand with developing capacity for collaborative management and monitoring, and for sharing responsibilities and rights between park management and local communities. On Mt Elgon and elsewhere, politicians need to stop using people and the park for their personal gain (Banana et al. 2010). This would benefit local people in the longer term and forest conservation too. There lies a role for the central government in setting the example and establishing rules of conduct. Park management authorities need to engage in building relationships of trust with local communities, either through collaborative management agreements or by other means, acknowledging local needs, and displaying consistent attitudes with regard to rules, rights and responsibilities (Hough 1988). Without trust, it is impossible to resolve long standing conflicts and build sustainable and effective management arrangements.

Finally, low-cost and rapid monitoring systems are essential to manage forests used by people in areas with limited financial resources for conservation. To monitor the local impacts of forest use, assessments by local people can be used but for the analysis of more heterogeneous patterns on larger scales such as needed in REDD+ projects, more formal assessments are necessary (Nagendra and Ostrom 2011). In an ecosystem such as on Mt Elgon, which is complex both from a human and a topographic point of view, quick and simple field methods in combination with radar imagery offers potential for establishing baselines and monitoring change, but a number of methodological issues still need to be resolved (Chapter 5).

6.5. Conservation and livelihoods in crowded places: conclusions

Competition over land and resource is exacerbated in areas of (extremely) high population density such as on Mt Elgon. A wide range of factors are involved at different levels creating pressures originating at multiple interacting levels. Mt Elgon, as an "island of tropical forest in a sea of agriculture and people" provides an illustrative case for other protected areas in this situation. My findings contribute to the evidence that simple models based on only a few drivers of deforestation (i.e. population and poverty) cannot explain local variation (Lambin et al. 2001). Conservation and development actors need to acknowledge the complexity and dynamics of underlying contexts and drivers of forest change (Putz and Romero 2012). Interventions need to address local factors while recognizing influences operating at national and global levels. Trade-offs between local uses and conservation are confounded by location specific characteristics, calling for actions that take into account these variations. Forest management that empowers local people combined with incentives to develop alternative resources, such as trees for various uses, including fuelwood, can support more sustainable forest management for both people and conservation. In a situation of historical conflicts such as on Mt Elgon, management authorities need to make a substantial effort to build trust with local communities, and show genuine acknowledgement of local needs.

Existing theories of change provide useful frameworks for studies of local variations in outcomes for forest conservation, while at the same time providing insights that can help inform the wider conservation and development debate. It is clear that there is a need to create support and new opportunities for conservation (and development) strategies that meet local needs and build on existing positive values people have for the forest and its protection. Both international conservation actors, which often have a strong influence on national policies and resource allocation for conservation and development, as well as national and local forest management authorities (e.g. Uganda Wildlife Authority) need to recognise that incentives that influence people's motivation for action in relation to the management of common pool resources vary locally and can therefore not be designed globally.

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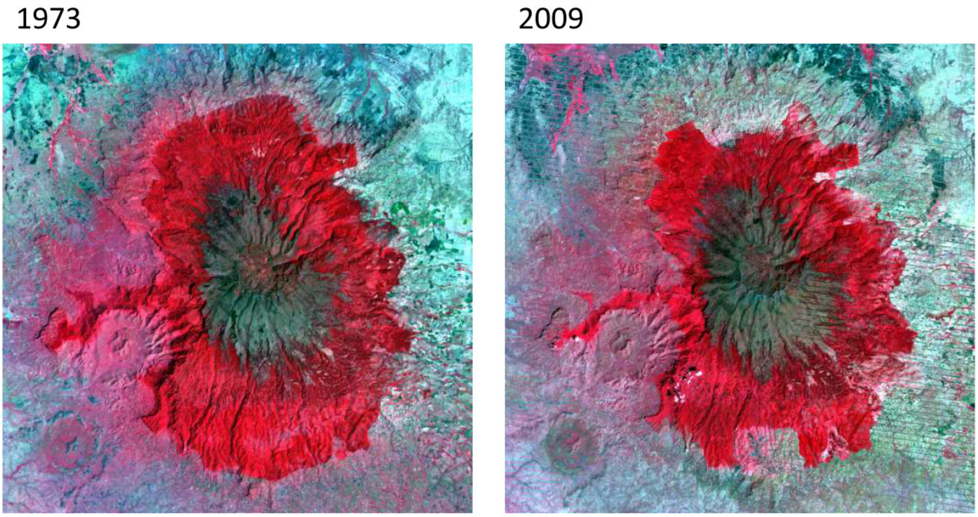
Appendices

Appendix 2.A. False color composites of Landsat images at the beginning and end of the study period: MSS (band 4,3,1) for 1973 and ETM+ (4,3,2) for 2009.



These raw dry season images show the generally sharp edges between forest and agricultural areas. These correspond to the boundary of the protected area (see also Figure 2.1). Forest is dark red while cultivated land is pale blue-green in the drier areas or reddish-pink on the wetter mountain slopes where coffee and bananas create a more permanent vegetation cover. Green-brown colors on top of the mountain correspond to the higher altitude moorlands and crater.

Appendix 2.B. Forest cover classification and evaluation, detailed methods

Data

Details of the Landsat satellite images used to create four forest cover maps for Mt Elgon are found in Table 2.B.1. They were taken during the second half of the dry season (January-February) to minimize cloud cover and when differences between evergreen vegetation and seasonal vegetation are greatest (Table 2.B.1).

Table 2.B.1. Satellite imagery used for classification in the study.

Date	Satellite and sensor	Path/Row	Nominal spatial resolution
01/02/1973	Landsat 1 MSS	182/059	57m
02/02/1973	Landsat 1 MSS	183/059	57m
01/02/1973	Landsat 1 MSS	182/060	57m
18/02/1988	Landsat 4 TM	170/059	28.5m
05/02/2001	Landsat 7 ETM+	170/059	28.5m
25/02/2008	Landsat 7 ETM+	170/059	30m
10/01/2009	Landsat 7 ETM+	170/059	30m

We georeferenced and stitched together 6 topographic map sheets (1:50.000) based on interpretation of 1959 and 1960 aerial photographs followed by field revisions (1967) by the Uganda Department of Lands and Surveys (Department of Lands and Surveys 1967). From the resulting map we then digitized the boundaries of forest and other vegetation cover on Mt Elgon. We also used the existing Land Unit Map of Mt Elgon by van Heist (1994), which was based on the interpretation of 1959 and 1989-1990 aerial photographs, SPOT images from 1991 and 1992, and extensive fieldwork. Additional reference data was provided by a 90m digital elevation model (Jarvis et al. 2008) and field observations from 2009 and 2010. These were limited due to logistical constraints.

Pre-processing

All satellite images were processed and analysed using ENVI 4.0 (RSI) software. The four images were already geometrically and radiometrically corrected. For 1973, 3 images had to be stitched together to cover the study area, while for 2009, an image from 2008 was used to fill gaps that are characteristic of Landsat 7 since 2003. The resulting 1973 and 2009 images were resampled to match the 28,5m pixel size and each image was co-registered to that of 2001 using Nearest Neighbour resampling. The 1988 image was registered to the 2001 image using 159 control points and a 2nd degree polynomial warp with a root mean square error (RMSE) of 0.33 pixels. For the1973 image we used 80 points with an RMS error of 0.25 pixels and a 3rd degree polynomial warp. Registration was verified visually by overlaying registered images (Schowengerdt 1997). The 2009 image was provided already registered to the 2001 image; this was visually checked and found to be accurate. A subsection of 2775 by 2775 pixels, encompassing Mt Elgon and its direct surroundings was extracted from all images.

Cloud masking

The 1988 and 2001 images had substantial cloud and cloud shadow cover at the higher altitudes, the 2009 image only a few dots. Clouds were masked out on the 1988 and 2001 images using visually estimated DN values in bands 1 and 6 (see Martinuzzi et al. 2007). Resulting masks were intersected and buffered. For cloud shadows the most effective method was found to be visual determination of training areas on a transformed image using principal component analysis. The result of the supervised maximum likelihood classification was manually corrected in ArcView and the resultant final mask buffered using a three-pixel buffer to include any "borderline" pixels. Finally the cloud and cloud shadow masks were combined into one mask for each image. Clouds on the 2009 image were digitized on screen.

Classification

The classification focused on the Afromontane and the Afromontane Rain Forest Zones inside the protected area boundaries, as defined by van Heist (1994). First, an unsupervised classification using the Iterative Self Organizing Data Analysis (ISODATA) helped identify natural spectral clusters (Schowengerdt 1997). Training areas were then identified using visual interpretation of the natural clusters from the unsupervised classification, false colour composites of the images and calculated band ratios (to minimize the effects of shade). Areas of shade were given separate training areas from illuminated areas with the same vegetation. Ancillary information composed of the topographic and vegetation maps aided interpretation of vegetation classes for the 1973 and 1988 images (Foody and Hill 1996). Spectral separability of the classes was investigated with the help of graphical displays and statistical analysis using the Jeffries-Matusita (JM) distance (Schowengerdt 1997). Classes with separability values lower than 1.9 (with a maximum of 2) were re-examined and refined or merged.

Finally a supervised classification was run using a maximum likelihood classifier (Schowengerdt 1997). The classes resulting from classification were first combined into 4-7 main classes per image (forest, bamboo, grassland, thicket, moorlands, burn scars (on moorland) and water), with the help of spectral plots, maps and Google Earth images (2003) then into two: "forest" (forest and bamboo, minimum canopy cover of 30%)) and "non-forest" (the other classes). Those classes were sieved, clumped and a 3x3 pixel majority filter was applied to remove isolated pixels and improve spatial coherence. We checked the resulting classifications on screen and when necessary edited areas of strong topographic gradients (cliffs), haze and pixel errors resulting from image mosaicking (1973 and 2009 images). For this we used the reference materials listed in the accuracy assessment section below. All gaps caused by clouds on the 2009 imager could be filled as they occurred only over higher altitude forest with little change. Remaining areas of cloud cover were excluded from the analysis for each affected period (1973-1988, 1988-2001, 2001-2009). One of the study villages is located next to a softwood plantation area inside the park (Figure 1.1). This area is managed by the park authorities and has been labelled “non-forest” for the purposes of this study. The resulting maps were exported to ArcGIS 10 (ESRI) where accuracy assessments were performed.

Accuracy assessment

We used post-classification comparison to detect changes between dates (Mas 1999, Song et al. 2001). With this method, high accuracies of the individual classifications are needed because the accuracy of the change analysis is affected by the accumulation of errors from each classification involved in the comparison (Singh 1989). The digitized forest cover map of 1967 and the Land Unit Map of Mt Elgon by van Heist (1994) provided a reference to validate the accuracy of the 1973 and 1988 forest cover maps respectively. We allocated forest or non-forest classes to 165 and 152 points on the 1973 and 1988 forest cover maps respectively and on their 1967 and 1990 reference maps. We verified and corrected reference point label allocations for the 1967 and 1994 maps when known changes had occurred between the map production date and that of our satellite imagery (e.g. new plantation areas). The 2001 and 2009 forest cover maps were assigned 139 and 168 points respectively. We did not have reference maps for 2001 and 2009 but instead allocated forest or non-forest classes to the sample points based on observations from high resolution imagery from Google Earth (2003, resolution 0.5-2.5 m) and from field observations (2009 and 2010) to validate the accuracy of the 2001 and 2009 forest cover maps. The 2010 field data (plots on transects in 4 sites) were not used directly as their location did not meet the conditions of probability sampling, but they helped interpretation (Stehman and Czaplewski 1998). The sample points for each year were randomly selected with a minimum distance of 1000m between points to minimize spatial autocorrelation effects (Koenig 1999). We generated four confusion matrices and for each calculated the overall accuracy and kappa coefficient (Cohen 1960, Foody 1992, Congalton and Green 1999) (Table 2.B.2). We also calculated separate quantity disagreement and allocation disagreement following Pontius and Millones (2011) (Table 2.B.2).

Table 2.B.2. Classification accuracy.

	1973		1988		2001		2009	
	F	**NF**	**F**	**NF**	**F**	**NF**	**F**	**NF**
Producer accuracy (%)	94	100	88	98	89	93	96	92
User accuracy (%)	100	69	99	77	95	86	95	93
Overall accuracy (%)	95		91		91		95	
Kappa	0.79		0.79		0.81		0.88	
Allocation disagreement (%)	0		1		6		5	
Quantity disagreement (%)	5		8		4		1	

Appendix 2.C. Variables used for the livelihood characterization at village level.

Ratio variables	N	Minimum	Maximum	Mean	Std. Dev.
Age of the village (years)	14	26.00	209.00	114.71	59.15
% of people with no education	14	.00	73.00	17.43	20.12
% of people with only primary education (at least 3-4 years)	14	.00	.73	.17	.20
% of people with secondary or higher education	14	.05	.50	.24	.14
Distance to any road (map)	14	0.01	6.49	1.92	1.92
Distance to tarmac road (map)	14	2.12	28.38	17.30	8.61
Distance to the nearest road usable in all seasons (km)	14	.00	15.00	5.31	4.41
Number of markets attended	14	1.00	6.00	3.00	1.36
Distance to the nearest market for selling produce (km)	14	.50	17.00	5.37	4.19
Time to the nearest market for selling produce (minutes)	14	5.00	330.00	111.43	89.97
Proportion of transport time done on foot	14	.33	1.00	.85	.20
Number of tree species that people plant in the village	14	.00	10.00	4.29	2.81
% people with a thatched roof	13	.00	.60	.29	.20
% people with a metal roof	13	.08	.95	.65	.25
Average area of land (acres)	13	.67	3.03	1.42	.69
Average number of livestock (equivalents)	13	.67	1.80	1.23	.31
Forest scores	13	28	67	42	3.19
Agricultural scores	13	33	72	58	3.19
Categorical variables	**N**	**Count**	**Percent**		
Access to formal credit	14	11	79		
Electricity (generator/solar)	13	4	71		
Built taps or wells (any)	13	10	77		
Forestry support	11	3	27		
Crops: coffee-banana	14	7	14		
Crops: coffee-maize	14	4[c]	71		
Crops: maize-maize	14	3	14		
Fodder: crop residues	14	10	71		
Fodder: planted grass	14	11	79		
Fodder: pastures	14	3	21		
Fodder: from forest	14	7	50		

[c] Including one Sabiny dominated (Village 12)

Appendix 4.A. Kendall's tau-b correlations between the number of trees, their density and the area of land owned by respondents and the importance of different sources of fuelwood.

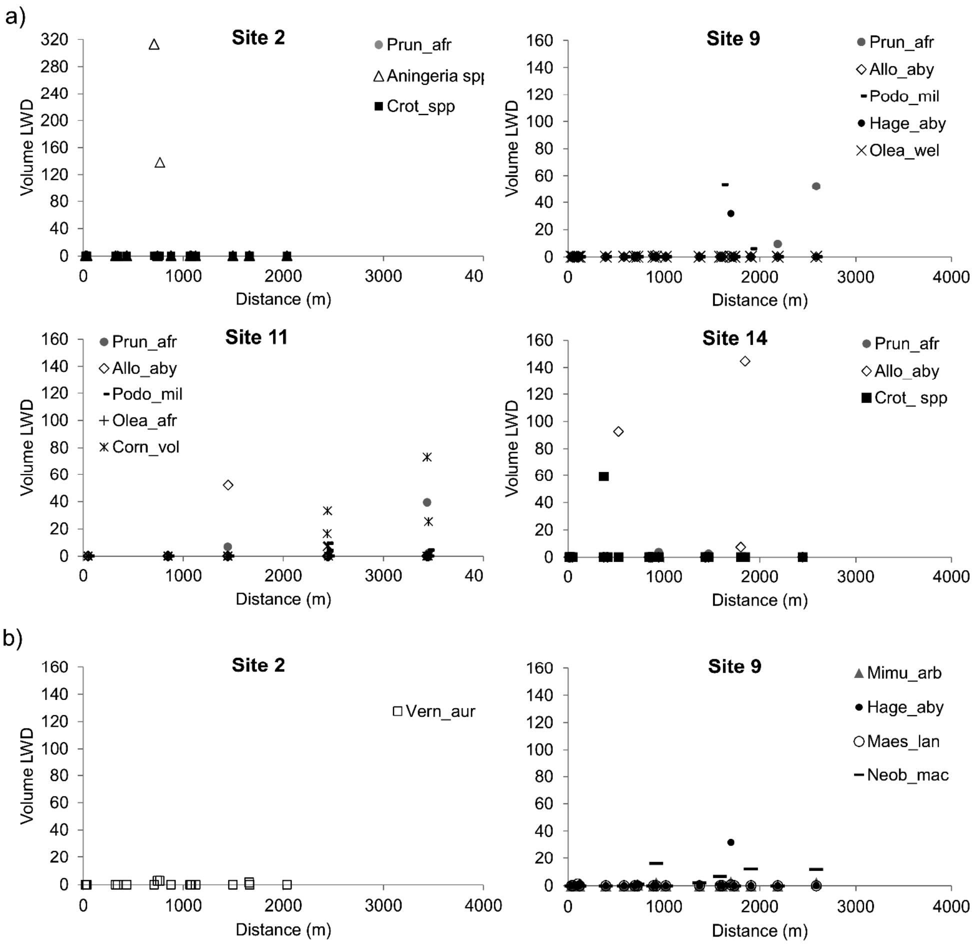
			Rank				Frequency				Head-loads			
			Old forest	Form. encr.	Own land	Market	Old forest	Form. encr.	Own land	Market	Old forest	Form. encr.	Own land	Market
All Sites	Nr of trees	Tau-b	-.095	**.240**	**.465**	.041	**-.125**	**.295**	**.415**	**.156**	**-.135**	**.286**	**.444**	**.145**
		P	.120	**.000**	**.000**	.468	**.035**	**.000**	**.000**	**.007**	**.017**	**.000**	**.000**	**.010**
		N	192	**192**	**192**	192	**192**	**192**	**192**	**192**	**192**	**192**	**192**	**192**
	Tree density	Tau-b	-.050	**.253**	**.355**	.077	-.101	**.311**	**.328**	**.183**	**-.121**	**.295**	**.351**	**.175**
		P	.434	**.000**	**.000**	.199	.103	**.000**	**.000**	**.003**	**.042**	**.000**	**.000**	**.003**
		n	172	**172**	**172**	172	172	**172**	**172**	**172**	**172**	**172**	**172**	**172**
	Land area owned	Tau-b	-.064	-.008	**.289**	-.045	-.030	-.012	**.247**	-.036	-.015	.014	**.248**	-.033
		p	.336	.905	**.000**	.464	.644	.850	**.000**	.561	.801	.825	**.000**	.587
		n	177	177	**177**	177	177	177	**177**	177	177	177	**177**	177
Site 2	Nr of trees	Tau-b	-.015	-.045	**.369**	**-.326**	-.140	-.096	**.327**	-.056	-.135	-.104	**.359**	-.046
		P	.896	.693	**.001**	**.002**	.190	.378	**.003**	.606	.191	.327	**.001**	.661
		N	53	53	**53**	**53**	53	53	**53**	53	53	53	**53**	53
	Tree density	Tau-b	.105	-.056	.176	-.183	-.029	-.037	.157	.029	-.054	-.080	.192	.069
		P	.365	.630	.104	.088	.788	.737	.162	.790	.605	.456	.082	.517
		n	51	51	51	51	51	51	51	51	51	51	51	51
	Land area owned	Tau-b	-.096	.009	**.342**	-.167	-.111	-.087	**.304**	-.124	-.031	-.008	**.269**	-.188
		p	.437	.941	**.003**	.141	.337	.455	**.010**	.288	.778	.945	**.022**	.095
		n	52	52	**52**	52	52	52	**52**	52	52	52	**52**	52
Site 9	Nr of trees	Tau-b	-.064	-.091	**.342**	-.090	**-.291**	.046	**.295**	.037	-.175	.067	**.305**	.153
		P	.601	.444	**.004**	.436	**.018**	.699	**.012**	.753	.151	.567	**.010**	.192
		N	45	45	**45**	45	**45**	45	**45**	45	45	45	**45**	45
	Tree	Tau-b	-.023	-.076	.206	-.063	-.222	.076	.102	.072	-.083	.057	.089	.114

	density	*P*	.863	.554	.108	.612	.092	.555	.424	.573	.523	.649	.487	.369
		n	39	39	39	39	39	39	39	39	39	39	39	39
	Land area owned	Tau-b	.000	-.161	**.396**	-.130	-.148	-.107	**.458**	-.186	-.237	-.080	**.498**	-.087
		p	1.000	.252	**.005**	.337	.305	.449	**.001**	.181	.096	.562	**.000**	.531
		n	39	39	**39**	39	39	39	**39**	39	39	39	**39**	39
Site 11	Nr of trees	Tau-b	.107	.	.178	.015	.044	-.107	-.007	.169	.096	.	.080	.123
		P	.408	.	.166	.904	.724	.408	.954	.156	.434	.	.534	.296
		N	51	51	51	51	51	51	51	51	51	51	51	51
	Tree density	Tau-b	.121	.	.028	.100	.013	-.121	-.017	.227	.064	.	-.095	.217
		P	.402	.	.847	.473	.924	.402	.904	.085	.644	.	.508	.095
		n	41	41	41	41	41	41	41	41	41	41	41	41
	Land area owned	Tau-b	-.068	.	**.278**	-.061	.014	-.057	.128	-.036	.061	.	.191	-.033
		p	.610	.	**.035**	.633	.916	.666	.330	.767	.633	.	.151	.785
		n	45	45	**45**	45	45	45	45	45	45	45	45	45
Site 14	Nr of trees	Tau-b	.224	-.181	.	.	-.082	.120	.	.	-.042	.120	.	.
		P	.106	.193	.	.	.551	.386	.	.	.730	.386	.	.
		N	43	43	43	43	43	43	43	43	43	43	43	43
	Tree density	Tau-b	.229	-.185	.	.	-.111	.103	.	.	-.034	.103	.	.
		P	.106	.193	.	.	.431	.467	.	.	.786	.467	.	.
		n	41	41	41	41	41	41	41	41	41	41	41	41
	Land area owned	Tau-b	.120	-.084	.	.	.247	.103	.	.	.187	.103	.	.
		p	.409	.564	.	.	.085	.478	.	.	.146	.478	.	.
		n	41	41	41	41	41	41	41	41	41	41	41	41

Appendix 4.B. Kendall's tau-b correlations between volumes of dead wood, distance inside the boundary, elevation, slope, basal area and tree density per ha in each site.

			Distance	**Elevation**	**Slope**	**Basal area**	**Tree density**
Site 2	Total DW	Tau-b	.479*	.120	.511**	.571**	.342
		p	.011	.525	.008	.003	.074
		n	16	16	16	16	16
	LWD	Tau-b	.443*	.244	.540*	.574*	.360
		p	.015	.181	.004	.002	.060
		n	17	17	17	17	16
	SWD	Tau-b	.424*	.248	.258	.430*	.311
		p	.023	.184	.180	.023	.112
		n	17	17	17	17	16
	Dead trees	Tau-b	.220	.051	-.157	.120	.122
		p	.304	.812	.473	.578	.577
		n	16	16	16	16	16
Site 9	Total DW	Tau-b	.589**	.337*	.086	.573**	.422*
		p	.000	.038	.602	.001	.011
		n	20	20	20	20	20
	LWD	Tau-b	.536*	.377*	.113	.610*	.414*
		p	.001	.021	.493	.000	.013
		n	20	20	20	20	20
	SWD	Tau-b	.253	.126	.032	.076	.032
		p	.119	.436	.845	.647	.844
		n	20	20	20	20	20
	Dead trees	Tau-b	.183	.250	.254	.120	.291
		p	.340	.193	.192	.541	.137
		n	20	20	20	20	20
Site 11	Total DW	Tau-b	.522**	.546**	-.393*	.518**	.499**
		p	.001	.001	.017	.002	.002
		n	22	22	22	22	22
	LWD	Tau-b	.476*	.465*	-.395*	.454*	.420*
		p	.002	.002	.011	.004	.011
		n	25	25	25	25	22
	SWD	Tau-b	.498*	.601*	-.238	.642*	.667*
		p	.001	.000	.127	.000	.000
		n	25	25	25	25	22
	Dead trees	Tau-b	.358*	.413*	-.084	.361*	.375*
		p	.040	.018	.640	.045	.037
		n	22	22	22	22	22
Site 14	Total DW	Tau-b	.470*	.423*	-.349	.552*	.589*
		p	.005	.013	.050	.001	.001
		n	19	19	18	19	19
	LWD	Tau-b	.404*	.380*	-.332	.543*	.606*
		p	.019	.027	.067	.002	.001
		n	19	19	18	19	19
	SWD	Tau-b	.387*	.292	-.295	.319	.429*
		p	.022	.085	.099	.065	.013
		n	19	19	18	19	19
	Dead trees	Tau-b	.222	.137	-.074	.493*	.317
		p	.237	.467	.712	.010	.098
		n	19	19	18	19	19

Appendix 4.C. Volumes of a) preferred and b) used dead wood species with distance from the boundary.



Volumes of a) preferred and b) used dead wood species with distance inside the boundary. Note the different scale for preferred species in Site 2; these were two large logs. Preferred and used species are the same in Site 11 and 14, except *Ekebergia capensis* in Site 14 but no dead wood was found for that species.

Appendix 4.D. Frequencies for the five highest scoring preferred and used species.

Preferred species			**Used species**		
	Times listed	**% of HH listing**		**Times listed**	**% of HH listing**
2					
Prunus africana	34	64	*Eucalyptus* sp.	35	66
Aningeria spp.	30	57	*Vernonia auriculifera*	21	40
Eucalyptus sp.	19	36	*Markhamia platycalyx*	29	55
Croton spp.	21	40	*Cordia africana*	25	47
Vernonia auriculifera	17	32	Maize stems/cobs	12	23
9					
Prunus africana	40	89	*Vernonia auriculifera*	41	91
Podocarpus milianjianus	30	67	*Hagenia abyssinica*	25	56
Allophylus abyssinicus	20	44	*Neoboutonia macrocalyx*	29	64
Hagenia abyssinica	21	47	*Maesa lanceolata*	28	62
Olea welwitschii	16	36	*Mimulopsis arborea*	23	51
11					
Cornus volkensii	*39*	76	*Cornus volkensii*	*32*	63
Olea chrysophylla	*39*	76	*Olea chrysophylla*	*32*	63
Prunus africana	*43*	84	*Allophylus abyssinicus*	*35*	69
Allophylus abyssinicus	*35*	69	*Prunus africana*	*31*	61
Podocarpus milianjianus	*30*	59	*Podocarpus milianjianus*	*24*	47
14					
Prunus africana	*34*	79	*Vernonia* spp.	36	84
Allophylus abyssinicus	*40*	93	*Solanum* sp.	25	58
Vernonia spp.	*33*	77	*Prunus africana*	14	33
Croton spp.	*25*	58	*Allophylus abyssinicus*	15	35
Ekebergia capensis	*24*	56	*Croton* spp.	9	21

Appendix 5.A. ALOS PALSAR image processing

Figure 5.A1 shows the basic processing steps for each of the radar images used.

1 Import and conversion of SAR image- and meta-data;
2 Radiometric absolute calibration: conversion of input data into backscatter intensity values Gamma nought (γ^0);
3 Geocoding (coarse and fine), use of SRTM 90m Digital elevation model (DEM)
4 Geometric Terrain Correction (GTC): geometric warping to a map grid with correction of slope distortions;
5 Radiometric Terrain Correction (RTC): correction of slope illumination differences;
6 Masking background values and conversion of *Gamma naught* intensity values to a decibel scale [dB]
7 Stacking of all data available

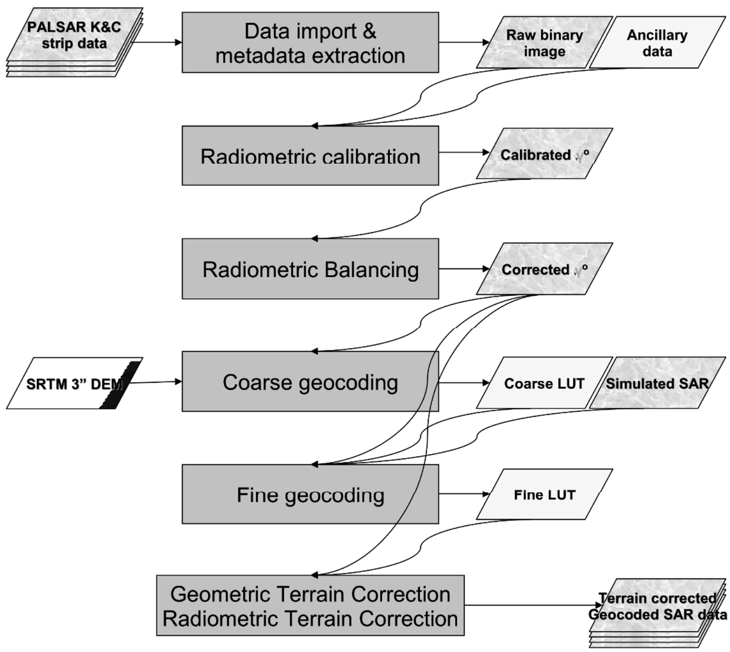


Figure 5.A1: Basic processing steps executed on the ALOS PALSAR strip data used in this assignment (Source: SarVision).

Calibration differences can be the result of inherent radar characteristics, like the incidence angle effect or to differences in vegetation and soil moisture conditions for the different strips (acquisition dates) or even within the same strip (rain events). These calibration problems if severe can be reflected in the accuracy of the classification. Detected calibration differences between images were corrected following specific guidelines found in the literature (Shimada et al. 2009). Inter- calibration of radar data was done to achieved comparable statistical levels in al images.

Geocoding was done using a combination of coarse and fine geocoding procedures. A coarse transformation matrix is generated on basis of orbit state vector information (satellite locations { x,y,z } and velocities { Vx,Vy,Vz } for 11 points and radar imaging geometry. With this transformation and given SRTM DEM, a simulated SAR image is generated and transformed back to SAR geometry. The next step is fine geocoding on a sub-pixel level, consisting of fine co-registration of the real SAR image and simulated SAR, and assessing local geometric corrections due to terrain-induced image distortions.
According to the specifications, the absolute location accuracy of SRTM is 20m, height accuracy 16m (geoid heights relative to EGM'96). FBD PALSAR images were orthorectified using Gamma Remote Sensing software and 90m Shuttle Radar Topography Mission (SRTM) elevation data.

Terrain slopes induce geometric and radiometric distortions in the radar image. Slopes facing the radar tend to fall towards the sensor (lay-over) or be shortened (fore-shortening) and appear very bright. Slopes at the backside of the hill, as seen by the radar system, appear darker or even induce shadows. The amount of distortion depends on the slope angle and orientation in respect to the radar look angle. Correction of slope-induced distortions, as performed in the SarVision radar processing chain, is twofold: correction of geometric distortions, i.e. pulling mountains and hill “straight up”. This is called Geometric Terrain Correction (GTC).The second step is correction for saturation and shadow effects in the radar brightness and is called Radiometric Terrain Correction (RTC). These corrections are shown in Figure 5.A2.

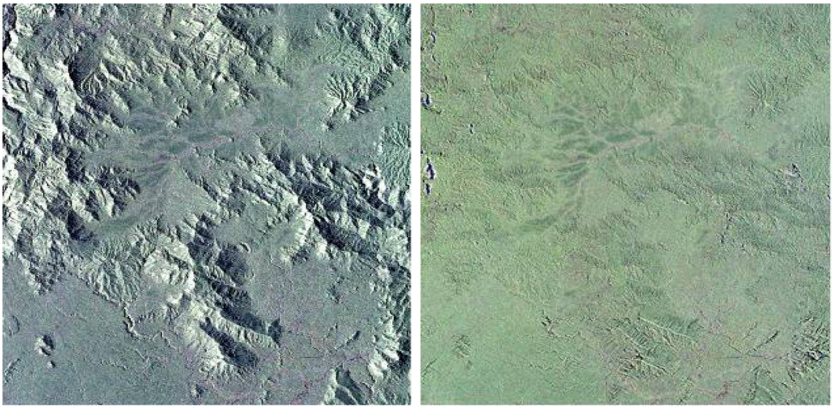


Figure 5.A2: Detail of Radiometric Terrain Correction applied to a PALSAR FBS-FBD color composite. Images left and right are scaled exactly the same. Image left is Geometric Terrain Corrected image before radiometric terrain correction, right image after correction.

Advanced radar classification approach: Forest cover type classification

Classification of radar images suffers from effects inherent to the radar system. Speckle, a so called "salt and pepper" effect, is a stochastic effect that generates very bright or very dark pixels. This effect has obvious consequences in the classification procedure generating misclassification of isolated pixels. The only way to overcome this effect is by using specialized software that cluster and filter the images. In this assignment, recently developed supervised and unsupervised classification algorithms specifically designed for the classification of strip PALSAR data have been used (Hoekman et al. 2011. Hoekman and Vissers 2003). This classification procedure includes the selection of a training dataset that gives the statistical ground to cluster the pixels and classify the image into different vegetation types. A post-processing step usually follows before the final map validation. For the creation of a training data set an extensive sampling of the different ecosystems over the study area is necessary to guaranty a robust classification procedure. For that reason a stratification of the image in different landscapes and vegetation types or ecosystems is necessary. Proper and consistent interpretation of radar images in relation to the landscapes is of great importance for the selection of the training dataset. For each stratum a sufficient number of samples should be included in order to include and understand all the statistical variation within one class.

Appendix 5.B. Above Ground Biomass (AGB) calculated with and without tree height, stem density, basal area, and average tree dbh and tree height per plot.

Site	Plot	AGB with height (Mg/ha)	AGB without height (Mg/ha)	Stem density (stems/ha)	Average dbh (cm)	Basal area (m^2/ha)	Average height (m)
9	1	0.0	0.0	0	0.0	0.0	0.0
9	5	0.0	0.0	0	0.0	0.0	0.0
9	10	64.3	146.0	472	35.2	14.6	12.1
9	15	128.3	209.3	1373	32.0	22.2	15.8
9	17	0.0	0.0	0	0.0	0.0	0.0
9	21	21.5	38.2	159	24.0	6.1	13.0
9	26	79.9	116.4	247	55.0	10.6	20.8
9	31	320.3	595.7	71	116.6	38.6	23.7
9	33	0.0	0.0	0	0.0	0.0	0.0
9	37	17.1	27.8	372	14.9	6.0	10.0
9	42	39.0	58.2	3826	13.1	18.1	10.1
9	47	166.1	201.2	636	46.5	27.1	23.4
9	48	9.0	43.1	1209	13.3	8.7	3.4
9	52	40.7	74.3	805	15.9	12.2	10.1
9	57	83.9	126.4	708	26.3	22.5	16.3
9	62	195.1	320.6	1521	42.9	43.8	17.3
9	67	0.0	0.0	0	0.0	0.0	0.0
9	72	7.1	19.7	34	27.4	2.0	9.0
9	76	144.9	240.4	326	37.2	22.8	16.2
9	81	255.3	426.0	932	43.6	46.5	18.4
2	85	0.0	0.0	0	0.0	0.0	0.0
2	89	6.0	18.0	18	37.4	2.0	10.0
2	94	84.4	132.2	184	41.2	15.2	20.0
2	99	54.8	72.7	142	55.7	10.1	22.2
2	104	18.4	53.3	207	35.7	6.0	8.2
2	109	90.9	162.0	46	73.7	13.2	21.8
2	114	425.1	610.3	502	65.5	48.2	24.6
2	118	0.0	0.0	0	0.0	0.0	0.0
2	123	17.8	34.3	384	18.0	8.4	9.9
2	128	137.6	245.3	6460	29.3	38.0	12.8
2	132	6.5	10.5	2596	4.6	4.3	5.0
2	137	40.5	84.1	455	29.9	13.4	11.6
2	141	29.9	48.2	15	61.4	4.2	24.0
2	146	0.0	0.0	0	0.0	0.0	0.0
2	150	204.8	268.5	180	75.3	26.2	28.0
2	155	127.5	172.5	123	52.3	18.7	25.9

Appendix 5.B. (Continued)

Site	Plot	AGB with height (Mg/ha)	AGB without height (Mg/ha)	Stem density (stems/ha)	Average dbh (cm)	Basal area (m^2/ha)	Average height (m)
11	161	0.0	0.0	0	0.0	0.0	0.0
11	165	0.0	0.0	0	0.0	0.0	0.0
11	168	51.3	103.8	19	70.6	6.1	20.0
11	173	632.0	848.2	442	53.0	61.2	25.5
11	178	44.8	80.2	82	38.8	6.4	16.7
11	186	0.0	0.0	0	0.0	0.0	0.0
11	189	14.9	29.3	1	200.0	2.0	25.0
11	194	300.3	428.9	76	77.3	26.0	27.8
11	199	496.9	667.9	155	71.8	44.0	28.8
11	203	0.0	0.0	0	0.0	0.0	0.0
11	207	0.0	0.0	0	0.0	0.0	0.0
11	210	26.7	60.9	8	78.9	4.0	18.0
11	215	731.1	980.2	477	64.4	66.8	26.6
11	220	594.1	724.9	255	57.7	50.3	29.3
11	222	0.0	0.0	0	0.0	0.0	0.0
11	226	0.0	0.0	0	0.0	0.0	0.0
11	229	39.3	58.1	12	84.7	4.1	28.0
11	234	685.3	760.9	258	58.0	49.7	32.3
11	239	495.9	769.2	213	79.7	51.5	24.7
11	249	49.9	74.0	6	107.7	4.2	30.0
11	254	668.1	761.8	336	58.8	48.8	31.5
11	259	500.9	755.2	97	92.2	47.1	28.5
14	262	28.8	42.9	4	86.9	2.2	29.0
14	266	185.4	301.7	74	72.4	20.9	24.2
14	271	18.2	38.9	28	43.8	4.2	16.0
14	276	384.1	541.5	116	79.4	37.1	28.7
14	280	0.0	0.0	0	0.0	0.0	0.0
14	284	141.2	230.0	356	66.2	17.4	22.3
14	289	106.1	151.6	1316	36.2	22.3	16.0
14	294	307.0	433.1	426	63.8	34.6	25.0
14	298	0.0	0.0	0	0.0	0.0	0.0
14	302	17.0	26.1	4	85.0	2.0	28.0
14	307	88.4	103.0	12	96.4	7.0	37.7
14	313	160.0	305.8	996	52.0	28.0	17.4
14	316	0.0	0.0	0	0.0	0.0	0.0
14	320	50.3	86.3	2789	28.4	14.1	13.1
14	325	33.8	98.7	36	58.9	6.3	12.2
14	330	240.0	395.3	931	49.7	36.0	19.3
14	331	0.0	0.0	0	0.0	0.0	0.0
14	335	0.0	0.0	0	0.0	0.0	0.0
14	340	158.2	319.4	347	78.0	24.0	18.2

Summary

At the global scale, the need to provide food and energy for a growing world population is leading to changes in land use with important consequences for remaining forest areas. At regional or national scales, poor and land-hungry farmers are considered the main threat to forest conservation. In this study I investigate how conflicting goals by different actors led to various outcomes for the forest on Mt Elgon, Uganda, under different historical contexts and what this means for long term conservation efforts.

Mt Elgon is a large isolated volcano straddling the Uganda/ Kenya border. Mt Elgon is an important water catchment area and has significant biodiversity values. The forest on Mt Elgon are also an important source of agricultural land, timber, fuel wood and other forest resources for local communities. All land within 20 km outside the protected area is under cultivation and population densities are high (150-1000 p/km^2).

On the Ugandan side, large scale deforestation took place in the 1970s and 1980s with subsequent recovery after 1993, when a national park was established to protect the forest and the higher altitude moorlands. Since then regeneration, renewed encroachment and local forest use have led to various degrees of recovery and degradation in different places inside the park. Mt Elgon has a history of conservation and development projects (since the early 1990s) and more recently pilot REDD+ have been implemented both inside and outside the protected area. In this study I explore the factors that influenced local people's motivations to respect rules and regulations about forest clearing (Chapter 2), their use of forest resources (Chapter 3) and their dependence on the forest as a source of fuelwood (Chapter 4) on Mt Elgon, Uganda. Finally I evaluate the use of radar satellite data to estimate above ground biomass on Mt Elgon, Uganda and Kenya (Chapter 5).

I used a combination of satellite image analyses together with historical information, population census data and interviews with local informants, to analyse the drivers of forest cover change in three periods between 1973 and 2009 on Mt Elgon, Uganda (Chapter 2). More than 25% of the forest cover on Mt Elgon in Uganda was lost in 35 years. But Mt Elgon did not experience only forest loss in this period. Locally, there were areas of recovery. By analysing local variations I found that it is the *context* (e.g. law enforcement, collaborative management, political interference) under which drivers such

as population, wealth, market access and commodity prices operate, rather than the drivers *per se*, that influences impacts on forest cover. Agricultural expansion on Mt Elgon cannot simply be linked to population and poverty or other individual drivers. High prices for cash crops were not always associated with increased deforestation but led to varying impacts. These impacts depended on the relationship between local people and park management (and whether there was a collaborative management agreement) and the main cash crop – i.e. annual versus perennial.

Using measures of forest structure and indicators of human activity in 343 plots in four study sites on Mt Elgon, Uganda I found that local forest uses strongly influenced forest structure, even in a study site where people had a collaborative management agreement with the park authorities (Chapter 3). The type of resources collected and the impacts thereof varied according to the land use system outside the park (and sometimes inside): where people grew crops that required supports such as bananas and climbing beans, impacts of small stem-harvesting were apparent on regeneration. In areas where cattle grazing was important this led to the almost absence of seedlings and a forest composed mainly of large trees. Human impacts also affected tree species richness and areas in intermediate states of disturbance showed higher richness than old-growth forest or more severely degraded areas. I show that impacts vary among sites according to their specific histories and contexts.

Interviews with 192 households about fuelwood use and a survey of dead wood in 81 plots inside the park, revealed depletion of dead wood on the edge of the park, particularly near the most densely populated sites (Chapter 4). Species that were highly preferred and used as fuelwood were affected by harvesting. Some showed signs of depletion - in particular when they were also valued as sources of timber or for other uses - with possible impacts on tree biodiversity. Allowing the collection of an important forest resource such as fuelwood is double-edged because it creates opportunities for more destructive activities such as tree cutting for timber or charcoal. On the other hand it contributes to improving relations between local people and park staff and is therefore a basis for further negotiation or improvement of management arrangements. I also found indications that trees on people's own land can provide alternative sources of fuel.

A relatively new approach attempting to reconcile local livelihood improvement and conservation on Mt Elgon involves PES schemes based on REDD+. Such schemes need information on the carbon content of the forest and would benefit from consistent remote sensing observations and space borne (ALOS PALSAR) biomass estimations to provide carbon levels, detect deforestation and degradation and monitor forest dynamics. The relascope method was effective for direct basal area estimations and gave consistent estimates of above ground biomass. I found that above ground biomass is very high in some areas of Mt Elgon, reaching above 800Mg/ ha (Chapter 5). However, the biomass map produced from the relationship between plot-AGB and radar backscatter values did not meet our expectations. Limiting factors likely included the sampling strategy and topography (Chapter 5).

The findings in Chapters 2, 3, and 4 give evidence that simple models based on single drivers of deforestation (i.e. population or poverty) cannot explain local variation. In the global debate on drivers of forest change and in discussions about devolved forest management, local use of forest resources and its impacts, simple models leading to "simplified institutional prescriptions" do not reflect complex local social and ecological realities. Each of the generally recognised drivers of forest change - from global theories on agricultural expansion and population, to theories about the role of wealth and poverty and that of markets and prices - in themselves do not explain forest change at the local scale. Rather, their importance is the result of the interaction of factors at different scales that determine the contexts under which local people make decisions on forest use. Local motivations and attitudes are shaped by the contexts and factors under which people operate. This has important implications for the design of more locally adapted and ecologically and socially sustainable management arrangements on Mt Elgon and elsewhere. These are necessary because current practices appear to lead to forest degradation and resource depletion, potentially affecting Mt Elgon's role as a water tower for the region as well as the habitat for important biodiversity in the long run.

Interventions with potential benefits for both forest conservation and local communities vary from agricultural intensification to agroforestry practices, building on existing land-use systems around Mt Elgon, to payments for environmental services schemes such as under REDD+. However, in a situation of historical conflicts such as on Mt Elgon, management authorities need urgently to engage in building relationships of trust with local communities, either through collaborative management agreements or by other means of showing genuine acknowledgement of local needs and consistent attitudes with regard to rules, rights and responsibilities. In the context of REDD+ projects, we have first shown the technical difficulties in assessing biomass values in a complex landscape such as Mt Elgon (Chapter 5). However, beyond these technical aspects, the findings from the previous chapters provide insights into how the success of such schemes may be influenced by other factors - and the complex contexts under which these play out - that affect people's motivations. These factors operate at multiple levels: from the interests of national REDD strategies to risks of appropriation of benefits by politicians or elites, to local motivations to participate or not.

Conservation and development interventions need to address local factors while recognizing influences operating at national and global levels. Also, generalisations about trade-offs between local uses and conservation are confounded by location specific characteristics, calling for actions that take into account these variations. Truly participatory approaches that empower local people combined with incentives to develop alternative resources, such as trees for various uses, including fuelwood, can support more sustainable forest management for both people and conservation. Both international conservation actors, which often have a strong influence on national policies and resource allocation for conservation and economic development, as well as national and local forest management authorities (e.g. Uganda Wildlife Authority) need to recognise that incentives that influence people's motivation for action in relation to the management of common pool resources vary locally and can therefore not be designed globally.

Samenvatting

Om de groeiende wereldbevolking van voldoende voedsel en energie te kunnen blijven voorzien, treden er veranderingen op in landgebruik, met verstrekkende gevolgen voor bestaande bosgebieden. Op nationale en regionale schaal worden vooral arme boeren met weinig land gezien als de grootste bedreiging voor de succesvolle bescherming van bossen. In deze studie, onderzoek ik hoe tegenstrijdige doelen van diverse belanghebbenden, onder verschillende historische contexten hebben geleid tot lokaal varierende uitkomsten voor het bosgebied op Mt Elgon, Uganda. Hieruit trek ik lessen voor bosbescherming op de lange termijn.

Mt Elgon is een grote geïsoleerde vulkaan op de grens van Uganda en Kenia. Het is een belangrijk stroomgebied met een hoge biodiversiteitswaarde. Daarnaast zijn de bossen op Mt Elgon voor de lokale gemeenschappen een belangrijke bron van landbouwegrond, (brand)hout en andere bestaansbronnen. Alle grond in een strook van 20 km rondom het beschermde bos is gecultiveerd en heeft een hoge bevolkingsdruk (150-1000 p/km^2).

In de jaren zeventig en tachtig van de vorige eeuw was de Ugandese kant van Mt Elgon onderhevig aan grootschalige ontbossing. Herstel volgde toen in 1993 het gebied werd uitgeroepen tot nationaal park. Dit had als gevolg dat het resterend bos en de gebieden boven de boomgrens een beschermde status kregen. Sinds die tijd is er een bos ontstaan met gebieden in verschillende mate van herstel of verval, door regeneratie, hernieuwd illegaal kappen voor landbouw en door lokaal gebruik van het bos. Mt Elgon heeft een geschiedenis van projecten met als gecombineerd doel het beschermen van het bos en het bevorderen van lokale economische ontwikkeling (sinds het begin van de jaren negentig). Deze projecten zijn recentelijk opgevolgd door proefprojecten voor de implementatie van REDD+ (dat zich richt op het economische waarde toekennen aan bos voor koolstofopslag) zowel binnen als buiten het beschermd gebied. In deze studie onderzoek ik de factoren die van invloed zijn op de motivatie van de lokale bevolking rond Mt Elgon om regels en regelgeving omtrent het kappen van bos te respecteren (Hoofdstuk 2), het gebruik van hulpbronnen uit het bos (Hoofdstuk 3) en hun afhankelijkheid van het bos als bron van brandhout (Hoofdstuk 4). Als laatste bestudeer ik de bruikbaarheid van radargegevens, vanuit een satelliet gemeten, om de bovengrondse biomassa van het bos op Mt Elgon in Uganda en Kenia te schatten (Hoofdstuk 5).

Om de factoren te analyseren die verantwoordelijk zijn voor de veranderingen in het bosoppervlak op Mt Elgon in Uganda, heb ik voor drie periodes, tussen 1973 en 2009, satelietbeelden analyseerd in combinatie met geschiedkundige informatie, gegevens over de bevolkingsdichtheid en gesprekken met lokale informanten (Hoofdstuk 2). De resultaten laten zien dat binnen deze 35 jaar het totale bosoppervlak op Mt Elgon met meer dan 25 procent is afgenomen. Maar er vond in deze periode niet alleen ontbossing plaats. Op sommige plaatsen is ook toename van bos gemeten. Door het analyseren van lokale variatie kon ik achterhalen dat vooral de *context* (bijv. rechtshandhaving, inspraak in beheer, politieke tussenkomst) de uitkomsten voor het bosoppervlak bepaalde, en dat drijvende krachten zoals bevolkingsdichtheid, rijkdom, toegang tot de markt en grondstoffenprijzen op zichzelf hierin minder bepalend zijn. Uitbreiding van landbouwactiviteiten binnen het beschermde gebied op Mt Elgon kan niet simpelweg verklaard worden door een toenamein bevolking en door armoede, of andere op zichzef staande factoren. Hoge prijzen voor handelsgewassen veroorzaakten niet noodzakelijkerwijs een toename in ontbossing. Ook de relatie tussen de lokale bevolking en de parkbeheerders (en of er afspraken waren over bosgebruik en inspraak in het parkbeheer) en het soort handelsgewas (d.w.z. eenjarig of overblijvend) hadden invloed op dit proces.

Het gebruik van het bos door lokale bevolking had een sterke uitwerking op de vegetatiestructuur, ook in gebieden waar de lokale bevolking inspraak heeft in het parkbeheer door middel van zogenaamd 'collaboratief' bosbeheer ('collaborative management'). Dit heb ik aangetoond aan de hand van metingen aan de vegetatiestructuur en indicatoren van menselijke activiteit in 343 plots verspreid over vier onderzoeksgebieden op Mt Elgon in Uganda (Hoofdstuk 3). Wat er uit het bos verzameld werd en het effect daarvan op de bosstructuur, verschilde afhankelijk van het soort landgebruik buiten het park (en soms ook daarbinnen). In gebieden waar mensen gewassen verbouwen die ondersteuning nodig hebben zoals bananen en bonen, werden veel stammen met een kleine diameter verzameld uit het bos, met negatieve gevolgen voor de regeneratie van bomen. In gebieden waar het weiden van vee een belangrijk onderdeel van het landbouwsysteem was, bestond het naburige bos voornamelijk uit volwassen bomen en ontbraken de zaailingen. Menselijke activiteiten hadden ook een effect op de soortenrijkdom van bomen: gebieden die matig verstoord waren vertoonden een hogere diversiteit aan boomsoorten dan oude bossen of sterk verstoorde gebieden. De invloed van menselijke activiteit verschilde per gebied, en hing af van de specifieke lokale geschiedenis en de context (Hoofdstuk 3).

Uit de resultaten van interviews met personen uit 192 verschillende huishoudens over hun brandhoutgebruik, in combinatie met de gegevens van de hoeveelheid dood hout in 81 plots binnen het park, werd duidelijk dat aan de rand van het park dood hout zeer snel verwijderd wordt, vooral in de buurt van dicht bevolkte gebieden (Hoofdstuk 4). Boomsoorten waarvan het hout zeer geliefd of veel gebruikt werd als brandhout vertoonden effecten van oogst. Sommige soorten vertoonden tekenen van populatieafname, vooral als ze ook gewild waren als hout voor constructie of andere gebruiken. Dit heeft mogelijk gevolgen voor de biodiversiteit van bomen op Mt Elgon. Het toestaan van verzamelen van brandhout uit het park heeft twee tegengestelde kanten. Aan de ene kant biedt het mensen de kans om andere, meer schadelijke, activiteiten te ondernemen, zoals bijvoorbeeld het kappen van bomen voor timmerhout of het maken houtskool. Aan de andere kant draagt het bij aan een goede verstandhouding tussen de lokale bevolking en de beheerders van het park. Het is daarmee een basis voor verdere onderhandelingen over bosgebruik of voor nieuwe beheerssystemen. Tijdens de interviews bleek ook dat bomen op privaat land als alternatieve bron kunnen dienen voor brandhout (Hoofdstuk 4).

Een relatief nieuwe aanpak om te pogen het lokale levensonderhoud te verbeteren en het bos op Mt Elgon beter te beschermen is via een systeem van directe belatingen voor milieudiensten (‘payments for environmetal services’ of PES in het engels) gebaseerd op REDD+. Deze aanpak omvat een financiële compensatie voor het niet exploiteren van bos (PES regelingen gebaseerd op het REDD+ programma). Dit soort regelingen kunnen alleen geïmplementeerd worden als er informatie beschikbaar is over de hoeveelheid koolstof die in een bos vastgelegd is. Consistente metingen hiervan zijn daarbij waardevol. Naast het berekenen van de hoeveelheid koolstof, kunnen consistente observaties via teledetectie en schattingen van de bovengrondse biomassa (met ALOS PALSAR radar data) gebruikt worden om ontbossing, degradatie en bosdynamiek te monitoren (Hoofdstuk 5). De relascoop-methode bleek een effectieve methode voor het direct schatten van het grondvlak (‘basal area’ in het engels). Het grondvlak is sterk gecorreleerd aan de bovengrondse biomassa. In sommige gebieden op Mt Elgon heb ik zeer hoge waarden voor bovengrondse biomassa gemeten, tot meer dan 800MG/ha. Een kaart van de biomassa op Mt Elgon gebaseerd op de gevonden relatie tussen de gemeten bovengrondse biomassa en de radarwaarden uit de ruimte, voldeed niet aan de verwachtingen. Beperkende factoren waren waarschijnlijk de bemonsteringsstrategie in het veld en de topografie van het landschap (Hoofdstuk 5).

De bevindingen in Hoofdstuk 2,3 en 4, laten zien dat simpele modellen gebaseerd op één enkele drijvende kracht voor ontbossing (bijvoorbeeld bevolking of armoede), lokale variatie niet kunnen verklaren. In het algemene debat over de drijvende krachten van veranderingen in bosoppervlak en in discussies over gedecentraliseerd bosbeheer, lokaal bosgebruik en de impact daarvan, leiden simpele modellen vaak tot “simplistische institutionele voorschriften” die de complexe lokale sociale en ecologische realiteiten niet reflecteren. Geen van de algemeen geaccepteerde drijvende krachten achter bosverandering – van de globale theorieën over uitbreiding van de landbouw en over de invloed van bevolking, tot de theorieën over de effecten van rijkdom en armoede of markten en prijzen – verklaardt, op zichzelf staand, lokale bosverandering. De uitwerking die deze krachten hebben is het resultaat van de interactie van diverse factoren op verschillende niveaus, die zo de context bepalen waarin lokale mensen beslissingen nemen over het gebruik van bos. Lokale motivaties en houdingen worden gevormd door de context en de factoren waaronder ze opereren. Dit heeft implicaties voor het ontwerpen van duurzamere beheerssystemen die beter afgestemd zijn op lokale ecologische en sociale omstandigheden op Mt Elgon en elders. Dit is noodzakelijk omdat er aanwijzingen zijn dat huidige praktijken leiden tot bosdegradatie en uitputting van natuurlijke hulpbronnen. Dit heeft op de langere termijn potentiële gevolgen voor de rol van Mt Elgon als watertoren voor de regio en voor de habitat van belangrijke biodiversiteit.

Strategieën die ten bate kunnen zijn van zowel bosbescherming als van lokale gemeenschappen op Mt Elgon varieëren van intensivering van de landbouw tot het ontwikkelen van boslandbouwsystemen (‘agroforestry’), het voortbouwen op bestaande landgebruikssystemen en betalingen voor milieudiensten zoals onder REDD+ programmas. Maar op plekken zoals Mt Elgon, met een geschiedenis van conflicten tussen locale bevolking en parkbeheer, is het vooral ook van belang dat deze laatste zich toe legt op het kweken van vertrouwen bij lokale gemeenschappen rondom beschermde gebieden. Zowel systemen van collaboratief bosbeheer als het werkelijk erkennen van lokale behoeften en een consistente houding ten opzichte van beleid, regels, rechten en verantwoordelijkheden kunnen hier sterk aan bijdragen. Wat betreft de potentie van REDD+ projecten: we laten zien dat er een aantal technische beperkingen zijn aan het meten van de biomass in een topografisch complex landschap zoals op Mt Elgon (Hoofdstuk 5). Desalnietemin geven de bevindingen uit voorgaande hoofdstukken inzicht in hoe de uiteindelijke uitkomsten van zulke systemen afhankelijk zijn van allerlei andere factoren die de motivate en handelingen van mensen beïnvloeden, en de complexe contexten waaronder deze factoren ageren. Deze factoren bestaan op verschillende niveaus, van de belangen die spelen binnen nationale REDD+ strategieën tot aan het risico van appropriatie van middelen door politici of elites of tot aan lokale motivaties om al dan niet “mee te doen”.

Interventies met als gecombineerd doel het beschermen van bos en het bevorderen van lokale economische ontwikkeling moeten zowel lokaal specifieke factoren en invloeden op national en mondiaal niveau erkennen. Generalisaties over de ‘trade-offs’ tussen bosgebruik door lokale gemeenschappen en bosbescherming worden verstoord door lokaal specifieke kenmerken die juist vragen om aangepaste acties. Benaderingen die gebaseerd zijn op oprechte participatie van locale gemeenschappen, in combinatie met het stimuleren van alternative hulpbronnen zoals het planten van bomen voor verschillende doeleinden, inclusief brandhout, kunnen duurzamere bosbeheerssystemen voor zowel mensen en natuurbescherming ondersteunen. Zowel internationale natuurbeschermingsactoren - die vaak een setrke invloed hebben op nationaal beleid en de toewijzing van middelen voor natuurbescherming en economische ontwikkeling - als nationale en locale bosbeheerautoriteiten(e.g. Uganda Wildlife Authority) zullen moeten erkennen dat de factoren die de motivatie van mensen bepalen in relatie tot het beheer van gemeenschappelijk hulpbronnen lokaal varieren en daarom niet globaal kunnen worden bepaald.

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I started thinking about a PhD after my Master's degree in Wageningen, but did not feel I knew enough to be able to decide on a topic. I resolved to first get some experience. Whilst working at CIFOR, various people I met strongly encouraged me to go ahead and pursue a PhD. Douglas Sheil, Robert Nasi, Jeff Sayer and Frans Bongers: you all contributed to my finally taking the first step on the journey that led me here. Thank you! I especially remember Jeff saying one day: "do you want to become a teacher? If so then fine, nothing wrong with that, but if not: go and get a PhD!"

I resolved to find a topic for a thesis. A partner moving to Kisumu brought me to Kenya and I took the opportunity to think about what I wanted to study in this environment where the discussion on balancing biodiversity conservation, local livelihoods and economic development has been relevant for many years. Thanks to Robert, I attended the CIFOR-ICRAF Biodiversity Platform workshop on Lombok in December 2006. Armed with ideas gained in that workshop, I later met with Ed Barrow at IUCN in Nairobi, to ask where such an approach would be useful and applicable within the region. One option that emerged was Mt Elgon, a large extinct and forested volcano on the border between Uganda and Kenya. I went on to study the existing literature on the area and became fascinated by the rich history of its people and forest; and the conflict between local people and the protected area management in both countries. So Mt Elgon it was. I first met George Sikoyo and Mathias Chemonges (MERECP) in Kampala. Thank you George for taking me along to the project area on the Eastern side of Mt Elgon. This gave me an opportunity to talk to local people in villages neighbouring the park, forest managers and local administration officials. Matthias, it was always good to meet you first in Kampala and later in the LVBC office in Kisumu and discuss my research and how it could be useful to MERECP's activities. I hope it is!

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Down the mountain, my refuge was in Mbale, a small town built on coffee. My usual accommodation, the Landmark Inn, was a friendly place but cleaning was not a standard service. When I'd had enough of the dirt and grime, I would escape to the Mount Elgon Hotel, an oasis of calm at the foot of one of Mt Elgon's impressive cliffs. I will never forget the day I came in from the mountain, dirty and tired, and was told that all rooms were full for a conference, only to suddenly be given a key because: "really, you look like you need it!" Thank you, Victoria, Stephen and the rest for making me feel so welcome. I wish I could say that I was grateful to the drivers of the various forms of transport that I used, as they all got me, and my assistants, where I wanted alive. And I am grateful, now. I was feeling slightly less benevolent whilst packed for hours in a 14 seater matatu with more than 20 people, or on the back of a small truck with 27 people on top of 20 bags of maize, at least 10 whole bunches of bananas and various household items on a rocky mountain road, or – when vehicles could not go any further, on the back of a motorcycle-taxi with a large backpack at the back and a small one in front, containing all I needed for two weeks of fieldwork in the village, slipping uphill through the mud.

My memories of Mt Elgon include some very uncomfortable ones, including bed-bugs, fleas, nightly safari-ant invasions, lost toenails, bruises, stings and dehydration. Managing local expectations could be mentally challenging and forest surveys were physically exhausting, but also hugely satisfying. The sight of the mountain and its cliffs in the distance when coming from Kampala or Kisumu will always be engraved in my mind. As will the unbelievably beautiful (and cold!) starry nights in the village. I will always remember the sounds early in the mornings, when the mountain's slopes became alive with the sounds of crowing cockerels, pots and pans for breakfast, wood-chopping and the voices of children on their way to school. Thank you Masaba for welcoming me!

Numerous people accompanied me on stretches of this journey, making the travel lighter. Many became friends. Ed: fond memories of Bwindi and Otley. Alastair and Michelle in Kampala: thank you for your company, for taking me out and for lending me your spare room! In Kisumu: Anja, Eric, Ellen, Kim, Freek & Nathalie & the kids, Marc & Machiel, Alie, Matt & Nelli, Stephanie& Per, Mille & Jeppe, Alkesh, Kolya, Emma and many more: thank you for being part of our life! In Wageningen the staff and students at PPS and CSA made me feel welcome and part of the group at each of my visits. The lunch-group in particular: Madeleine, Pytrik, Jochem, Sheida, Rik, Christiaan, Bob, Marjolein, Katrien, Esther, Renske, Lotte, Greta, and others: thank you! And Ria and Charlotte, the rocks of PPS: what would I have done without you? Thank you Manon, Ronald and Esther for picnics and dinners! Diego, Renake and Gatien (on occasion) helped making N9 a home after Rik, Rio and Ebel moved back to Laos. Diego, I did as you said: "just focus on it and you will get there!". Valerio thank you for your help with the data and, more importantly, our talks; Ben, we should go for a beer again now that I am done, and resume our talks about Uganda, fieldwork, coffee, family and others. Caperune, you provided the rhythm to my time in Wageningen and hopefully for some more to come! Now in Cambridge: I am grateful to my new colleagues and friends of the "usual crowd" at UNEP-WCMC for the moral support whilst I worked fulltime and finished a PhD thesis at the same time. And Ruth: I will be a better housemate after this!! Thank you for the moral support, the meals, the tea and the chocolate whilst I sat in my room, typing away.

Moving back in with my extended family Rio, Rik and Ebel at N9 in 2011 truly was coming home (where were you though Alex?). I will forever treasure our evenings on the couch, talking with chocolate and tea – and something stronger for Rik of course, our walks, runs, beers at the Zaaier and weekend breakfasts. Thank you for always being there! Dear "Keek op de week" - Marcela, Laurens, Emma, Bart, Joost, Monique, Erik, Sebastien, Madeleine, Paul and more occasional joiners- thank you for the beers at the Zaaier, coffee at the Villa, the spontaneous evenings, the New Year and Christmas parties, the friendship. Because of you I may have found a home-town after all! Jacqueline, Judith and Susanne, your friendship and support have been invaluable to me throughout this journey. I hope there will be many more Oerols to come and that we continue to share the good and the sometimes not so great times. Erika, thank you for more than 20 years of friendship! And for designing the beautiful cover of this thesis. Erwan, I owe a large part of this achievement to you. Thank you for your support and our time in Kisumu. I will always remember our house on the lake, the garden, the sailing, the sound of hippos, the Fish Eagles and the screeching Hamerkops. And Mouse.

Madeleine and Marcela, my paranymphs; thank you for your friendship! I am immensely grateful for your company, your wisdom and support on my way here. Madeleine: thank you for all those coffee-times at PPS, the occasional sneaking-out for lunch on Wednesdays, the gardening and our talks. Marcela: what can I say… I hope we get to do yoga together, cook, talk and talk for a long time to come, and publish our paper! Jasper, the end of this PhD was not the easiest start. Thank you for bearing with me, for the statistics, the cooking, and for supporting me on the final stretch of this journey. Finally, I am grateful to my family: Lineke, Joachim and the kids: thank you for being there! To my parents: thank you for making this possible. Your constant support and our life in Africa and elsewhere are what led me here today.

PE&RC Training and Education Certificate

With the educational activities listed below the PhD candidate has complied with the educational requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

Review of literature (6 ECTS)

- Interactions between human processes, resource use and biodiversity conservation; the case on Mount Elgon, Uganda, Kenya

Writing of project proposal (4.5 ECTS)

- Interactions between human processes, resource use and biodiversity conservation; the case on Mount Elgon, Uganda, Kenya

Post-graduate courses (7.1 ECTS)

- Analysing farming systems and rural livelihoods in a changing world: vulnerability and adaptation; PE&RC /University of Zimbabwe (2008)
- Analysing of land dynamics and sustainable development in an interdisciplinary perspective; PE&RC (2009)
- Geostatistics PE&RC (2011)

Deficiency, refresh, brush-up courses (2.1 ECTS)

- Multivariate statistics; CreaSience, Montreal (2005)
- ArcGIS; CIRAD, Montpellier (2005)
- Introduction to R for statistical analysis (2008)

Competence strengthening / skills courses (2.1 ECTS)

- Career assessment; WGS (2012)
- REDD Science governance; PE&RC (2012)
- Project management; UNEP-WCMC (2013)

PE&RC Annual meetings, seminars and the PE&RC weekend (2.4 ECTS)

- PE&RC Weekend (2008)
- PE&RC Day (2011)
- PE&RC Last year weekend (2012)
- PE&RC Day (2012)

Discussion groups / local seminars / other scientific meetings (6.4 ECTS)

- CIFOR-ICRAF Biodiversity Platform workshops; Lombok, Indonesia (2007)
- ICRAF 2 Scientific meetings (2008-2011)
- Spatial Methods (SPAM) discussion group (2008-2011)
- ITFC information sharing workshop (2010)
- Plant production Systems group lunch-time presentations; 1 hour weekly (2011-2012)
- Symposium on ‘Conservation Science for the Future’; Utrecht University (2012)
- UNEP-WCMC lunch-time presentations (2013)

International symposia, workshops and conferences (3.7 ECTS)

- International congress of Ethnobotany; Yeditepe University, Turkey (2005)
- ISEE Meeting; Nairobi, Kenya (2008)
- East African Ecological Society (ESEA) meeting; Kampala, Uganda (2009)
- DIVERSITAS; poster presentation; Cape Town (2010)

About the author

Marieke Sassen was born in 1974 in Morogoro Tanzania, where she had her first experiences with Africa's wildlife and landscapes as her parents took her on bumpy rides into the Ngorongoro crater and the Serengeti plains. After short stays in various other countries around the world, the family moved to Côte d'Ivoire where she attended French primary and secondary schools in Abidjan. She obtained her Baccalauréat D (Natural Sciences) at Lycée Blaise Pascal in Abidjan in 1993, after which she went on to study at Wageningen University in the Netherlands. She graduated in 1999, with a specialisation in natural resource management in the tropics, focusing on conservation ecology and tropical livestock systems. Her main Master's thesis, in collaboration with the Centre of Environmental Sciences in Leiden, was a field study of the population ecology of an antelope (*Kobus kob kob*) in a hunting area of Northern Cameroon. Her second Master's thesis, at the Department of Animal Production Systems in Wageningen, studied the use of wildlife as an alternative for cattle ranching in Africa. Marieke graduated from Wageningen University in 1999.

After graduating, Marieke held several jobs including as a course assistant supporting an international course on Livestock Farming Systems at now Wageningen International. However, her studies and the time spent in the field in Cameroon gave her a strong interest in the challenges of reconciling nature conservation with the needs and aspirations of local people living in or near conservation areas. From 2002 she worked on this topic as a Junior Professional Officer at the Centre for International Forestry Research (CIFOR) in Yaoundé, Cameroon. She worked in CIFOR's Environmental Services and Sustainable Use of Forests Programme and switched her focus to tropical rainforests, plants and people. She led field research teams in village and forest based surveys of local people's perspectives on forest conservation and biodiversity in Cameroon and Gabon. In 2006, she briefly worked as a consultant for Wageningen International, to support the development of course material on "Landscape Functions and People: applying strategic planning approaches for good natural resource governance", and various smaller conservation-related initiatives.

In 2007 she left for Kisumu, Kenya and developed a multidisciplinary research project for work on Mt Elgon, an old volcano straddling the Uganda/ Kenya border, with long-standing conflicts between forest conservation and local communities over land and forest resources. Prof. Ken Giller (Plant Production Systems Group, Wageningen University), Dr. Douglas Sheil (CIFOR and Institute for Tropical Forest Conservation (ITFC), Uganda) and Dr. Jean-Marc Boffa at the World Agroforestry Centre (ICRAF, Nairobi) agreed to be her supervisors. From 2008, she was a PhD candidate at Wageningen University and a degree fellow at ICRAF (until 2011).

Since January 2013 Marieke has been working at UNEP's World Conservation Monitoring Centre (UNEP-WCMC) in Cambridge (UK), on assessing trade-offs and synergies between agriculture development and biodiversity, and to support the integration of biodiversity concerns into land use policy and planning.

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