# Spatial Measures of Human Disturbance around Kibale National Park, Uganda

by

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# *Abstract*

Communities neighbouring protected areas often disproportionally accrue the costs of conservation, but they can also benefit from the existence of a protected area. In this dissertation, I investigate the value and spatial distribution of benefits and losses accrued by villages next to Kibale National Park in Uganda, and determine if the accrual of benefits and losses affect the levels of illegal resource extraction from the park. Since other factors may also influence illegal off-take from the park, this assessment is carried out within the context of local demographic and socio-economic landscapes.

Illegal resource extraction was measured within the boundaries of the park near 25 study villages. Data for benefit and loss accrual, as well as demographics, and livelihood activities, were collected through focus groups, a household survey, and key informant interviews. Losses to crop raiding by park-protected animals were also collected by physically verifying damage over a six month period. The influence of local urban markets was represented by road distance to urban centres.

Accrued financial benefits included park-based employment, tourism revenue sharing, non-governmental organization development projects, and resource access agreements. Although benefits positively influenced attitudes towards the park, only revenue sharing lowered illegal resource extraction. Losses were primarily caused by park-protected animals raiding crops, and preying on livestock, and loss aversion dominated attitudes about living next to the park. Eight villages accrued an annual net benefit as a result of the park, while 17 villages accrued a net loss. Losses were highest within 0.5 km of the park boundary, while benefits accrued up to 15 km from the park.

Human disturbance inside the park was spatially clustered, identifying hotspots for extraction of particular resources. Although influenced by benefit provision, illegal resource extraction from the park, particularly tree harvesting, was more strongly driven by local and urban demand and the proximity of households to the park. Household wealth was found to generally mitigate resource extraction, with the exception of livestock-based wealth. Illegal in-park livestock grazing increased where more livestock was owned.

To reduce illegal resource extraction from the park, improve attitudes towards the park, and address the spatial inequity of benefit and loss distribution, I recommend that the Uganda Wildlife Authority, in partnership with local governments and non-governmental organizations, focus benefits closer to the park boundary. In particular, they should direct revenue sharing funds, and additional funding sources that can be developed, to mitigate crop raiding and predation. To help manage the demand for wood outside the park, the authorities could incentivize farmers to grow trees, provide tree seedlings to local communities, and expand existing wood saving stove programs.

# *Résumé*

Les communautés vivant aux abords des aires protégées sont souvent imposées de façon disproportionnée des coûts reliés à la conservation mais bénéficient toutefois de l'existence de l'aire protégée. Dans cette étude, j'analyse la valeur et l'étalement géographique des pertes et bénéfices cumulées par les villages environnants le parc national de Kibale en Ouganda, afin de déterminer si ceux-ci affectent le niveau d'appropriation illégale des ressources du parc. Puisque d'autres facteurs peuvent également influencer l'appropriation illégale des ressources du parc, cette évaluation est réalisée dans le contexte démographique et socio-économique de la région.

La mesure de l'appropriation illégale des ressources a été faite à l'intérieur des limites du parc, tout près de là où se trouve 25 villages visés par cette étude. Les données relatives aux pertes et bénéfices des communautés ainsi que les données sur la démographie et les activités de subsistance, ont été recueillies via des groupes de discussion avec les villageois, un sondage auprès des familles, et d'entrevues individuelles. Les pertes dues à la destruction des cultures par les animaux sous la protection du parc ont été recueillies en mesurant physiquement les dommages causés par ceux-ci sur une période de six mois. L'influence des marchés urbains locaux est représentée ici par la distance les séparant des centres urbains.

Les bénéfices financiers pour les communautés environnantes comprennent: les emplois dans le parc, le partage des revenus dû au tourisme, les projets de développement d'organismes non-gouvernementaux et les ententes sur l'accès aux ressources. Bien que ces bénéfices influencent positivement l'opinion des communautés environnantes face au parc, seul le partage des revenus provenant du tourisme influence à la baisse l'appropriation illégale des ressources.

Les pertes sont principalement dues à la destruction des cultures et à la prédation des animaux de la ferme par les animaux sous la protection du parc. C'est l'aversion envers ces pertes qui domine dans l'opinion des communautés vivant aux abords du parc. En tout huit villages démontrent un bénéfice annuel net grâce au parc alors que 17 villages cumulent une perte nette. Les pertes les plus importantes ont été répertoriées à l'intérieur d'un cercle de 0.5 km aux abords du parc alors que les bénéfices se sont retrouvés jusqu'à 15 km du parc.

La perturbation humaine dans le parc a été localisée ainsi que l'identification des "point chauds" pour l'appropriation illégale des ressources. Malgré l'influence positive due à l'avantage financier des bénéfices, l'appropriation illégale des ressources (particulièrement l'abattage des arbres) est reliée à la demande des populations locales et urbaines et à la proximité des familles établies près du parc. Les biens cumulés par les familles ont réduit l'appropriation illégale des ressources du parc, exception faite des communautés élevant du bétail pour lesquelles le pâturage illégal des animaux dans le parc a été plus fréquent.

Afin de réduire l'appropriation illégale des ressources du parc, améliorer les comportements envers le parc et gérer la répartition équitable des pertes et bénéfices, je recommande que l'autorité de la faune d'Ouganda, en partenariat avec les organisations gouvernementales et non-gouvernementales, distribue les bénéfices aux communautés ceinturant les limites du parc. Plus particulièrement, ils devraient utiliser les fonds communs de revenu ainsi que d'autres formes de financement potentielles, afin de contrôler et réduire la destruction des cultures, et la prédation du bétail. De plus, ces autorités devraient aider à la gestion des demandes en bois en encourageant les agriculteurs à cultiver des arbres, en fournissant les semis d'arbres aux communautés locales, et en élargissant les programmes existants sur les fours 'écono-bois'.

# *Contribution of Authors*

The five results chapters in this thesis are written as journal manuscripts that have been submitted to journals for publication. The contributions of the co-authors are as follows:

***Spatial Patterns of Illegal Resource Extraction in Kibale National Park, Uganda***

Published in *Environmental Conservation* 39, 38-50.

| Author | Contribution |
|---|---|
| Catrina MacKenzie | Conducted field work, with Uganda field assistants, performed all data analysis, and wrote the manuscript |
| Colin Chapman | Field supervision of data collection, verification of tree species successional stage classification, and manuscript editing. |
| Raja Sengupta | Guided spatial analysis and data visualization |

***Demand and Proximity: Drivers of Illegal Forest Resource Extraction***

Accepted in revised form by *Oryx* (in press)

| Author | Contribution |
|---|---|
| Catrina MacKenzie | Conducted field work, with Uganda field assistants, performed all data analysis, and wrote the manuscript |
| Joel Hartter | Assisted in the development of the household survey, provided protocols for effective social research in villages around Kibale National Park, and edited manuscript. |

***Trenches like Fences make Good Neighbours: Revenue Sharing around Kibale National Park, Uganda***

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| Author | Contribution |
|---|---|
| Catrina MacKenzie | Conducted field work, with Uganda field assistants, performed all data analysis, and wrote the manuscript |

***Elephants in the Garden: Financial and Social Costs of Crop Raiding***

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| Author | Contribution |
|---|---|
| Catrina MacKenzie | Conducted field work, with Uganda field assistants, performed all data analysis, and wrote the manuscript |
| Peter Ahabyona | Supervised and audited data collection in the six month crop raiding study, translated and entered data, provided valuable insights on local community crop raiding issues, and provided comments on manuscript. |

***Accruing Benefit or Loss from a Protected Area: Location Matters***

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# 1 Introduction

## 1.1 Thesis overview

It is estimated that the world has experienced five extinction spasms throughout geologic history (Pimm & Brooks, 1997). Today, a sixth extinction spasm has started, however, unlike prior mass extinction events this one is caused by humans as a result of hunting, habitat loss, and climate change (Pimm & Raven, 2000; Barnosky *et al*., 2004). All ecosystem habitats have been affected, and are losing biodiversity as a result of human activity, although wetlands and forests have been the hardest hit (Wilson, 1999; Pimm & Raven, 2000). It is estimated that each year the world loses 13 million hectares of forest and up to 40,000 species who depend on those forests for survival (Kremen *et al*., 2000), highlighting a need for high biodiversity forest habitats to be protected (Pimm & Raven, 2000). In an attempt to counter the extinction trend, the global community has established a Convention on Biological Diversity (Convention on Biological Diversity, 2010), calling for at least 10% of each signatory nation's land area to be protected (Naughton-Treves *et al*., 2005). However, even when habitat is protected, the protected areas (PAs) continue to be threatened by anthropogenic pressure (Bruner *et al*., 2001; Mugisha & Jacobson, 2004).

Biodiversity conservation, although often argued for on intrinsic grounds of ethical or cultural values, typically rests with local utilitarian perceptions of economic benefits and losses of living with wildlife in areas where species co-exist with the ever increasing pressures of human expansion and anthropogenic change (Ninan *et al*., 2007). This dissertation endeavours to answer the question: *For the people living in communities that directly border Kibale National Park, Uganda, do benefits and losses accrued as a result of the existence of the national park, affect their conservation behaviours towards the park?*

Following an introduction to conservation management strategies, and conservation incentives, I provide a brief overview of my study site, Kibale National Park (KNP) in Uganda, and the research methodologies employed. Five manuscript-style chapters document the findings of this research where illegal resource extraction from the park is used to represent the conservation behaviours of the people living in communities bordering KNP (Bruner *et al*., 2001). Progressive contextualization (Vayda, 1983) is used to build layers of understanding about the illegal resource extraction behaviours of communities living next to KNP. In chapter 3 the illegal resource extraction, measured just inside the park boundary next to each of the 25 study villages, is quantified and compared with the household stated desire for, and admitted extraction of, park resources. In chapter 4 the socio-economic landscape of the villages is summarized, providing information on population density, adult education levels, household wealth, income sources, and market influence of the nearest urban centre, Fort Portal (population 40,988; Uganda Bureau of Statistics , 2010). These socio-economic variables are then used to model the measured off-take from chapter 3. The following three chapters turn to the benefits and losses accrued by the villages as a result of the existence of KNP. Chapter 5 presents a review of the revenue sharing program, through which Uganda Wildlife Authority (UWA) shares 20% of the gate revenues from KNP with local communities for development projects. In chapter 6 the financial and social costs of crop raiding by park protected animals are quantified. All benefits and losses accrued by local communities are discussed and valued in chapter 7. The village-valued benefits and losses are correlated against the measured resource extraction levels, and the household perceived benefit or loss of living next to KNP is compared with attitudes towards the park. Lastly, I present a conclusions chapter where the research question is answered, and the relative influence of benefits and losses on levels of illegal resource extraction from the park is contextualized relative to demographic, socio-economic, and market factors.

## 1.2 Protected areas and evolving management strategies

Throughout recorded history, humans have restricted access to certain areas of land, protecting valuable natural resources and as a consequence conserving nature. Initially, spiritual sanctions restricted access to sacred groves in India and Africa (Gadgil & Vartak, 1976; Sheridan & Nyamwero, 2007), where hunting and other resource collection was denied under the threat of religious or social retribution. Tribal and state royalty have

traditionally proclaimed areas of land for their own hunting pleasure, restricting access from the local populace (Anderson & Grove, 1987; Adams, 2004). Under colonial rule in Africa, after initial colonial settlement and ivory hunting had decimated wildlife in large areas (Adams, 2004), hunting and forest reserves were created for use by the colonial elite, prohibiting people from settlement, cultivation, and extraction of natural resources (Borgerhoff Mulder & Coppolillo, 2005). In England, the Society for the Preservation of the Fauna of the Empire (SPFE) considered the hunting practices of Africans to be inhumane and lobbied successfully to outlaw indigenous Africans from killing African game (Neumann, 1998; Oates, 1999). Although protected for the purpose of elite hunting, the reserves effectively conserved wildlife and natural habitat (Adams & McShane, 1992).

During the 1800s in the United States, the need to protect the American frontier wilderness was being espoused by artists, philosophers, and policy makers. The basis for their arguments included: the aesthetic desire to retain the beauty of the landscape and native Indian heritage (Caitlin, 1990), the desire to retain wilderness as an escape from civilization (Muir, 1912), a fundamental environmental ethic to protect the world in which we live (Leopold, 1949; Thoreau, 1992), and a utilitarian desire to scientifically manage natural resources for the use of man (Pinchot, 1910). In 1864, Yosemite became America's first reserve and in 1872, Yellowstone, the world's first official National Park, was opened (Lockwood *et al*., 2006). Counter to Caitlin's vision of protecting both wilderness and human landscape, the creation of Yellowstone resulted in the American Indians being forcibly relocated to reserves outside the national park boundaries (Borgerhoff Mulder & Coppolillo, 2005), creating a 'pristine' wilderness, supposedly untouched by humans (Cronon, 1995). Yellowstone National Park, although created to preserve scenic and natural beauty, became the protectionist model for biodiversity conservation in many other countries (Adams, 2004).

During the 20$^{th}$ century, protection efforts moved from national efforts, organized and implemented by specific countries, to global efforts to identify and help protect key areas of biodiversity throughout the world (Lockwood et al., 2006). Non-governmental organizations (NGOs) dedicated to the preservation of nature and biodiversity were founded, including: the International Union for the Conservation of Nature (IUCN, now known as the World Conservation Union), the World Wildlife Fund (WWF, now the Worldwide Fund for Nature), The Nature Conservancy (TNC), Conservation International (CI), and the Wildlife Conservation Society (WCS). In 1972, with a view to international conservation governance, the United Nations established their Environmental Program (UNEP), and international meetings were held to draft agreements on biodiversity conservation, culminating in 1992 at the Rio Earth Summit, with the Convention on Biological Diversity (Convention on Biological Diversity, 2010). Article 8 of the convention states that member countries are required to "establish a system of protected areas or areas where special measures need to be taken to conserve biological diversity" (Convention on Biological Diversity, 2010), and to manage these areas effectively. As of 2006, 168 countries had signed the Convention (Lockwood *et al*., 2006).

At the World Congress in Bali in 1982 the attending country delegates agreed that all countries should aim to protect at least 10% of their land area. There has been substantial global growth in the number of PAs, now numbering greater than 100,000 and covering 11.5% of the world's land surface (Naughton-Treves *et al*., 2005). Early parks tended to be created in more remote areas, of little development value. Today, protection is more targeted to areas of high biodiversity or areas where species are particularly threatened (Brooks *et al*., 2006). Not all protected areas are strictly protected, with 58% allowing human use (Zimmerer, 2006).

As African countries gained independence from colonial rule, many retained the reserves and national parks; however, political instability and population growth allowed for protected areas to be settled, resulting in the decline of many native species (Naughton-Treves, 1999). Pressure and donor money from western conservation organizations often lead to the forced eviction of many of these communities in the 1980s and 1990s (Feeney, 1998; Brockington & Igoe, 2006), resulting in the re-establishment of the wilderness areas as national parks, improving prospects for many critically endangered species (Adams & McShane, 1992; Struhsaker, 2002; Naughton-Treves *et al*., 2005).

Conservation policies in East Africa have favoured the protectionist approach, also known as 'fortress conservation' or 'fences and fines' (Neumann, 1998;

Brockington, 2002), where people are excluded from within the park boundaries, and human encroachment is managed by law enforcement, often by paramilitary style rangers (Terborgh & van Schaik, 2002). Fortress conservation prioritizes the intrinsic value of biodiversity (Adams *et al*., 2004), advocating top-down management of conservation areas (Buscher & Wolmer, 2007), and is viewed by many conservationists to be the most effective means of biodiversity conservation (Terborgh & van Schaik, 2002). However, strict protectionist conservation has been criticized for being socially unjust (Adams & McShane, 1992; Adams & Infield, 2003; Brockington & Igoe, 2006; Cernea & Schmidt-Soltau, 2006; Adams & Hutton, 2007), and for not recognizing that humans have long played a role in shaping the protected habitats that were now placed under exclusionary protection (Anderson & Grove, 1987). PAs tend to protect only parts of ecosystems upon which protected species depend, and when migrating outside the PA, these species do come in contact with humans (Berkes, 2004; Borgerhoff Mulder & Coppolillo, 2005). In addition, resistance by local communities to exclusionary protectionism was gaining global support (Hutton *et al*., 2005), and newly independent African countries with limited budgets and pressing issues of poverty and development were finding it difficult to justify the expense of policing the national parks (Borgerhoff Mulder & Coppolillo, 2005).

Community-Based Conservation (CBC) was developed at two World Congresses on National Parks and Protected Areas in 1982 and 1992 (Adams, 2004), stressing a need to recognize the rights of indigenous communities, the desire to increase benefits for local communities, and the reality that lack of support for conservation by local communities would eventually erode the boundaries of protected areas. CBC advocates integrated conservation and development (Barrett & Arcese, 1995), direct community management of resources (Child, 2004), appreciation of traditional environmental management (Anderson & Grove, 1987; Berkes, 2004), and at least negotiated access to PA resources (Barrow & Murphree, 2001), with the objective that "conservation goals should contribute to and not conflict with basic human needs" (Adams, 2009, p. 121). CBC encompassed many different approaches to community inclusion in conservation management (Adams & Hulme, 2001); however, there are three principle forms of CBC (Barrow & Murphree, 2001): protected area outreach, collaborative management, and community-based natural resource management (CBNRM). Protected area outreach, the primary form of CBC in East Africa, retains conservation as the primary goal, but tries to address conflicts with local communities, providing benefits to help offset losses incurred as a result of the PA. Collaborative management allows limited access to specific resources with the understanding that the local community will help police the park (Chhetri *et al*., 2003). CBNRM devolves the management of resources from the state to the community (Borgerhoff Mulder & Coppolillo, 2005), development and the improvement of local livelihoods take priority over conservation objectives (Barrow & Murphree, 2001), and focus on sustainable use of natural resources.

Outreach and collaborative management acknowledge that poverty in communities near PAs may be a constraint to conservation, and that ethically, the creation of the PA should not compromise poverty reduction (Adams *et al*., 2004). These two forms of CBC do not fully empower local communities to manage the PA, and are therefore referred to as the 'Park and Neighbour' approach to conservation management (Ghimire, 1994; Jones, 2006). CBNRM starts from the assumption that poor people are extremely dependent on natural resources and need those resources for their survival (Adams *et al*., 2004). Therefore management should be devolved to the community level because they will help conserve the resources on which they depend if they have access to sustainable harvest of those resources (Barrow & Murphree, 2001). However, CBNRM is far more complex to implement than expected because rural African communities can be large, ethnically diverse, and led by local elites (Heinen, 1996; Agrawal & Gibson, 1999). Therefore, benefit distribution can be inequitable (Fisher *et al*., 2008), inter-community conflict can be normative (Kellert *et al*., 2000), and development and conservation objectives can be inconsistent (Berkes, 2004). In fact, fears that CBC prioritizes development over conservation have led to resurgence of protectionist discourse (Oates, 1999; Terborgh & van Schaik, 2002; Hutton *et al*., 2005) and calls for more critical assessment of CBC (Buscher & Wolmer, 2007).

## 1.3 Anthropocentric benefits & losses of conservation

Although setting aside land to protect endangered species habitat is the primary focus of conservation strategies, it

must be realized that local communities can perceive the loss of access to these lands as limiting their ability to survive and at the very least limiting their ability to build economies that might have been based on natural resource extraction from, or agricultural development of, these PAs (McNeely, 1993; Czech, 2002; Adams & Infield, 2003). Adams and Infield (2003) estimated that the local people around Mgahinga National Park in Uganda were annually foregoing US$0.85 million in agricultural revenue as a result of the creation of the park. Unfortunately, the creation of PAs can also involve the forced displacement of people from the land designated for conservation. Although the World Bank now calls for this to be done in an ethical and economically equitable manner (World Bank, 2001), evictions in the name of biodiversity conservation can result in hardship and loss for those displaced (Feeney, 1998; Cernea & Schmidt-Soltau, 2006; Brockington & Igoe, 2006). Finally, animals from the PAs may roam outside of the park boundaries, trampling and eating crops, predating domesticated livestock and even maiming and killing local residents (Nyhus *et al*., 2000; Treves & Karanth, 2003). As a result, residents in communities situated on the boundaries of protected areas often disproportionately bear the cost of conservation (Nyhus *et al*., 2005; Ninan *et al*., 2007). In Tanzania and Uganda, over 86% of farmers living close to PAs reported crop raiding (Newmark *et al*., 1994; Hartter, 2007; Weber *et al*., 2007). Predation by protected species on domesticated livestock, of rural farmers in developing countries, has been estimated to cost between 12 and 52% of annual household income (Oli *et al*., 1994; Mishra, 1997; Butler, 2000; Wang & MacDonald, 2006).

Although the list of losses to local residents hampers their ability to prosper economically, there are also benefits to living next to a PA that help offset these losses. The most quoted benefit for local people is the conservation of ecosystem services upon which subsistence livelihoods depend (Millennium Ecosystem Assessment, 2005). Ecotourism, defined as low ecological impact, socially responsible travel to nature-based destinations, is the fastest growing sector of the tourism industry (Balmford *et al*., 2009). Ecotourism is becoming a very promising revenue source for many developing countries around the world, providing employment and marketing opportunities for communities near the tourist locations (Boo, 1990; Honey, 1999). Conservation authorities and local governments also benefit from the tourist industry and the foreign exchange that the PAs can attract (Adams & Infield, 2003). The international community values the protection of biodiversity and look to PAs as carbon sinks to help mitigate global warming (Grieg-Gran & Bann, 2003; Naughton-Treves *et al*., 2005). Global warming has placed specific focus on forested PAs, attracting carbon sequestration projects that can also provide employment for local people (Watson *et al*., 2000). Additional PA benefits include conservation compensation in the form of direct compensation for losses incurred (Jackson *et al*., 2008), direct payments for conservation of ecosystem services (Ferraro & Kiss, 2002), and incentive schemes such as the sharing of hunting or tourism revenues (Lewis & Alpert, 1997; Archibald & Naughton-Treves, 2001). In some cases, local communities can also benefit from negotiated access to resources inside the PAs (Chhetri *et al*., 2003).

This brief summary of the benefits and losses associated with PAs highlights the global/local dichotomy of PA benefits and losses; a majority of benefits, such as recreation and carbon sequestration, accrue to the global community while losses, such as crop raiding and predation, tend to accrue to local communities situated close to PAs (Wells, 1992). It is for this reason that financial incentives for local communities have been proposed to offset the losses of PA creation, and to make conserving biodiversity economically beneficial for local people (McNeely, 1988; Nyhus *et al*., 2005).

Exchange theory dictates that rational humans base their behavioural choices on maximizing gains and minimizing costs (Shogren *et al*., 1999; Tisdell, 2005). If true, wildlife protection requires not only public policies to conserve biodiversity, but also economic incentives to make conservation behaviours beneficial to residents close to PAs. These incentives are often referred to as conservation compensation or direct conservation payments and take many forms: return on sales from land sold for conservation (Armsworth *et al*., 2006), tax relief or compensation packages to limit land development (Main *et al*., 1999), direct payment for conservation performance (Nyhus *et al*., 2005), revenue sharing from hunting (Lewis & Alpert, 1997) and ecotourism enterprises (Archibald & Naughton-Treves, 2001), green credits for reforestation (Grieg-Gran & Bann, 2003), and loss compensation for the damage caused by wildlife (Jackson *et al*., 2008). In developing nations, incentive mechanisms are atypical, and if they exist, tend to include

hunting and tourism revenue sharing, compensation for losses incurred, and payments for conservation performance.

Conclusions on the effectiveness of these incentive programs have been mixed, with effectiveness often linked to the equity of benefit distribution (Archibald & Naughton-Treves, 2001; Adams & Infield, 2003; Spiteri & Nepal, 2006), whether the benefits are sufficient to alter behaviour (Archibald & Naughton-Treves, 2001; Kaltenborn *et al*., 2008; Spiteri & Nepal, 2008), and the empowerment of local communities to influence how benefit money is used (Agrawal & Gibson, 1999; Honey, 1999; Spiteri & Nepal, 2006). Another potential issue with conservation incentives is that rational choice (Tisdell, 2005), or that people will always do what is in their own self-interest, is assumed. Numerous studies have found that humans often make choices with bounded rationality (Conlisk, 1996; Camerer, 1998; Venkatachalan, 2008), behaviour that is anomalous with rational choice (Thaler *et al*., 1992). These anomalous behaviours include: endowment effect or loss aversion, where the individual believes they are endowed with a commodity, the loss of which is valued inconsistently higher than the acquisition value of any other equivalent replacement commodity (Thaler *et al*., 1992), preference disparity, often linked to contingent valuation, where the 'willingness to pay/accept' an environmental change often differs significantly depending on whether the individual stands to gain or lose from the change (Knetch, 2005), and non-rational reciprocal behaviour, where people have been found to forgo their own gains to punish free-loaders or be overly benevolent to those in need (Camerer, 1998; Gowdy *et al*., 2003).

Therefore, assessing the influence of accrued benefits and losses on levels of illegal resource extraction requires a broad understanding of all the benefits and losses caused by the existence of the PA, and the socio-economic factors influencing local people to risk illegally extracting resources from the PA (Heinen, 1996). This is further complicated, because conservation research requires a multi-disciplinary approach.

## 1.4 Conservation in context

Many different disciplines study biodiversity conservation. Conservation biology, a crisis-oriented discipline, started in the early 1980s, is dedicated to trying to conserve biodiversity (Soule, 1985). This field is not limited to investigation of biology and ecology, but also studies the threats to biodiversity by focusing on human demography (Ehrlich & Ehrlich, 1970) and conservation policy (Borgerhoff Mulder & Coppolillo, 2005). A related field, situated within the geography discipline is conservation biogeography (Ladle & Whittaker, 2011), which focuses on the geographic distribution of species as it pertains to biodiversity conservation (Larson & Sengupta, 2004). Practitioners of conservation biology and conservation biogeography tend to favour protectionist conservation strategies (Oates, 1999; Terborgh & van Schaik, 2002; Redford *et al*., 2006).

Since the current loss of biodiversity is being caused overwhelmingly by the actions of humans, it is not surprising that social sciences also study biodiversity conservation, but with a focus on how human institutions and human behaviour influence the relationship between people, other species and the environment. Human ecologists, ecological anthropologists, demographers, and some specialties of human and cultural geography, study the relationship of humans with their environment and more specifically, how human interdependencies influence the adaptation of human communities to changes in environment and natural resource availability (Hawley, 1950; Orlov, 1980; Steiner, 2002; Zimmerer, 2004). Their focus is on how communities have organized themselves to adapt to specific environmental factors.

Political ecologists have been among the most vocal opponents of protectionist conservation, often taking an activist role to promote community based conservation (Brockington, 2002). They focus almost exclusively on the political institutions that shape how biodiversity and habitat are destroyed or protected (Robbins, 2004), highlighting the inequities of empowerment within the context of scale (Zimmerer & Bassett, 2003): global, national, and local. Much of the political ecology literature has focused on issues of PA creation, especially eviction (Brockington & Igoe, 2006), loss of access to resources (Adams & Infield, 2003), and the power dynamics of conservation NGOs to leverage national governments to protect land for biodiversity conservation (Adams & Hutton, 2007). Generally, the political ecologists agree with the need to conserve biodiversity (Brockington *et al*., 2008), but use discursive analysis (Escobar, 1996) to critique how conservation is operationalized. They also contribute to our broader understanding of conservation by analysing the validity

and influence of different narratives to apportion blame for biodiversity loss which in turn shape perceptions about conservation.

Traditionally, little account has been given to the environmental costs of human economy. More recent economic thought includes the social costs of environmental degradation (Dixon & Sherman, 1991), supporting the need to financially value nature, through such methods as contingent valuation, so that less detrimental economic decisions can be made (Gowdy & Erickson, 2005). Ecological economics takes concern for the natural environment one step further, advocating for sustainable development or development without growth (Daly & Farley, 2004). Although it is difficult to quantify the financial value of ecosystem services and nature's intrinsic value (Tisdell, 2005), ecological economists attempt to include these environmental assets, as well as utilization value, when asking if the marginal gain in utility is worth the marginal cost (Daly & Farley, 2004; Tisdell, 2005); thus making a case for the creation and retention of PAs (Dixon & Sherman, 1991; James *et al*., 1999; Brauer, 2003).

There are shortfalls of working within only one theoretical framework when trying to understand the multifaceted context of how local communities interact with PAs. Conservation biology focuses on the threat that ever growing human population presents for biodiversity conservation (Ehrlich & Ehrlich, 1970; Struhsaker, 2002; Brook & Bradshaw, 2006; Rondinini *et al*., 2006; Burgess *et al*., 2007). By positioning human interactions within the unit of a specific community (Hawley, 1950), human ecology takes a very anthropological approach studying the interactions within a specific social group or tribe, potentially ignoring external influences from larger scales (Vayda, 1983). Conversely, political ecology starts with the hypothesis that external political factors are responsible for environmental degradation, ignoring other potential influences (Vayda & Walters, 1999). Ecological economics reduces all variables to monetary values and marginal cost curves (Tisdell, 2005), only recently accounting for irrational human behaviour (Gowdy & Erickson, 2005; Venkatachalan, 2008).

Biodiversity conservation has unfortunately created two opposing positions: protectionism and community based conservation (Redford *et al*., 2006). The perspectives each camp holds is based largely on their research discipline, with social scientists generally supporting community based conservation and natural scientists supporting protectionist strategies for PAs (Adams, 2004). Continuing to pit two sides against each other detracts from finding practical solutions to improve biodiversity conservation and the livelihoods of people who live near PAs. Realizing this, there have been numerous calls to take a more holistic approach to studying PAs and biodiversity conservation that step outside disciplinary biases (Agrawal & Redford, 2006; Redford *et al*., 2006; Wilkie *et al*., 2006). However, this requires a more complex, and logistically challenging, research design to include multi-disciplinary perspectives.

This dissertation attempts to take a broader view. I use an approach known as progressive contextualization defined as "focusing on significant human activities or people-environment interactions and then explaining these interactions by placing them within progressively wider or denser contexts" (Vayda, 1983, p. 265). Although the avenues of investigation are guided by prior conservation research, the approach is not restricted to the research direction or methodologies used by any one discipline (Borgerhoff Mulder & Coppolillo, 2005). Progressive contextualization is not limited to any specific unit or scale, allowing the research to investigate influences of local, national, or even global origin (Vayda, 1983). Also, the aim of this approach is to provide research results that are of practical use to local PA managers. However, since the avenues of investigation are informed by site-specific findings (Vayda, 1983) this may limit the generalization of some policy findings to other PAs.

I start by measuring the level of illegal extraction of resources from Kibale National Park, and then search for predictor variables that model the measured level of illegal extraction. The predictor variables include the benefits and losses accrued by communities living next to the park as well as socio-economic factors such as demographics, local livelihood activities, and market economics. I believe the methodology and findings in this dissertation provide a means for assessing the effectiveness of the conservation incentives and disincentives that exist as a result of PAs, and that the specific findings for Kibale National Park will help guide the evolution of conservation policies within the Ugandan rural socio-economic context and beyond.

# 2 Study Site and Methods Overview

## 2.1 Kibale National Park

Kibale National Park (KNP) is located in western Uganda, in the Albertine Rift Valley near the Rwenzori Mountains (Fig.2-1). It was gazetted as a National Park in 1993, although before that date, the area was a Forest Reserve and Game Corridor (Struhsaker, 2002). The park covers 795 km² and is a particularly rich area for primate biodiversity (Struhsaker, 1997). Although some areas were originally harvested for timber products until the mid 1970s, all commercial logging has now been stopped in KNP (Struhsaker, 1997), with the exception of limited paid access agreements with UWA to extract exotic tree species. However, illegal extraction of resources from the park continues as evidenced by animal poaching and harvesting of trees for building materials, fuel wood, and charcoal production (Struhsaker, 2002; Naughton-Treves *et al*., 2007; PAWAR, 2009).

Most local inhabitants are subsistence farmers, although additional income is available from cash crops and off-farm work on tea plantations, as research assistants, in the tourism industry, and planting trees for carbon sequestration (Mulley & Unruh, 2004; Hartter, 2010). Households tend to be affiliated with the Batooro tribe in the north and the Bakiga tribe in the south, although inter-tribal marriage is common and many households reported affiliation with two or more tribes. All cooking and heating, as well as charcoal, alcohol, and brick production is fuelled by wood (Naughton-Treves *et al*., 2007). Rural households also depend on KNP for craft materials, medicinal plants, and as places to put beehives for honey production (Hartter, 2010). Although the boundaries of KNP remain intact, the landscape around KNP is becoming denuded of tree cover (Hartter & Southworth, 2009). As wood becomes scarce, households are planting trees (typically exotics), stopping neighbours from accessing trees on their property, and entering KNP to collect firewood and poles (Hartter *et al*., 2011). In a prior survey conducted in 2000, 31.7% of the people admitted to entering the park to gather resources (Mugisha, 2002).

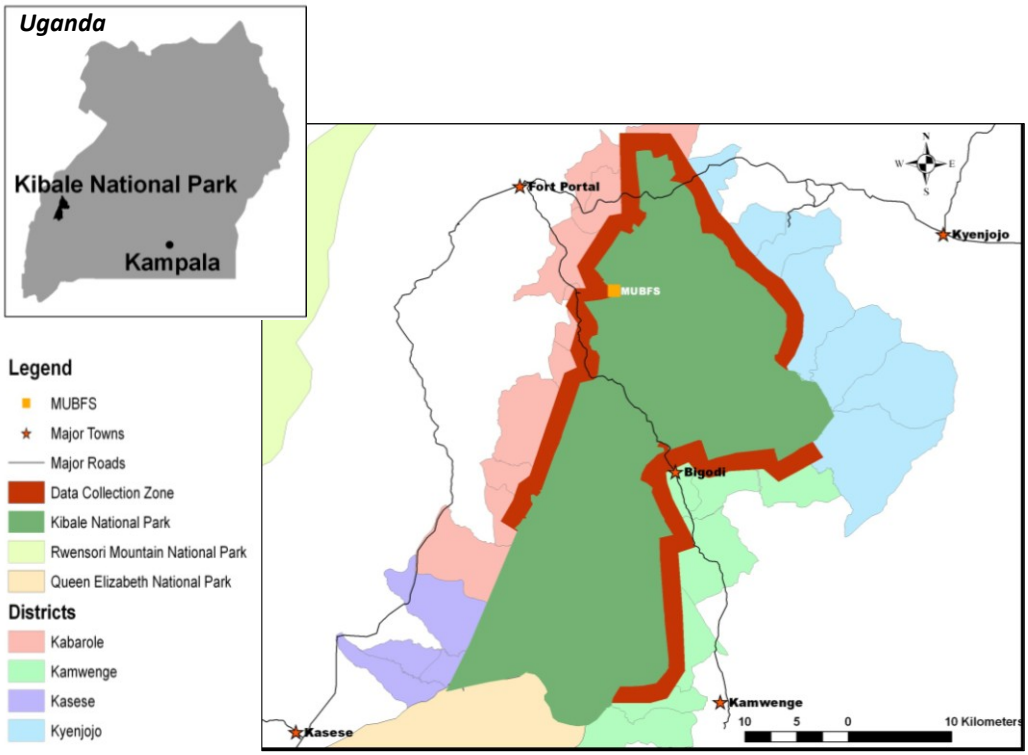



**Figure 2-1: Site map of Kibale National Park, Uganda**

Uganda Wildlife Authority manages KNP using a 'Park and Neighbour' conservation strategy (Jones, 2006), including seven specific components: 1) strict monitoring and enforcement of the boundaries, 2) restoration of the park by a carbon sequestration program (FACE the Future, 2011), 3) conservation research, 4) community and education outreach, 5) negotiated resource access for community associations (Chhetri *et al*., 2003), 6) encouraging tourism (Naidoo & Adamowicz, 2005), and 7) sharing 20% of the park entrance fees with local communities (Archibald & Naughton-Treves, 2001).

Conservation of KNP is assisted by the dedication of many researchers to long-term research programs (Wrangham & Ross, 2008). Much of the research conducted in and around KNP has focused on primate ecology (*c.f.* Snaith & Chapman, 2007; Twinomugisha & Chapman, 2007; Kahlenberg & Wrangham, 2010), aquatic conservation and fish morphology (*c.f.* Chapman *et al*., 2004), disease transmission between park animals, domestic animals and humans (*c.f.* Goldberg *et al*., 2007; Goldberg *et al*., 2008), and forest ecology and restoration (*c.f.* Struhsaker, 1997; Omeja *et al*., 2009; Chapman *et al*., 2010). In support of community conservation policy development, other researchers have studied the relationship between KNP and the human communities around the park (Hartter, 2009; Hartter & Goldman, 2009; Hartter, 2010; Hartter & Goldman, 2011), historical access rights to wild animals (Naughton-Treves, 1999), the influence of crop raiding by park protected animals on local communities (Naughton-Treves, 1998; Hartter, 2009), the use of, and access to, woody biomass in local productive practices (Naughton-Treves *et al*., 2007; Hartter, 2010), and the poaching of wild animals (Solomon *et al*., 2007).

Specific studies of UWA conservation policies included a promising assessment of the initial implementation of the revenue sharing program, although concerns were raised as to whether the most vulnerable were going to benefit from the program (Archibald & Naughton-Treves, 2001), and a favourable report on community support for resource access agreements (Chhetri *et al*., 2003). Another study compared people's attitudes towards the state-led, protectionist conservation strategy, used for KNP, with a CBNRM project in an adjacent wetland outside KNP near Bigodi (Fig. 2-1). People were more positive about conservation when the CBNRM approach was used (Lepp & Holland, 2006). Most of the social studies have occurred along the north western boundary of the park, or near the tourist centre of Bigodi (Fig 2-1). This dissertation considerably extends the spatial extent of knowledge about the interaction of local communities with KNP, and the influence of the revenue sharing program, resource access agreements, and crop raiding, encompassing 25 communities within the data collection zone (Fig. 2-1).

Using a constructionist framework of social scale (Marston, 2000; Campbell, 2007), KNP conservation is influenced by activities that occur at global, national, regional, and local scales. Funding to support the management of the park comes from international tourist revenues, international donors, and the national government (Uganda Wildlife Authority Annual Report, 2007). A carbon sequestration program based in the Netherlands has partnered with UWA to reforest areas of the park (Watson *et al*., 2000; FACE the Future, 2011) and numerous globally funded Non-Governmental Organizations (NGOs) provide benefits to people around the park. Conservation policy is defined through national governmental legislation (Uganda Wildlife Statute, 1996), managed regionally by the park wardens, but enforced at the local level through individual arrests and fines when people illegally enter the park. Revenue sharing benefits are distributed through regional and local councils, their decisions influenced by the village chairperson speaking on behalf of household members in their villages (Ssempala, 2008). Crop raiding and predation losses affect individual households, primarily those closest to the park (Naughton-Treves, 1998), and the decision to poach or illegally take wood from the park is made at the household level. Ultimately, the effectiveness of conservation strategy for KNP is a question of relational influence between multiple scales (Zimmerer & Bassett, 2003).

This study focuses on the local village scale, accounting the benefits and losses within a local socio-economic and demographic context, while acknowledging the global source of many of the benefits. Although it is recognized that African rural communities are very heterogeneous (Agrawal & Gibson, 1999) and that benefits and losses accrue differentially among households (Brockington *et al*., 2008), many community projects accrue at the village scale, and illegal resource extraction, measured inside the park near study villages, is a village-aggregate value that cannot be attributed to any

one household. Therefore, the village has been chosen as the unit of analysis for this research, while recognizing that the decision to extract resources from the park is made by the individual.

In Uganda, the local governance structure evolved from the Resistance Council System set up by the guerrilla resistance to the Obote government (Saito, 2003). The National Resistance Movement (NRM) took power in 1986 and used the Resistance Councils as local governance for the regions. By 1997 the Resistance Councils were converted to Local Councils (LC) and government administration reported to the LCs. Five levels of LC make up the hierarchical governance structure: The highest level is the district council (LC5), followed by the county council (LC4), the sub-county council (LC3), the parish council (LC2), and finally the village council (LC1) (Saito, 2003). All of the council members are elected, but are aligned with national government policy. Although authorization to carry out this research was provided at the national level from the Uganda National Council of Science and Technology (UNCST) and UWA, I also sought authorization from the Regional District Commissioners (RDC), district chairpersons (LC5), sub-county chairpersons (LC3), and most importantly the village council chairpersons (LC1) before commencing research in any village.

Four government districts surround KNP and benefit from tourist proceeds through the revenue sharing program: Kabarole, Kamwenge, Kyenjojo, and Kasese (Fig. 2-1). Only three districts were included in this study. Kasese District was excluded based on: proximity to and influence of Queen Elizabeth National Park (Fig. 2-1), having only received 8% of the revenue sharing funds distributed to date by KNP (Ssempala, 2008), having few villages directly adjacent to the park since a prison farm and army training grounds occupy large areas next to the park boundary, and due to logistical constraints of travelling from my research base in the Makerere University Biological Field Station (MUBFS; Fig. 2-1) to villages in Kasese district. Study villages were limited to a 2.5 hour drive from the research station to allow for at least 5 hours of work per day in the farthest study villages. This resulted in a 128 km data collection zone around the perimeter of the park (Fig 2-1). Within the data collection zone, 25 villages volunteered to participate. For this study, a village is defined by the spatial extent of households associated with a village name under the leadership of one village chairperson. Study villages were located approximately 5 km apart, and were chosen based on some village members owning and/or cultivating land directly adjacent to the park boundary.

Since this study deals with illegal extraction, there is the potential for local villagers to be punished for illegal entry into the park should the study villages be explicitly identified (Robbins *et al*., 2006). Thus, geographic masking (Leitner & Curtis, 2004; Armstrong & Ruggles, 2005) was used to protect village identity. Although spatially specific data was used for analysis, graphic representations of data and results are interpolations, using inverse distance weighting or Kreiging (Childs, 2004) to a 1000m raster grid, re-sampled to a resolution of 30m within the data collection zone (Fig.2-1). The legends in these graphic representations indicate the actual minimum and maximum values of village-specific data.

## 2.2 Methods overview

### 2.2.1 Data collection

Details of the data collection methods employed are presented in each of the manuscript chapters. However a brief overview is presented here to illustrate how the various datasets integrate into the manuscript chapters (Table 2-1).

Information about the benefits and losses associated with living next to KNP and the demographic and socio-economic landscape was collected using focus groups, a household survey, and interviews. Focus groups were conducted in 15 study villages (60%) in June and July 2008. The village chairperson, as per cultural norms, approved and organized the meetings, inviting men and women from a range of age groups representing a cross-section of the village. The number of participants ranged from 16 to 51, with women representing up to 65% of the participants. After agreeing to the verbal informed consent, focus group participants were asked to list all problems of living next to KNP and then all benefits. Specific questions about the revenue sharing program were asked to understand whether people were aware of

**Table 2-1: Data collection overview for results chapters**

| | *Illegal Resource Extraction* | *Focus Groups* | *Household Survey* | *Interviews* | *Six Month Crop Raiding Study* |
|---|---|---|---|---|---|
| **Chapter 3: Spatial patterns of illegal resource extraction in Kibale National Park, Uganda** | Trees,<br>Entry Trails,<br>Grazing,<br>Poaching | Comments on activities in the park,<br>Comments on negotiated access | Desired resource access,<br>Admitted resource access,<br>Income generating productive practices,<br>Crop raided (yes/no),<br>Periods of food insecurity,<br>Energy saving stove(yes/no),<br>Perceived benefit of resource extraction | UWA: Resource access agreements (RAA)<br>RAA Chairpersons:<br>Scope of RAA | |
| **Chapter 4: Demand and proximity: Drivers of illegal forest resource extraction** | Trees,<br>Entry Trails,<br>Grazing,<br>Poaching | | Household composition,<br>Head-of-household birth place,<br>Education attained,<br>Capital wealth assets,<br>Income generating activities | | |
| **Chapter 5:Trenches like fences make good neighbours: Revenue sharing around Kibale National Park, Uganda** | Trees,<br>Entry Trails,<br>Grazing,<br>Poaching | Knowledge & perception of revenue sharing | Attitudes towards the park,<br>Knowledge of revenue sharing projects,<br>Perceived benefit of revenue sharing projects | UWA, District, Sub-county & Village chairpersons:<br>Revenue sharing program (projects & process) | |
| **Chapter 6: Elephants in the garden: Financial and social costs of crop raiding** | | Perceived problems caused by crop raiding | Crop raided (yes/no),<br>How troubled by crop raiding,<br>Fraction of crop yield lost,<br>Household composition,<br>Days spent guarding crops,<br>Crop defence methods,<br>When crop raiding is worse,<br>Self reported human & livestock disease,<br>Income generating activities,<br>Capital wealth assets | Primary school head teachers: Scholastic achievement data | Number of incidents,<br>Species of raider,<br>Area damaged, crop yield & valuation data |
| **Chapter 7: Accruing benefit or loss from a protected area: Location matters** | Trees,<br>Entry Trails,<br>Grazing,<br>Poaching | Problems & benefits of living next to KNP,<br>How do benefits compare with problems? | Rate perceived benefits (or troubles) of living next to KNP,<br>Describe & quantify benefits gained by household,<br>Describe & quantify losses caused by KNP,<br>Attitudes towards KNP & UWA | Financial value: employment by UWA, tourist, FACE & research managers<br>Financial value: resource access agreements<br>Financial value: benefits from NGO managers<br>Financial value: UWA revenue sharing | Value of crop raiding losses |
| **Chapter 8: Conclusions** | Trees,<br>Entry Trails,<br>Grazing,<br>Poaching | | Household composition,<br>Head-of-household birth place,<br>Education attained,<br>Capital wealth assets,<br>Income generating activities,<br>Benefit & loss values | Benefit and loss values,<br>Village chairperson perceived benefit of revenue sharing | |

the program, and at the end of the meeting we asked if the disadvantages of living near the park were offset by the benefits received. Discussion was facilitated by a senior field assistant in the local tribal languages, translating the responses verbally during the meeting, while another assistant recorded all that he heard from the villagers, allowing triangulation of notes for a more complete record of the meeting.

In July and August 2009, a household survey was conducted in all 25 villages. No sampling frame existed for the villages, so all village households had to be mapped while noting the construction standard of primary dwellings. The number of households per village ranged from 41 to 242 with 64% of the villages having less than 100 households, 20% having 100 to 150 households, and the remaining 16% having over 150 households. Within each village 24 households were surveyed (10% to 59% of the village households) representing 24% of all households in all villages (596 of 2480). The sample size of 24 households had been based upon the 2008 GPS mapping of 21 study villages in Kaberole and Kamwenge districts (41 to 154 households per village). National authorization to extend the data collection to Kyenjojo district was provided in April 2009 and GPS mapping of the four villages added in Kyenjojo could only start in July 2009, after local authorization was given by village chairpersons in May and June 2009. By this time, household survey data collection had already been completed for 12 villages in Kaberole and Kamwenge. Therefore the disproportional size of villages in Kyenjojo (116 to 242 households) was found too late to increase the sample size for the larger villages considering logistic constraints. Therefore, the mean village sampling error, or the expected variability of different samples taken from the same population, increased to 13% from the originally targeted 10%.

The distribution of survey households as a function of the distance of the household from the park boundary also appears skewed towards the park boundary (Fig. 2-2). However, this is not a sampling bias, but a geographic artefact of the spatial extent of the villages. Some villages extended along the park boundary, while others extended away from the boundary (Fig. 2-3), resulting in the outer boundary of the village extent varying from 873 to 4520 m (mean 2043 m) from the park edge. Since all villages were directly adjacent to the park, this resulted in a heavier concentration of households closer to the park. Correlations against 'distance to park' therefore ran the risk of being skewed. To validate that significant 'distance to park' correlations represented a real trend, the survey responses for the correlating variable within 500 m radial buffers of the park boundary were compared with the distribution of households within those same buffers using a $\chi^2$ test prior to reporting the result as significant.

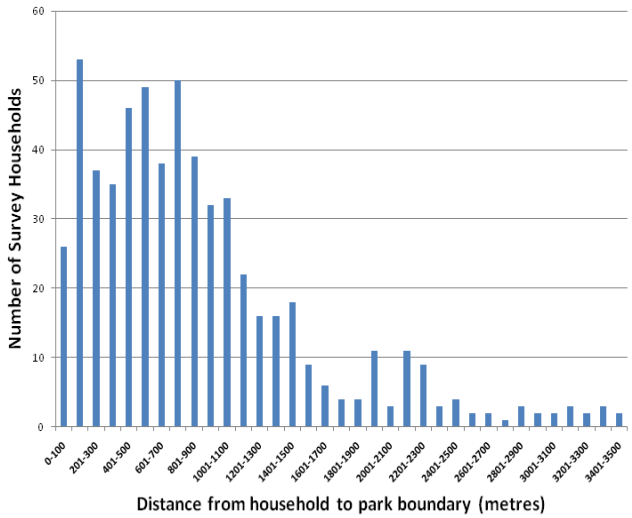



**Figure 2-2: Distribution of survey households relative to park boundary**

Household distance to park: mean = 882 m, median = 732 m, standard deviation = 695 m

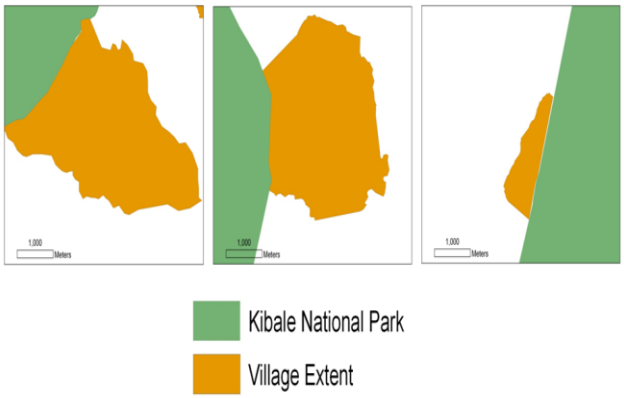



**Figure 2-3 : Typical spatial extent of study villages**

The households were chosen by random stratified sampling, with stratification based on house construction standard (Ellis & Bahiigwa, 2003; Hartter, 2009), a proxy for household wealth. Although stratification could have been done against other factors, such as household size, distance from the park or education attained, the literature stressed that poorer households were more dependent on natural resources and were least likely to be represented in surveys (Scoones, 1995). The survey was administered by four Ugandan field assistants, three men and one woman, in two local languages (Rutooro and Rukiga). Since literacy rates in the region are low, a verbal informed consent was agreed with the participant prior to the 1.5 hour survey being started.

Structured interviews, also preceded by a verbal informed consent, were conducted from May 2008 to January 2010 with UWA wardens, district, sub-county and village chairpersons, resource access agreement association chairpersons, and managers of tourist facilities, research operations, NGOs and FACE the Future (FACE the Future, 2011). These interviews collected detailed information about the benefits accrued around KNP.

The most reported loss incurred by households near KNP was the raiding of their crops by park protected animals. Since people tend to overestimate losses due to crop raiding in the hopes of receiving compensation (Tchamba, 1996), data were collected to physically verify the damage incurred. The type of crop and area damaged was recorded weekly for six households in each study village for six months from August 2009 to January 2010 to more accurately value the financial loss.

### 2.2.2 Analysis

Each manuscript chapter describes the methods used for analysis. Therefore, this section will only describe the path analysis or Structural Equation Modelling (SEM) used in the conclusion chapter to understand how or even if the benefits and losses accrued by the 25 villages influence the levels of illegal off-take from the park when tested alongside the contextual variables representing local productive practises, demographics and the socio-economic landscape. SEM is a means of assessing relative influence of each parameter in a studied system (Keith, 2006), however, unlike regression analysis, SEM allows the interrelationships between observed variables to be included based on relative linear covariance. To build the model, causal relationships between observed variables must be supported by theory or prior research (De Vaus, 2002).

The observed variables were collected in each of the 25 villages (n=25), however, the potential variables to be included in the model, as described in section 8.1, totalled 23 (five benefit variables, three loss variables, five productive practise variables, three demographic variables, and seven socio-economic variables). To build a viable model, a down-selection process had to be used to reduce the potential variables in the model. Linear regression with backward variable elimination ($p<0.1$) was used to identify the variables that were significantly influencing the illegal extraction, and then the number of variables was further reduced during the path analysis to ensure that the model exhibited good goodness of fit (Fig. 2-4); analogous to using the Akaike (AIC) or Bayesian (BIC) Information Criteria in general regression models (Burnham & Anderson, 2002), but optimizing goodness of fit measures specific to structural equation modelling. A good model fit requires Chi squared ($\chi^2$) to be relatively small with a non-significant probability ($p>0.1$; Keith, 2006). Since $\chi^2$ is influenced by sample size, four other fit measures were used to determine that an adequate model had been developed. The Goodness of Fit Index (GFI) is analogous to $R^2$, the Comparative Fit Index (CFI) is an estimate of how much better the model fit is than a null model, the Tucker-Lewis Index (TLI) is more independent of sample size as it corrects for the degrees of freedom in the model, and the Root Mean Squared Error of Approximation (RMSEA) represents "the degree of misfit per degree of freedom" (Keith, 2006, p. 270). GFI, CFI, and TLI should be as close to 1.0 as possible, and preferably above 0.9 (Tanaka, 1993; Hu & Bentler, 1999; Keith, 2006), while RMSEA should be <0.05 for a model to be considered very good, and <0.08 to be considered adequate (Brown & Cudeck, 1993; Hu & Bentler, 1999).

Comparing the sign and relative magnitude of path coefficients between observed variables permits a relative ranking of the influence of each of the remaining variables on the illegal resource extraction variable. This is of particular interest for conservation managers, as it allows intervention strategies to focus on the factors that most strongly influence the particular illegal resource extraction.

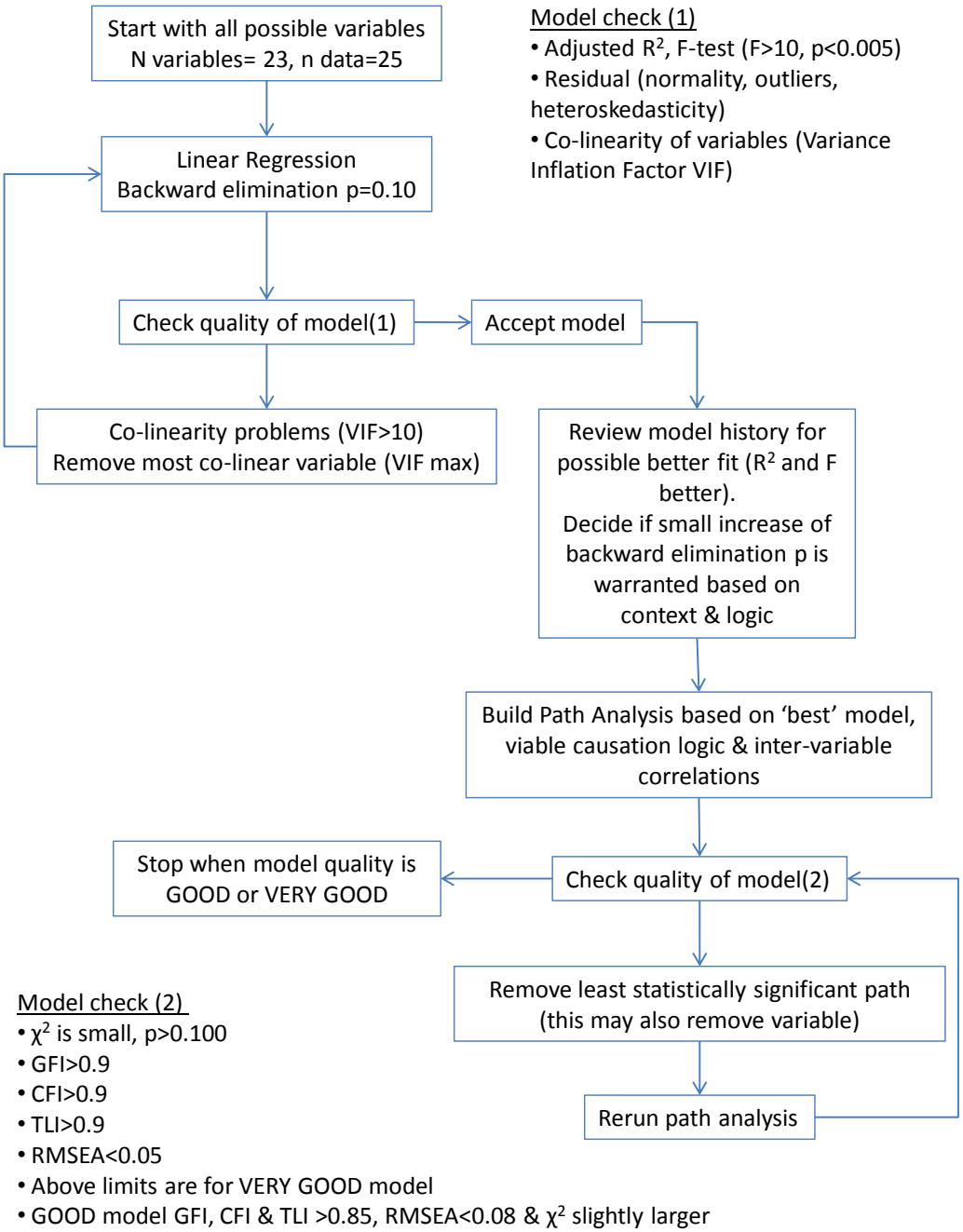



**Figure 2-4: Structural equation modelling flowchart**

# 3 *Spatial Patterns of Illegal Resource Extraction in Kibale National Park, Uganda*

In this chapter I describe the illegal resource extraction measured around Kibale National Park, comparing it with the stated desire for resources and the admitted access to resources by village survey respondents. The influence of resource access agreements to reduce illegal off-take is also examined. This chapter was first published as MacKenzie, C.A., Chapman, C.A., & Sengupta, R. (2011)"Spatial patterns of illegal resource extraction in Kibale National Park, Uganda", *Environmental Conservation* 39(1), 38-50 Copyright © 2011 by *Environmental Conservation* and reproduced here by permission.

**SUMMARY:** Conservation policy typically excludes people from national parks and manages encroachment by law enforcement. However, local people continue to extract resources from protected areas by boundary encroachment and poaching. This paper quantifies the patterns of illegal resource extraction from Kibale National Park in Uganda, the demand for park resources by communities bordering the park, and examines whether designated resource access agreements reduce illegal extraction. Sections of the park boundary were examined and human entry trails, wood extraction, livestock grazing, and animal poaching signs were quantified. Levels of illegal extraction were compared with the demand for and admitted illegal access to resources inside the park, collected in a survey of households located near the park. Extraction was also compared between villages with and without negotiated resources access agreements. The most wanted and extracted resource from the park was wood for fuel and construction. Implementation of resource access agreements with local community associations was found to be an effective means of reducing illegal extraction, but only if the association members profited from the agreement.

## 3.1 Introduction

Biodiversity conservation is often characterized by two polarized narratives: the 'parks' position looks to exclude people from protected areas (PAs) (Terborgh & van Schaik, 2002), and manages encroachment by law enforcement, whereas the 'people' position demands access and management rights for local residents (Neumann, 1998; Adams & Hulme, 2001). The 'parks' position, alternatively known as 'fences and fines' or 'fortress conservation' (Neumann, 1998; Brockington, 2002), developed from the colonial creation of PAs in Africa (Neumann, 1998; Oates, 1999) and the American 19th century environmental protection movement (Brockington, 2002). The 'parks' position prioritizes the intrinsic value of biodiversity (Adams, *et al.,* 2004) and advocates top-down management of conservation areas (Buscher & Wolmer, 2007). The 'people' position developed from social injustices resulting from PA creation, such as forced eviction (Brockington & Igoe, 2006; Cernea & Schmidt-Soltau, 2006), human wildlife conflict (Treves & Karanth, 2003), and losses incurred by communities living adjacent to PAs (McNeely, 1993; Adams & Infield, 2003), as well as resistance by communities to the exclusionary 'parks' position (Hutton *et al*., 2005). To counter these injustices, community-based conservation (CBC) was introduced (Adams & Hulme 2001), which advocated integrated conservation and development (Barrett & Arcese, 1995), direct community management of resources (Child, 2004), and at least negotiated access to resources (Barrow & Murphree, 2001). In return, communities help protect the park from illegal activities (Adams & Hulme, 2001; Archibald & Naughton-Treves, 2001; Chhetri *et al.,* 2003; Child, 2004).

Either extreme in the people or parks arguments minimizes the interdependency between PAs and local people (Redford *et al.,* 2006). Exploitation of the park purely for resource extraction would lead to the decline of critically endangered species, loss of ecosystem services for local communities and potentially loss of tourist revenues (Naughton-Treves *et al.,* 2005; Burgess *et al.,* 2007). Fears that CBC prioritizes development over conservation have led to resurgence in protectionist discourse (Oates, 1999; Terborgh & van Schaik, 2002;

Hutton *et al.,* 2005) and for more critical assessment of CBC (Buscher & Wolmer, 2007). Ideally, conservation policy needs to consider all PA stakeholders: conservation organizations that value the protection of biodiversity and look to protect carbon sinks to combat global warming (Naughton-Treves *et al.,* 2005; FACE the Future, 2011), conservation authorities and local governments who need the tourists and foreign exchange that PAs can attract (Adams & Infield, 2003), and local people who need resources inside PAs to support subsistence livelihoods (Naughton-Treves *et al.,* 2007).

Conservation policies in East Africa have favoured the 'parks' approach, however, people continue to illegally extract resources from PAs (Robbins *et al.,* 2006; Holmes, 2007). Trees are used as fuel wood for cooking, and heating (Naughton-Treves *et al.,* 2007), protected grasslands are coveted by livestock herders, especially in times of drought and as pasture land becomes scarce (Neumann, 1998; Infield *et al.,* 1993; Kideghesho *et al.,* 2007), and wild animals are hunted for meat (Neumann, 1998; Chapman *et al.,* 2006). Rural households depend on tropical forests for craft materials, medicinal plants, and as places to put beehives for honey production (Adams & Infield, 2003; Mbile *et al.,* 2005; Bleher *et al.,* 2006). Lack of access to PAs requires local residents to find or procure these resources elsewhere, which can be costly (Emerton, 1999). However, illegal extraction of natural resources, and wildlife poaching, can lead to loss of habitat and further species endangerment (Chapman *et al*., 2006), resulting in the productive practices of local communities being perceived as threats to conservation (Mbile *et al.,* 2005).

Studies have measured human disturbance inside PAs (Barve *et al.,* 2005; Bleher *et al*., 2006; Baranga, 2007; Olupot *et al*., 2009); however, most focused on ecological conservation by monitoring extraction without linking the drivers of disturbance to the productive practices of local communities. Alternatively, most social studies on conservation management rely on attitudes of local people towards PAs without linking these attitudes to measured illegal resource extraction (Infield, 1988; Gillingham & Lee, 1999; Mugisha, 2002; Gadd, 2005; Mbile *et al.,* 2005; Kideghesho *et al*., 2007; Hartter & Goldman, 2011). These studies link the demand for PA resources with socioeconomic factors such as population density (Burgess *et al.,* 2007), household well-being (Infield, 1988; Gillingham & Lee, 1999), education (Kideghesho *et al.,* 2007), and enforcement capacity (Abbot & Mace, 1999).

This research does not try to explain the drivers of demand for resources, but aims to describe the patterns of measured illegal resource extraction from Kibale National Park in Uganda, link this extraction to the stated demand for park resources, and investigate if negotiated resource access agreements reduce illegal extraction from the park.

## 3.2 Methods

### 3.2.1 Study site

Kibale National Park (KNP) is a 795 $km^2$ protected mixed evergreen forest and savannah grassland located in western Uganda (Fig. 3-1). Prior to 1993 KNP was a game corridor and forest reserve, selectively used for logging and exotic tree plantations (Struhsaker, 1997). In 1971, a 15-year long war started, during which *c.* 55,000 people settled inside the forest reserve and game corridor (Naughton-Treves, 1999). After the war, the Uganda Forest Department reclaimed forest reserves throughout Uganda, evicting approximately 35,000 people from Kibale Forest Reserve and Game Corridor (Feeney, 1998; Naughton-Treves, 1999), although estimates of the actual number evicted vary greatly (Chapman & Lambert, 2000). In 1993 Kibale became a national park. Evictees were eventually resettled far from KNP (Feeney, 1998) therefore few current residents around KNP personally experienced eviction (Hartter & Goldman, 2011).

Conservation policy, defined through government legislation (Uganda Wildlife Statute 1996) is implemented by the Uganda Wildlife Authority (UWA). The core policy is exclusion, tempered by limited CBC, such as access agreements for specific resources (Chhetri *et al.,* 2003), conservation education and sharing 20% of park entrance fees to benefit people living next to the park (Archibald & Naughton-Treves, 2001). UWA also partner with FACE the Future to reforest areas cleared by settlers (FACE the Future, 2011), sometimes employing over 300 people to plant indigenous trees.

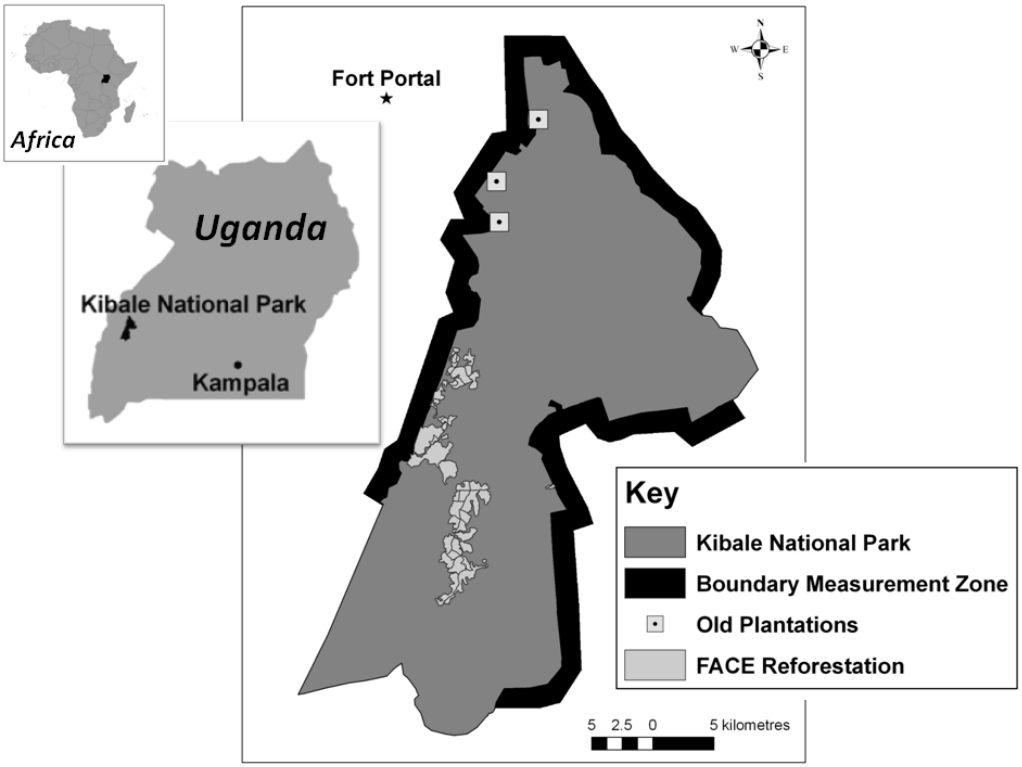



**Figure 3-1: Location and diagrammatic map of Kibale National Park, Uganda**

Old Plantations: Exotic tree plantations created when the park was a forest reserve.

FACE Reforestation: Areas where FACE foundation has been planting indigenous trees for carbon sequestration.

There are two forms of permitted resource access: resource access agreements (RAA) and memoranda of understanding (MOU). RAAs are legal agreements entered into by an association. In return for access, members promise to protect the park by managing their own activities so as not to endanger animals, by reporting unauthorized people inside the park and by helping to sensitize other villagers about conserving KNP. Current agreements permit keeping beehives in the park (four), collection of craft materials (one), and fishing (two). In the past there were also agreements for firewood and/or non-timber forest product (NTFP) collection (three), watering cattle (one) and picking wild coffee (one), but these were discontinued due to the exhaustion of the resource (two firewood), association members caught in non-compliance of park rules (one cattle watering and one NTFP), or because the association that had entered the agreement was disbanded (one wild coffee). In the seven active agreements, association members typically came from one to seven villages.

Communities around KNP are primarily subsistence agriculturalists from two dominant ethnic groups: Batooro to the north and Bakiga to the south. Between 2000 and 2006, UWA found 233 poaching signs and 272 signs of encroachment while on patrols inside KNP (PAWAR [Protected Area Watch for the Albertine Rift] 2009). Although patrol data can be biased by patrol location and intensity (Gavin *et al.,* 2010) this does indicate that violations of park rules persist. In a survey of residents close to KNP, 31.7% of people admitted entering the PA to gather resources (Mugisha, 2002).

Although official permission to do this research had been provided at the national level, written authorization was also sought from the Regional District Commissioner and District Chairperson in the three districts where we collected data, and from the Sub-county and Village Chairpersons responsible for the 25 villages in our study. The Sub-county Chairpersons provided lists of villages located next to the park, from which we chose our study villages. Our village selection was based on village members holding and/or cultivating land directly adjacent to the park and geographic location; we chose an approximate spacing between village centres of five

kilometres to evenly distribute the study villages along the 128 km of the boundary measurement zone (Fig.3-1).

### 3.2.2 Field methods

Studies to measure human disturbance in tropical protected forests in India, Kenya and Uganda (Barve *et al.,* 2005; Bleher *et al.,* 2006; Baranga, 2007; Olupot *et al.,* 2009), recommended measuring trees cut for any purpose (all sites), signs of poaching (two sites), evidence of burning (two sites), in-park cultivation (three sites), pits to saw timber (one site), livestock grazing (four sites), charcoal-making (two sites) and extraction of a wide range of non-timber forest products (three sites).

Using a method developed from a study in Bwindi Impenetrable National Park, Uganda (Olupot *et al.,* 2009), an observational transect 600–850 m in length was conducted along the boundary between each village and the park. The boundary was approached via a village path or through cultivated land. Human entry trails into the park were counted as a measure of access for resources extraction. Each trail was followed to its terminus, as determined by our Ugandan field assistants, unless the trails were created for research or UWA patrols, as confirmed by UWA. While walking the boundary and trails, we recorded poaching signs (snares and pit-fall traps), the number and species of grazing livestock inside the park, the number of charcoal-making operations in or directly adjacent to the park, boundary encroachment for cultivation, and evidence of fire inside the park. We counted harvested trees, measured the diameter at the cut location and where possible, skilled assistants, identified the species of tree and approximately how long ago harvesting had occurred. We recorded resource extraction within 5 m either side of the trail, but found that extraction quickly diminished within 3 m. Since effective boundary demarcation has been correlated with effective PA management (Bruner *et al.,* 2001), we also noted whether any harvested trees had been planted to demarcate the boundary.

Household surveys were carried out in all 25 villages to quantify the demand and admitted extraction of specific resources from the park. Resource extraction is illegal so responses about extraction were prone to non-response and social undesirability biases (St. John *et al.,* 2010). Innovative methods, such as the randomized response technique (Solomon *et al.,* 2007; St. John *et al.,* 2010) and the nominative technique (St. John *et al.,* 2010) have been tried to capture illegal extraction behaviours and to account for these biases. So far, randomized response has been the most successful, but requires more survey time, larger sample size, increased enumerator training and may result in respondents thinking they are being tricked (Gavin *et al.,* 2010).

Given the logistic challenge of reaching 25 villages, we opted for direct questioning, but attempted to identify an upper bound to actual levels of extraction by asking, 'which park resources would you like to have access to?', while capturing a lower bound asking about admitted extraction from the park. We employed a format that empathized with the respondent for the later question, as this can elicit a more honest response (Blair *et al.,* 1977): 'People need to survive and sometimes the park has the resource we need to survive. For the following resources, how often do you get them from the park: never/rarely/sometimes/often/always?' Participants were specifically asked about access for beehive placement, and extraction of firewood, construction poles, exotic trees, medicinal plants, grasses, water, wild coffee, bushmeat and fish, although respondents were encouraged to report any other resources they wanted from the park.

No sampling frame existed for the villages, so all households were mapped while noting the construction standard of primary dwellings. The number of households per village ranged from 41 to 242 with 64% of villages having less than 100 households, 20% having 100 to 150 households and the remaining 16% having over 150 households. Within each village, 24 households were surveyed (10% to 59%, average sampling error = 13%) representing 24% of all households in all study villages (596 of 2480, sampling error = 3.57%). Households were chosen by random stratified sampling, with stratification based on house construction standard (Hartter, 2009), a proxy for household wealth. Although stratification could have been done against other factors, such as household size, distance from the park or education attained, poorer households are more dependent on natural resources and are least likely to be represented in surveys (Scoones, 1995). The survey was administered in July and August 2009 by four Ugandan field assistants, three men and one woman, in two local languages (Rutooro and Rukiga). Since literacy rates in the region are low, a verbal informed consent was used at the start of the 1.5 hour survey.

Income generating activities may be linked to resource extraction, so respondents were asked if household

members owned a woodlot, made money from charcoal-making, brick making or firewood selling and the quantity of livestock they owned. Other potential incentives for resource extraction, such as wild animals raiding crops or if the household had experienced food insecurity in the prior year, were also recorded. Since there have been fuel efficient stove projects around KNP, we inquired if the household had an energy-saving stove and how much their firewood usage had reduced as a result. Finally, respondents were asked to rate their perceived benefit of taking resources from the park, structured as a five point scale.

A list of associations having resource access agreements was provided by UWA. The Lake Kabalika Fishing Association was located in the far south-western corner of the park, outside the boundary measurement zone of this study, and was therefore excluded from this research. The leaders of all other resource access associations were interviewed to confirm the scope and financial benefit of the agreement. Records of species and quantity of permitted exotic tree extraction from the park were also provided by UWA.

### 3.2.3 Analysis

Data were corrected for sanctioned activities, by removing legally harvested trees, and entry trails for valid resource access agreements, FACE operations, UWA patrols and research. Identified species were classified as exotic or indigenous and to successional stage (Zanne & Chapman, 2005; Naughton-Treves *et al.,* 2007; Lebamba *et al.,* 2009; Omeja *et al.,* 2009). Michaelis Menten Means extrapolation and Mao Tau species accumulation curve (Colwell, 2009) were used to estimate species richness inside the boundary of KNP. Collecting data adjacent to villages may have resulted in more measured disturbance than if the entire boundary had been measured. However with the exception of tea plantations in the north, it is rare to find locations with no human habitation within a few hundred metres of the park boundary.

Since other factors outside the scope of this paper, such as livelihood assets, income opportunities and access to resources outside the park, could also affect measured levels of extraction, we limited the analysis to correlations aimed at identifying possible trends that can be tested in a future multi-variate analysis as other co-factors become available. All village-scale variables were normally distributed and suitable for parametric analysis (Pearson correlations), with the exception of poaching signs, number of livestock grazing and illegal exotic tree species extraction where non-parametric analysis was employed (Spearman correlations). Household-scale variables were not normally distributed, so non-parametric analysis was used (Kendall Tau-b correlations). With the exception of inherently spatial correlations (straight line distance) and correlations between ordinal variables, correlation residuals were tested for spatial autocorrelation using Moran's I ($p < 0.05$), and, if significant, a simultaneous autoregressive correction (Anselin, 1988) was applied.

To minimize the potential for local villagers to be punished for illegal extraction as a result of this research (Robbins *et al.,* 2006), we used kriging interpolation to rasterize data to a 1000 m grid and then re-sampled the result to a resolution of 30 m within the boundary measurement zone (Fig. 3-1). This permitted the visualization of high and low extraction areas, without specifically identifying which village had participated.

Measured illegal extraction could not be attributed to any one household or any survey household, however since the independent unit of illegal extraction measurement is the village, we assumed that survey data are representative of village conditions and could be averaged to the village scale for comparison with observed extraction. For statistical purposes, disturbance data from each boundary transect and entry trails from that transect were summed, normalized by the length of boundary sampled, and treated as an independent data point to be compared with survey data from the adjacent village. Measured extraction of trees and illegal entry trails were tested for spatial clustering using Moran's I and if clustered, data were checked for high and low value clustering using Gedis-Ord General G (Haining, 2003). Signs of grazing and poaching were not normally distributed so could not be tested for spatial clustering using spatial statistics, as small sparse samples are sensitive to non-normality (Griffith & Layne, 1999).

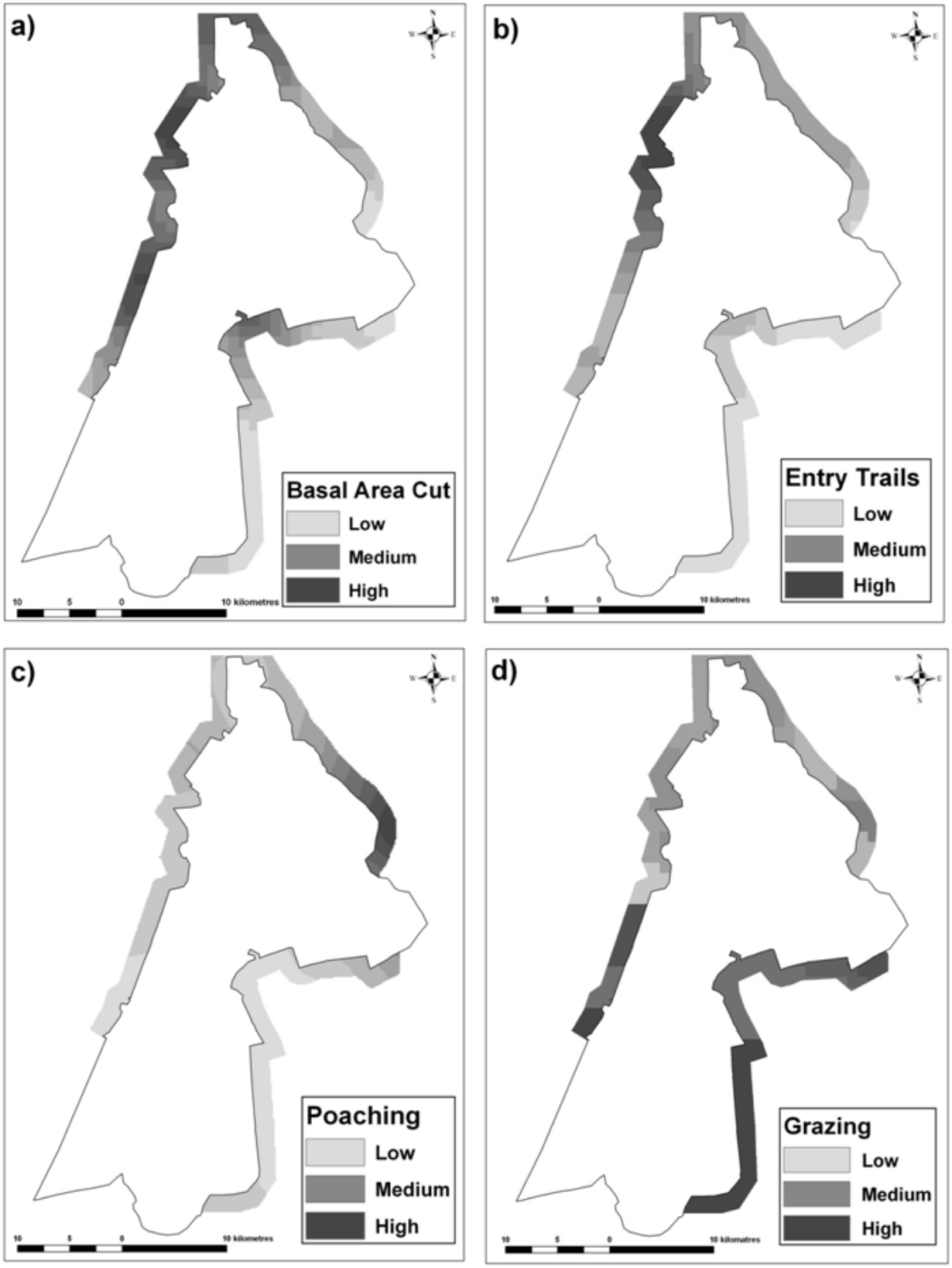



**Figure 3-2: Patterns of illegal extraction along the boundary of Kibale National Park, Uganda**

*(a)* illegal tree/shrub basal area harvested, (*b*) illegal entry trails, (*c*) poaching signs and (*d*) in-park livestock grazing.

## 3.3 Results

### 3.3.1 Patterns of illegal extraction

Illegal extraction was recorded along 19.5 km or 15.2% of the boundary measurement zone (Fig. 3-1), from May to August 2008 and in June 2009 (Table 3-1, Fig. 3-2).

**Table 3-1: Human disturbances found along the boundary of Kibale National Park, Uganda**

Number recorded: [1]per village boundary, [2]per tree, [3]per trail and [4]per village where the activity occurred.

| *Human disturbance* | *Mean (± SD)* | *Percentage of villages where disturbance was found (n = 25)* | *Total recorded for 19.5 km of boundary* | *Per km of boundary measured* |
|---|---|---|---|---|
| Number of harvested trees | 112 (± 99)[1] | 100% | 2794 | 143.6 |
| Basal area harvested ($m^2$) | 0.15 (± 4.6)[2] | – | 46 | 2.4 |
| Number of boundary trees | 26.8 (± 49.8)[1] | 64% | 726 | 37.3 |
| Number of illegal trails | 2.8 (± 3.7)[1] | 60% | 73 | 3.8 |
| Length of trails (m) | 63.5 (± 56.7)[3] | – | 4637 | 238.3 |
| Number of poaching signs | 0.89 (± 2.8)[1] | 24% | 24 | 1.2 |
| Number of grazing livestock | 13.8 (± 41.9)[1] | 36% | 373 | 19.0 |
| Charcoal making | 1.0 (± 0)[4] | 20% | 5 | 0.3 |
| Boundary encroachment | 1.0 (± 0)[4] | 4% | 1 | 0.1 |
| Fire inside park | 1.0 (± 0)[4] | 16% | 4 | 0.2 |

**Trees:** A total of 3035 trees and shrubs were harvested along the boundary zone, 87% of which were identified to species (Table 3-2); 51% of trees were indigenous, while 36% were exotic. The estimated species richness was 66 indicating harvested species represented 88% of the species existing along the boundary. Indigenous species were 25% old growth, 9% mid-successional and 67% early successional. Ninety per cent of the harvest measured < 20 cm in diameter (Table 3-3), suggesting these trees were taken for firewood and construction poles, while the remaining 10%, > 20 cm in diameter, would likely have been taken for timber or charcoal-making.

Of 3035 trees and shrubs harvested, 2794 were illegal. On average, 144 trees or shrubs were illegally harvested per km of boundary. Although tree extraction was found adjacent to all 25 villages, the spatial distribution (Fig. 3-2*a*) was not random (Moran's I=0.11, $Z_{score}$=3.25), with high levels of extraction clustered along the western boundary (Getis-Ord General G=8.61, $Z_{score}$=2.74), possibly owing to the proximity of Fort Portal (Fig. 3-1), a major urban centre (Ahrends *et al.* 2010). Fifty-one per cent of extracted trees were found along entry trails with most recorded within 200 m of the boundary (also see Olupot *et al.,* 2009). Twenty-four per cent of the harvested trees were boundary trees planted by UWA. Across the 25 villages, illegal extraction of all trees was greater where boundary tree harvesting was high ($r_{Pearson}$=0.738, p<0.001, n=25).

Boundary data collected in 2008 indicated illegal tree extraction was increasing over time. Two boundary segments observed in 2008, covering 1.4 km of boundary and seven entry trails were revisited in 2009 (Fig. 3-3). Between visits, 120 trees had been extracted, 15% less than the off-take measured in 2008, with age of cut estimated at less than one year. Tree extraction recorded in 2008 and estimated to have occurred 0–12 months earlier compared well with tree extraction recorded in 2009 and estimated to have occurred 12–24 months prior, provided only trees > 9 cm diameter were included. This indicates smaller tree stumps were decaying over the year between measurements, a trend confirmed by the 2008 dataset of 3035 trees or shrubs, of which 35%, cut more than one year prior, were < 9 cm in diameter, while 69% of the trees cut within one year of measurement were < 9 cm in diameter. Thus tree extraction does not appear to be increasing over time.

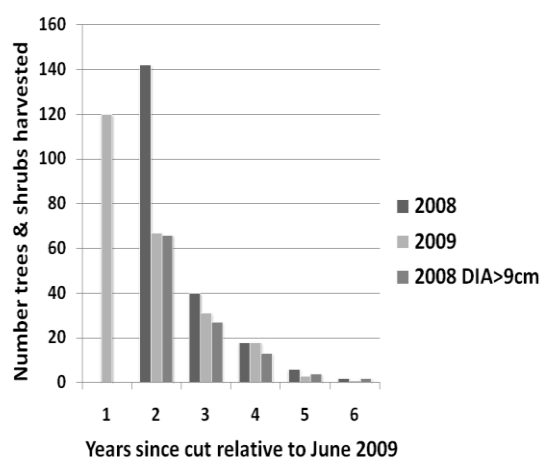



**Figure 3-3: Temporal distribution of tree and shrub harvest**

Boundary segments measured in both 2008 and 2009, covering 1.4 km of boundary and 7 illegal trails.

## Table 3-2: Legally and illegally harvested tree species recorded on 19.5km of boundary sampling done in Kibale National Park, Uganda

ES = early successional, MS = mid-successional, OG = old growth, EX = exotic.

Only species with harvested basal area >0.1m$^2$ have been included.

| *Species* | *Number harvested* | *Diameter (cm) (mean ± SD)* | *Total basal area harvested (m$^2$)* | *Time since cut (months) (mean ± SD)* | *Form* | *Stage* |
|---|---|---|---|---|---|---|
| *Eucalyptus* spp. | 845 | 12.87 ± 8.88 | 16.219 | 18.07 ± 22.12 | Tree | EX |
| *Albizia grandibracteata* | 120 | 9.94 ± 12.9 | 2.487 | 12.17 ± 18.43 | Tree | ES |
| *Parinari excelsa* | 6 | 55.23 ± 36.4 | 1.958 | 20.63 ± 26.2 | Tree | OG |
| *Olea welwitschii* | 6 | 53.69 ± 36.83 | 1.891 | 38.13 ± 39.78 | Tree | MS |
| *Ficus* spp. | 49 | 14.20 ± 15.01 | 1.625 | 10.38 ± 16.0 | Tree | ES |
| *Prunus* spp. | 68 | 11.31 ± 11.78 | 1.413 | 10.64 ± 10.95 | Tree | ES |
| *Funtumia* spp. | 84 | 11.15 ± 9.18 | 1.369 | 7.45 ± 8.63 | Tree | ES |
| *Macaranga schweinfurthii* | 57 | 10.8 ± 12.64 | 1.224 | 6.57 ± 6.32 | Tree | ES |
| *Acrocar Pearsonus* sp. | 19 | 21.31 ± 18.41 | 1.157 | 17.16 ± 17.0 | Tree | EX |
| *Markhamia* spp. | 135 | 7.9 ± 5.64 | 0.997 | 10.62 ± 9.12 | Tree | OG |
| *Celtis* spp. | 74 | 9.69 ± 7.07 | 0.832 | 15.11 ± 16.71 | Tree | ES |
| *Strombosia scheffleri* | 24 | 10.04 ± 18.14 | 0.785 | 13.08 ± 11.72 | Tree | OG |
| *Maesa lanceolata* | 112 | 8.61 ± 3.43 | 0.754 | 12.72 ± 11.48 | Tree | ES |
| *Sapium* spp. | 54 | 10.29 ± 8.19 | 0.728 | 7.86 ± 7.63 | Tree | ES |
| *Spathodea campanulata* | 77 | 9.26 ± 5.83 | 0.722 | 10.42 ± 6.14 | Tree | ES |
| *Millettia dura* | 70 | 10.19 ± 5.28 | 0.721 | 11.33 ± 16.05 | Tree | ES |
| *Bridelia micrantha* | 81 | 9.02 ± 5.3 | 0.694 | 8.94 ± 12.90 | Tree | ES |
| *Senna spectabilis* | 128 | 7.42 ± 3.52 | 0.677 | 4.29 ± 3.83 | Tree | EX |
| *Neoboutonia* sp. | 33 | 13.54 ± 5.95 | 0.564 | 7.11 ± 11.29 | Tree | ES |
| *Pseudospondias microcarapa* | 6 | 28.33 ± 18.75 | 0.516 | 10.67 ± 9.85 | Tree | OG |
| *Dombeya mukole* | 47 | 9.10 ± 5.93 | 0.432 | 11.34 ± 10.5 | Tree | OG |
| *Polyscias fulva* | 4 | 27.61 ± 24.74 | 0.384 | 34.25 ± 57.2 | Tree | ES |
| *Dracaena steudneri* | 36 | 10.46 ± 4.33 | 0.361 | 2.31 ± 2.39 | Tree | ES |
| *Newtonia buchananii* | 31 | 9.86 ± 7.26 | 0.361 | 8.6 ± 7.7 | Tree | OG |
| *Blighia* spp. | 33 | 9.16 ± 6.95 | 0.339 | 12.5 ± 10.48 | Tree | ES |
| *Uvariopsis congensis* | 103 | 5.52 ± 3.26 | 0.331 | 7.61 ± 6.27 | Tree | OG |
| *Trema* spp. | 3 | 27.91 ± 27.59 | 0.303 | 24.67 ± 26.63 | Tree | ES |
| *Erythrina abyssinica* | 26 | 10.75 ± 5.1 | 0.287 | 10.54 ± 8.9 | Tree | ES |
| *Fagara angolensis* | 11 | 12.73 ± 13.67 | 0.287 | 13.32 ± 11.98 | Tree | ES |
| *Teclea nobilis* | 68 | 6.79 ± 2.76 | 0.287 | 9.9 ± 14.42 | Tree | MS |
| *Jacaranda* sp. | 34 | 7.85 ± 2.67 | 0.183 | 8.59 ± 6.03 | Tree | EX |
| *Aphania* sp. | 23 | 8.94 ± 4.6 | 0.181 | 5.09 ± 5.15 | Tree | OG |
| *Tabernaemontana* spp. | 29 | 7.05 ± 4.24 | 0.153 | 9.5 ± 9.96 | Tree | MS |
| *Pancovia turbinata* | 1 | 43.93 ± 0 | 0.152 | 60 ± 0 | Tree | MS |
| *Diospyros abyssinica* | 13 | 10.21 ± 5.28 | 0.133 | 19.46 ± 16.25 | Tree | MS |
| *Psidium* sp. | 57 | 4.73 ± 1.86 | 0.115 | 2.24 ± 3.14 | Tree | EX |

## Table 3-3: Diameter and stage classification of extracted woody species along the boundary of Kibale National Park, Uganda

Stages as defined in Table 2.2

| *Diameter at cut* | *Species type* | | | | | *Total* | | *Productive practice* |
|---|---|---|---|---|---|---|---|---|
| | *Old growth* | *Mid-successional* | *Early successional* | *Exotic* | *Unknown species* | *n* | *%* | |
| < 10 cm | 306 | 100 | 699 | 635 | 298 | 2038 | 67 | Firewood and poles |
| 10–19.9cm | 56 | 28 | 254 | 293 | 70 | 701 | 23 | Poles |
| 20–29.9cm | 5 | 0 | 40 | 125 | 14 | 184 | 6 | Timber |
| 30–39.9cm | 6 | 0 | 16 | 40 | 4 | 66 | 2 | Timber and charcoal |
| 40–49.9cm | 5 | 1 | 7 | 5 | 2 | 20 | 1 | Timber and charcoal |
| 50cm and over | 4 | 3 | 13 | 4 | 2 | 26 | 1 | Timber and charcoal |
| Total (*n*) | 382 | 132 | 1029 | 1102 | 390 | 3035 | | |
| Total (%) | 13 | 4 | 34 | 36 | 13 | | | |

**Trails:** Ninety-nine human entry trails were found adjacent to 22 study villages, while three villages had no entry trails. Sixteen trails were legally sanctioned, although we still found evidence of illegal activities on these trails, namely tree harvesting, livestock grazing and two pit-fall animal traps. After removing legal trails, 73 remained (3.75 illegal entry trails per km of boundary observed). Trail length varied from 9 m to 362 m, with an average length of 63.5 m (median=47 m). The distribution of illegal trails was not random (Moran's I=0.12, $Z_{score}$=3.29), with the density of trails highest along the north-western boundary (Getis-Ord General G=9.9, $Z_{score}$=2.75), where up to 20 trails were observed along 1 km of boundary (Fig. 3-2*b*). The density of illegal entry trails was correlated with number of trees extracted ($r_{Pearson}$=0.682, p<0.001, n=25) because we often found tree harvesting at the end of entry trails.

**Animal poaching:** Twenty-four signs of poaching were found near six villages: 17 pit-fall traps, two snares and five snare holding depressions. The mean number of poaching signs per kilometre of boundary was 0.012, but 62.5% were found along the north-eastern boundary (Fig. 3-2*c*). One survey respondent commented that people in his village always took meat from the park, but they travelled far from the village to hunt, which would suggest poaching signs found during boundary observations under-represent actual levels of poaching for a given village.

**Livestock grazing:** Livestock grazing was observed in the park near nine study villages: six goat herds (< 20 animals), and three large herds of cows (100–200 animals). On average, 19 livestock were seen grazing inside the park per kilometre of boundary; however, 86% were observed in the southern half of the Park (Fig. 3-2*d*). Since boundary observation adjacent to a village occurred over 1-4 days, there was a concern livestock grazing in the park might be under-observed, however, the number of observed livestock was predicted by the number of households reporting fines for grazing in the park ($R^2$ = 0.566, p<0.001, n=25). Therefore, we believe hotspots for grazing were identified by this study.

**Other disturbances:** Charcoal-making was observed inside or adjacent to the park in five villages. Encroachment of the boundary to expand agricultural land was found in only one village, but there was disagreement about the location of the boundary and no visible demarcation. Finally, evidence of fire inside the park was found adjacent to four villages. Two incidents were small (<0.005 $km^2$) and not intentionally set: one from burning grass while maintaining a crop raiding protection trench and the other the result of a beehive catching fire. Two incidents were large (>10 $km^2$), but discussion with UWA indicated these were started further away and therefore not attributable to the study villages.

### 3.3.2 Demand for park resources

Non-response rates increased as questions about resource extraction became more direct. All respondents answered which resources they wanted access to, while three (0.5%) declined to answer questions about the perceived benefit of resource extraction and 42 (7%) declined to answer questions about admitted extraction of park resources. The percentage of respondents admitting extraction was lower than the percentage desiring access to park resources (Table 3-4). However, the percentage of respondents admitting extraction of a specific resource was aligned with the percentage wishing to harvest that resource ($r_{Pearson}$=0.750, p=0.005, n=12 resources).

**Trees:** The two most desirable resources were firewood and construction poles, consistent with 90% of measured tree extraction being of this size. Eighty-nine per cent of respondents wanted firewood, and this was evenly distributed around the park, except along the north-eastern boundary, where only 77% of respondents wanted access to firewood and where a villager told us "we desire nothing from the park, because we have all the resources we need on our own land" (Household Survey, 28/7/09). Demand for firewood was correlated with admitted extraction, and marginally with measured extraction of trees < 10 cm in diameter (Table 3-4). The percentage of households wanting exotic trees was positively correlated with measured legal plus illegal off-take of exotic trees (Table 3-4).

Extraction of firewood from KNP may be perceived as socially acceptable, especially since the highest admitted extraction was in two villages located adjacent to UWA outposts and, in both villages, we were told rangers occasionally allowed firewood collection within the park and that in other villages, rangers allowed access for special occasions. One UWA ranger said, "I cannot allow people to eat raw food when there is dead wood in the forest" (25/6/09).

Charcoal-making was reported in 10 villages, but only three coincided with villages where we found charcoal-

making along the boundary. Brick-making was reported in 11 villages, while brick-making kilns were observed in 22 villages. The percentage of survey respondents making charcoal or bricks was not correlated with overall or large (> 20 cm diameter) tree extraction; however the percentage of households selling firewood did correlate with admitted firewood extraction ($r_{Pearson}$=0.641, p=0.001, n=25) and with measured extraction of trees between 10 and 20 cm in diameter ($r_{Pearson}$=0.554, p=0.004, n=25).

Villages where more respondents wanted access to firewood also had more woodlots ($r_{Pearson}$=0.401, p=0.047, n=25), which in turn was correlated with tree extraction from the park ($r_{Pearson}$=0.503, p=0.010, n=25), perhaps indicating people are planting trees in villages with higher demand for fuel wood, but that woodlots are not mitigating cost-free extraction from the park. Use of energy-saving stoves was also higher in villages where more woodlots had been planted ($R^2$ =0.311). Energy saving stoves were owned by 85 respondents living in 14 study villages, with 65% of users reporting they had halved their firewood consumption, while 25% said they used only one quarter of the wood they used before getting an energy saving stove.

**Non-timber forest products (NTFP):** The next most wanted park resources were medicinal plants (82%), grasses (66%), access to put beehives in the park (62%) and water (60%). Respondents also wanted other NTFPs, such as wild coffee, creeping plants, palm leaves, handcraft materials, mushrooms, grasshoppers, sand, stones and clay. Ninety per cent of survey households wanted access to at least one NTFP, and 32.5% admitted entering the park to collect them. Village averaged demand for and admitted extraction of NTFPs were correlated (Table 3-4).

Park access for medicinal plants appeared to be allowed. A villager told us "we can request to pick medicine and it is usually accepted" (Focus Group, 10/7/08), but for all other resources "if you enter the park you are arrested" (Focus Group, 28/7/08). Restricting access to NTFPs was perceived by some as removing their traditions, with one villager stating, "Traditionally, Bakiga make papyrus plates, but now we have to use modern plates because we lack access to materials" (Focus Group, 23/7/08).

**Animal poaching and fishing:** Thirty-one per cent of households wanted access to bushmeat, and 5%, located in 17 villages, admitted entering the park to hunt. A study in two southern villages estimated that almost 40% of households were engaging in illegal hunting, even though less than 2% admitted doing so (Solomon *et al.,* 2007). Households in this study wanted bushmeat, a resident commenting that "poachers sell bushmeat to people and it is very delicious" (Household Survey, 26/6/09). Along the north-eastern boundary we were told by a village chairperson that "people have lost a lot to [crop raiding] elephants and baboons, so locals go to the park to get meat" (8/6/09).

Seven per cent of households reporting a period of food insecurity in the prior year admitted taking meat from the park. Seven households reported setting animal traps to protect their crops from wild animals, only two of which admitted taking meat from the park. One government official did comment that, "there may still be poor behaviours such as hunting that are culturally based" (District Chairperson, 12/6/08), but we did not find any significant linkage between poaching and tribal affiliation.

Forty-three per cent of respondents wanted access to fish from park lakes and 7% admitted fishing inside the park. Admitted fishing was correlated with demand for fish (Table 3-4) and with straight line distance from the village to the nearest lake within the park ($r_{Pearson}$=–0.440, p=0.028, n=25).

**Livestock grazing:** Fifty-six per cent of respondents wanted to graze livestock in KNP, but only 8% admitted doing so. Villagers on the southern boundary of the park admitted that during the dry season they grazed their cattle and went for water near the park, and requested they be allowed access in times of drought. However the most frequent explanation for livestock grazing in the park was that animals did not understand boundaries.

Around KNP, illegal grazing within the park increased with cattle ownership. Cattle ownership was higher along the south-eastern park boundary and the number of livestock observed in the park was predicted by village-averaged cow ownership ($R^2$=0.721, p<0.001, n=25). However, some respondents stated that they "did not want to graze livestock in the park because there is a risk of domestic animals contracting wild animal diseases" (Household Survey, 16/7/09), or "the animals might be predated" (Household Survey, 16/7/09).

**Perceived benefit of extraction:** After removing 59 households that reported having resource access agreements, 76% of respondents said they did not benefit from extraction, 9.0% reported 'a little benefit', 7.7% 'some benefit', 4.0% 'a considerable benefit' and 3.1%

**Table 3-4: Resources wanted from Kibale National park, Uganda and admission of illegal extraction**

n/d = no data collected.

[1]Residuals of linear model ($R^2$ = 0.234) were spatially auto-correlated (Moran's I = 0.2613, $p < 0.001$), after spatial lag correction $R^2$ = 0.464. [2]'Other' includes creeping plants, palm leaves, handcraft materials, mushrooms, grasshoppers, sand, stones, clay, access to hot springs and access to land for cultivation.

| | | *Resources* | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Woody Biomass* | | | *Non-timber forest products (NTFPs)* | | | | | | *Protein sources* | |
| | | *Firewood* | *Construction poles* | *Exotic trees* | *Medicinal plants* | *Grasses* | *Beehives In park* | *Water* | *Wild coffee* | *Other*[2] | *Fishing* | *Bushmeat* | *Livestock grazing* |
| % respondents wanting access (n=596) | | 89 | 83 | 59 | 82 | 66 | 62 | 60 | 47 | 11 | 43 | 31 | 56 |
| % respondents admitting access (n=596) | | 38 | 28 | n/d | 28 | 17 | n/d | 13 | n/d | 4 | 7 | 5 | 8 |
| Wanted versus admitted access (village mean, n=25) | $r_{Pearson}$ | 0.553 | 0.356 | n/d | | | 0.453 | | | | 0.493[1] | 0.340 | 0.201 |
| | $p$ | 0.004 | 0.080 | | | | 0.023 | | | | 0.012 | 0.097 | 0.335 |
| Perceived benefit of extraction verses admitted extraction frequency (n=596) | $r_{Kendall\text{-}Tau}$ | 0.444 | 0.363 | n/d | | | 0.374 | | | | 0.115 | 0.140 | 0.117 |
| | $p$ | <0.001 | <0.001 | | | | <0.001 | | | | 0.006 | 0.001 | 0.005 |
| Wanted access versus observed extraction (village mean, n=25) | $r_{Spearman}$ | 0.403 | -0.153 | 0.414 | | | n/d | | | | n/d | -0.090 | 0.320 |
| | $p$ | 0.046 | 0.466 | 0.039 | | | | | | | | 0.669 | 0.119 |
| Observed extraction | | Trees <10cm diameter | Trees <20cm diameter | Exotic trees (legal & illegal) | | Extraction of NTFPs not measured | | | | | n/d | Poaching signs | Livestock grazing in Park |

benefited 'a lot'. When compared with household admitted resource extraction frequency, the results were all significant (Table 3-4), with the greatest contribution to perceived benefit coming from firewood, NTFP and construction pole extraction. Perceived benefit of extraction increased the closer a household was to the park ($r_{Spearman}$=-0.210, p<0.001, n=536), as did the admitted frequency of extraction of firewood ($r_{Spearman}$=-0.319, p<0.001, n=516), poles ($r_{Spearman}$=-0.223, p<0.001, n=516) and NTFPs ($r_{Spearman}$=-0.211, p<0.001, n=516), indicating extraction may be opportunistic for those who live close to the park. Admitted frequency of grazing and poaching within the park was not correlated with household distance from the park.

### 3.3.3 Resource access agreements

Eight study villages had residents who participated in active RAAs: beekeeping associations (five villages), drama and craft associations (two villages) and a fishing agreement (one village). To determine if RAAs were improving conservation behaviours, basal area of harvested trees, number of poaching signs and number of livestock grazing in the park were compared for villages with (n=8) and without (n=17) RAAs. No significant difference was found when all types of RAAs were considered (t-test, p=0.688), however, if only beekeeping RAA villages (n=5) were compared with all other villages (n=20), the level of illegal tree harvesting was significantly lower near villages with beekeeping associations (t-test, p=0.006). The protection of the park by beekeepers was supported by a village chairperson telling us that "beekeepers guard the park and stop others entering in case UWA thinks it is them that do any damage" (21/7/08). The beekeepers consider protecting the park to be akin to protecting an investment because as one stated; "hives situated inside the park give better honey yields" (RAA Chairperson, 10/3/10). Members of these beekeeping associations make up to US$250 per year, more than is made from crafts or fishing, suggesting that RAAs can be effective, if the benefit to association members is sizable.

Some MOUs were with individuals who paid for access to harvest a specified exotic tree species. Other MOUs were with villages and did not require payment. Eight study villages had either individuals or communities with agreements to extract exotic trees. The number of exotic trees legally extracted by MOU correlated with the demand to access exotic timber ($r_{Spearman}$=0.478, p=0.016, n=25), so many who want exotic trees are negotiating access. However, villages with MOUs also had higher illegal tree extraction (t-test, p=0.035) and a significantly higher number of the illegal harvest were unauthorized exotic tree species (Mann-Whitney U, p=0.011). Most MOUs in the north were near tree plantations, planted when the park was a forest reserve (Fig. 3-1). In the south, where FACE operations exist (Fig. 3-1), workers are permitted to take exotics home, but not everyone has access to the harvest. One villager said, "It is unfair that only FACE workers get to take the trees, if they can have the exotics then so should I" (20/6/08). Illegal exotic tree extraction was higher when a village was closer to an old plantation or FACE operation ($r_{Spearman}$=-0.546, p=0.005, n=25). Access agreements, plantation logging history and the perception of unfair distribution of resources may allow people to perceive it is acceptable to harvest exotic trees.

## 3.4 Discussion

Although human disturbance studies inside forested national parks are rare (Robbins *et al.,* 2006), in East Africa, extraction of woody biomass is consistently identified as the most widespread disturbance (Bleher *et al.,* 2006; Baranga, 2007; Olupot *et al.,* 2009). The most pervasive need for wood is for fuel (Dovie *et al.,* 2004; Naughton-Treves *et al.,* 2007), followed by poles for construction and hardwoods to make charcoal (Baranga, 2007). This was also the case in KNP, with most extraction being trees or shrubs used for fuel wood and poles. Planting boundary trees did not reduce illegal extraction and boundary trees became another source for harvesting woody biomass.

The 'parks' conservation narrative, as typically described in the literature, would prescribe increased patrolling and fines for wood extraction (Terborgh & van Schaik, 2002), however increased enforcement to limit extraction of such a basic domestic need is seen by many as environmentally unjust (Abbot & Mace, 1999). The 'people' conservation narrative would argue that conservation cannot deny basic needs and would demand more access, to ensure communities can meet their subsistence requirements (Adams & Hulme, 2001). Woody biomass in KNP can accumulate rapidly after disturbance (Chapman & Chapman, 1999), and extraction of early successional species may be beneficial for forest biodiversity (West *et al.,* 2000). So the illegal tree extraction recorded in KNP may be sustainable, although

a long term study would be required to confirm the regeneration rate after harvest and to understand if off-take increases as resources outside the Park diminish (Chhetri *et al.,* 2003). However, current levels of illegal extraction exist with the threat of fines and arrests if perpetrators are caught. Given land outside the KNP is largely denuded of trees with the exception of forest fragments and individually owned woodlots (Hartter & Ryan, 2010), that extraction of fuel wood from these forest fragments is perceived as a right by most, even if the fragment belongs to an individual (Hartter & Ryan, 2010), and that legal access to woody biomass inside the park appeared to increase illegal tree extraction, increasing community access to trees inside KNP is not a solution that will maintain habitat for biodiversity conservation.

Local people however need wood for fuel and construction, so it would be beneficial to develop strategies to support this need. Supplying tree seedlings, setting up woodlots and energy-saving stove programmes have been tried with success in some African countries (Stocking & Perkin, 1992; Barnes *et al.,* 1994), and have been introduced in limited areas around KNP. The reported benefit of energy-saving stoves highlights this option as one that should reduce firewood consumption, although stove efficiency can vary and actual firewood reduction may not be as significant as perceived by our respondents (Wallmo & Jacobson, 1998). If funding can be secured, UWA would like to provide tree seedlings to communities (UWA warden, personal communication 2009). Increased frequency of admitted tree extraction for households close to KNP indicates tree seedling programmes should be spatially targeted towards the boundary. However, these programmes may not be adopted unless people perceive fuel wood is in short supply (Barnes *et al.,* 1994), therefore increased policing along park boundaries may be required in parallel with development options (Abbot & Mace, 1999), and illegal off-take should be monitored to determine if these initiatives are reducing illegal extraction while meeting the needs of communities.

Animal poaching by people living close to strictly protected PAs in East Africa is usually attributed to food insecurity, for protection against crop raiding or because the activity is traditional (Naughton-Treves, 1998; Neumann, 1998; Chapman *et al.,* 2006). This 'people' conservation narrative, excusing hunting inside PAs as a survival response, was not supported by our research, as most households were not food insecure and were not trapping animals to defend against crop raiding. Therefore the policy of exclusion for animal poaching is warranted and is not counter to environmental justice. However, households losing crops to park animals are disproportionally bearing the costs of conservation and compensation is warranted. However, this compensation is currently not affordable by local authorities. In recent years, much of the shared gate revenues have been targeted towards deep trenches to protect farmers from crop raiding and where these trenches exist, we observed lower levels of human disturbance inside the KNP, indicating this practice should be continued.

Collaborative resource management attempts to find a middle ground between the parks and people narratives, allowing local communities to sustainably access specific resources from the park, while helping to police the park for poachers (Adams & Hulme, 2001; Chhetri *et al.,* 2003). Although these agreements do improve relations between UWA and the associations that enter into them, as evidenced by RAA members reporting illegal activity to UWA (Chhetri *et al.,* 2003), lower levels of illegal tree extraction were only found near communities with lucrative beekeeping associations, suggesting this conservation strategy should be applied to resources that create income generating opportunities rather than for extraction of subsistence resources.

## 3.5 Conclusions

Human disturbance indicators were found to cluster and be spatially coherent, identifying hotspots for extraction of particular resources. The most wanted and extracted resources were fuel wood and construction poles. Although the concentration of extraction on exotic and early successional tree species indicates current levels of extraction may be sustainable, increased access for local communities is not recommended because illegal wood extraction was higher adjacent to villages with agreements to legally extract wood. Instead development programmes to increase wood supplies outside the park and to reduce the demand for fuel wood are encouraged.

Admitted poaching did not align with households experiencing food insecurity or using traps to protect against crop raiding animals. We did find livestock grazing in the park to be higher where households owned more cows, a sign of wealth. Since lack of access to bushmeat and grasslands for cattle does not appear to hinder the

livelihoods of the communities, our data support the policy of exclusion for animal hunting and in-park grazing.

Finally, illegal woody biomass extraction was low adjacent to villages where residents were members of beekeeping associations with legal access to the park for beehive placement. Therefore, the implementation of resource access agreements with local community associations was found to be an effective means of reducing illegal tree extraction if association members profited substantially.

# 4 Demand and Proximity: Drivers of Illegal Forest Resource Extraction

In this chapter I look for socio-economic predictors of illegal resource off-take from Kibale National Park. Using linear regression, the measured off-take from Chapter 3 is modelled using socio-economic variables, including population density, adult education, household wealth, the number of household income generating activities, and access to the local urban market in Fort Portal. This chapter, co-authored with Dr. Joel Hartter, has been accepted for publication by *Oryx* following the final submission of this dissertation.

**Summary:** Illegal extraction from protected areas is often shaped by the socio-economic landscape within which the protected area is located. We coupled village scale socio-economic parameters collected using a household survey, with measured levels of illegal resource extraction, proximate to study villages, to investigate the socio-economic drivers of illegal extraction from Kibale National Park in Uganda. The level of illegal tree harvesting and the number of illegal entry trails into the park were driven by subsistence demand from villages adjacent to the park and by for-profit extraction to supply the local urban market in Fort Portal, while in-park grazing was linked to high livestock ownership. Capital asset wealth, excluding livestock, was found to mitigate illegal resource extraction from the park. We also found high human population density to be spatially coincident with tourism, research and carbon sequestration operations, all providing park-based employment opportunities. We stress the need for conservation strategies to help meet the resource needs of local communities, and to manage urban fuelwood demand, in order to reduce illegal extraction from protected areas.

## 4.1 Introduction

It is estimated that each year the world looses 13 million hectares of forest and up to 40,000 species that depend on those forests for survival (Kremen *et al*., 2000). Many threatened forests are located in regions of chronic rural poverty (Sunderlin *et al*., 2005). Households often need the natural resources available in protected areas (PAs), such as wood for cooking, heating, and house construction (Naughton-Treves *et al*., 2007), grasslands for cattle (Infield et al., 1993; Neumann, 1998; Kideghesho *et al*., 2007), and wild animals for meat (Neumann, 1998; Chapman *et al*., 2006). As a result, subsistence-based livelihoods near PAs are often perceived as a threat to conservation (Mbile *et al*., 2005).

Land conversion for agriculture and demand for wood has driven deforestation in many African countries (Tole, 1998; Dovie *et al*., 2004); with increasing human population density further accelerating deforestation rates (Cropper & Griffiths, 1994; Tole, 1998). Remaining forest areas are also under pressure from urban centres, as the demand for charcoal and fuelwood extend beyond urban boundaries (McDonald *et al*., 2009; Ahrends *et al*., 2010). Conservationists cite increased demand for natural resources, spurred by growing human populations, as the greatest threat to PAs (Brook & Bradshaw, 2006); specifically threatening endangered species in African tropical mountain forests (Rondinini *et al*., 2006; Burgess *et al*., 2007). The pressure may be further magnified by migration of people to the boundaries of PAs, either in search of arable land (Mwamfupe, 1998) or employment (Newmark & Hough, 2000; Wittemyer *et al*., 2008).

Dependency on forest resources has been linked to household wealth and education, suggesting that illegal extraction from PAs could be mitigated by development activities in local communities (Adams *et al*., 2004). Increased household wealth has been reported to result in more positive attitudes towards PAs (Infield, 1988; Gillingham & Lee, 1999), however wealthier households have also expressed the desire to extract PA resources for profit (Holmes, 2003; Fisher & Shively, 2005). Lack of education was found to result in negative attitudes toward conservation and a desire to de-gazette parks in South Africa (Infield, 1988), while in Tanzania, the perceived benefit of wildlife increased with higher levels of education attained (Gillingham & Lee, 1999).

In Africa, attitudes towards PAs are often shaped by the socio-economic landscape (Masozera & Alavalapati, 2004; Kideghesho *et al*., 2007). However, less clear is whether attitudes translate into behaviours (Holmes, 2003). A pragmatic measure of support for conservation is whether or not communities illegally extract resources from a PA (Bruner *et al*., 2001), either to support

subsistence livelihoods (Masozera & Alavalapati, 2004), or to profit from selling bushmeat (Chapman *et al*., 2006), timber (Holmes, 2003) or charcoal (Naughton-Treves *et al*., 2007). The decision to extract resources from the PA is made by individuals in the household. Unfortunately resource extraction measured inside a PA cannot be attributed to any one individual or household. Therefore, we have aggregated socio-economic survey data from villages adjacent to Kibale National Park, a mid-altitude forest park in western Uganda, for comparison with measured resource extraction within the boundary of the park, near the study villages. Acknowledging that aggregating data to the village scale masks individual motivations to enter the park, we use village-scale socio-economic factors that predict illegal resource extraction as the basis for our recommendations to improve conservation for tropical forest PAs.

## 4.2 Methods

### 4.2.1 Study site

In Uganda, human population density is increasing by 3.2% per annum (Uganda Bureau of Statistics, 2009), with the well-being of that population marginally improving as evidenced by the Human Development Index (+1.87% per annum (UNDP, 2010)). However deforestation has also increased in Uganda with forest cover declining on average 0.9% per year (Tole, 1998), making the protection of forest habitats for endangered species even more critical.

Kibale National Park (KNP) is a 795 km$^2$ mixed forest and savannah PA in western Uganda (Fig. 4-1), located within 15 km of the urban centre of Fort Portal. The core Ugandan conservation policy is exclusion and enforcement of park boundaries by the Uganda Wildlife Authority (UWA), tempered by limited resource access agreements, partial sharing of tourist revenues and education outreach (Uganda Wildlife Statute, 1996). KNP is home to one of the largest populations of chimpanzees in East Africa (Plumptre *et al*., 2003) and 12 other primate species (Chapman & Lambert, 2000). Habituated chimpanzees attract over 7,000 foreign tourists per year (UWA, 2009), for whom tourist facilities have been developed, providing full and part-time employment for

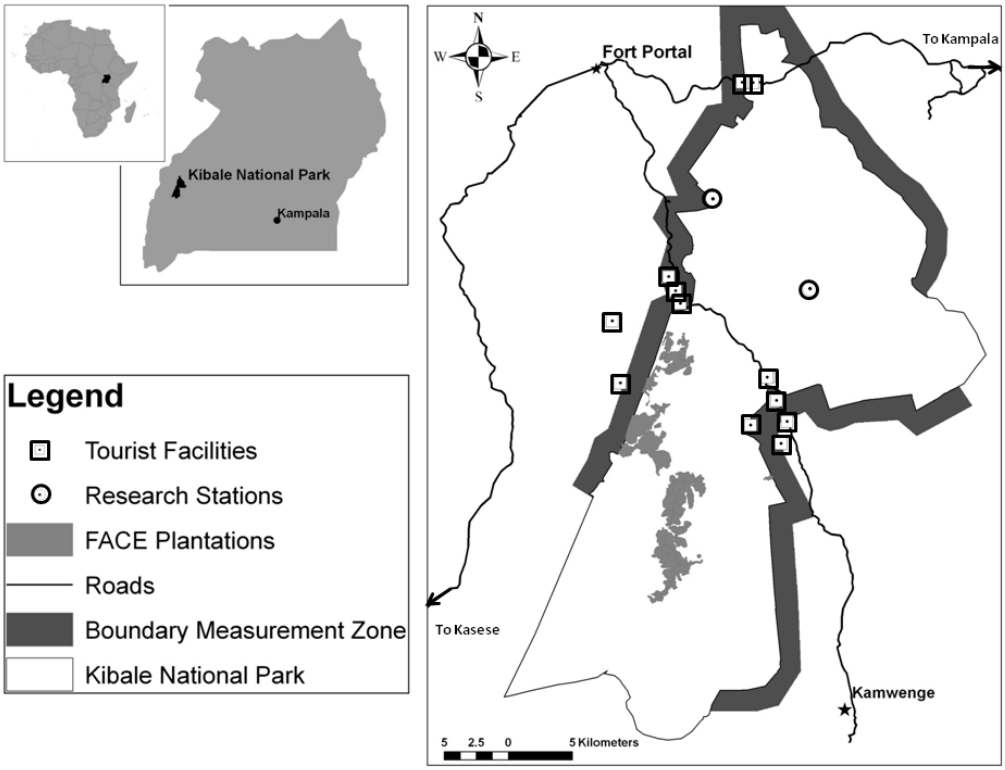



**Figure 4-1: Study site map of Kibale National Park in Uganda**

over 250 people (87% from local villages). The park also attracts researchers to the Makerere Biological Field Station (MUBFS) to study primates (Chapman & Lambert, 2000), aquatic systems (Chapman *et al*., 2004), forest ecology (Struhsaker, 1997), and conservation management (Lepp & Holland, 2006; Omeja *et al*., 2009; Hartter & Goldman, 2011). In January of 2010, MUBFS and active research projects employed 93 people, 91% from local villages. A carbon sequestration partnership has been established between UWA and the FACE the Future Foundation (FACE) employing over 300 people in 2008, to plant indigenous trees.

Most local inhabitants are subsistence farmers, affiliated with either the Batooro tribe in the north or the Bakiga tribe in the south. All cooking and heating, as well as charcoal, alcohol, and brick production is fueled by wood (Naughton-Treves *et al*., 2007). Rural households also depend on KNP for craft materials, medicinal plants and bark, and as places to put beehives for honey production (Hartter, 2010). In a prior survey, 31.7% of the people admitted to entering the park to gather resources (Mugisha, 2002).

People live in villages of about 100 households, governed by a village council, and led by a village chairperson. For this study, a village is defined by the spatial extent of households associated with a village name under the leadership of one village chairperson. Study villages were located approximately every five kilometres along the boundary measurement zone (Fig. 4-1) and were chosen based on some village members holding and/or cultivating land directly adjacent to the park boundary. Twenty-five villages participated in the study from May to August in 2008 and 2009. Households within the study villages were located from 15m to 3,300m from the park boundary. Eleven villages had residents who were employed full or part-time by researchers, tourism facilities, or the FACE Foundation.

### 4.2.2 Data collection

**Socio-economic data:** Data for population density, education, wealth assets, and income generating activities were collected using a household survey. All village households were Global Positioning System (GPS) located while simultaneously noting the number of buildings in the household compound and the construction standard of each primary dwelling. Villages ranged from 41 to 242 households, with 64% of the villages having less than 100 households.

Twenty-four household surveys were conducted in each village (10% to 59% of village households), using a stratified random sample. The sample population was stratified by wealth, categorized by the type of primary dwelling construction, where mud and wattle construction indicated a poorer household, and brick construction a richer household (Ellis & Bahiigwa, 2003; Hartter, 2009). Wealth stratification was chosen because poorer households are more dependent on natural resources and least likely to be represented in a random sample (Takasaki *et al*., 2000). The survey was administered to 596 households, representing 24% of all households, by four Ugandan field assistants, in two local languages (Rukiga and Rutooro).

Data collected included household composition, where the household head had been born, and education attained by each household member. Information about wealth assets, including land holdings, the number and type of livestock, and number of radios, cell phones, bicycles, and motorcycles, owned by the household, were recorded and valued using local market prices from districts around KNP. We also noted the number and type of income-generating activities that the household was engaged in (*e.g*., employment outside of the household, retail sales, growing cash crops and woodlots). Village population density was calculated by multiplying the village-mean number of people per household by the number of households in that village, divided by the village area.

The value of buildings within a household compound, including primary and secondary dwellings and outbuildings, was converted to a Building Wealth Unit (BWU), by dividing the summed value of all buildings by the value of a medium category primary dwelling. Livestock ownership was converted to a Cattle Equivalent Unit (CEU) by summing the market value of all household livestock and dividing by the value of one cow (Bahiigwa *et al*., 2005). Similarly, the summed market value of radios, cell phones, bicycles, and motorcycles owned was divided by the value of one bicycle to create a Bicycle Equivalent Unit (BEU) to represent communication and transportation devices owned by the household. Ownership of land, buildings, livestock and

communication and transportation devices are the primary means of capital asset wealth accumulation in rural Uganda (Ellis & Bahiigwa, 2003). Capital asset wealth was calculated for each household by summing the market value of these assets.

Urban market access for each village was represented by the road distance from the village to the nearest urban centre, Fort Portal (population 40,988; Uganda Bureau of Statistics , 2010). Since a road map was not available, a road network was built from GPS tracks collected as we travelled around KNP. There are three other smaller urban centres within 15 km of KNP: Kyenjojo Town (population 20,100), Kamwenge Town (population 16,100), and Bigodi (population 11,070). However, only road distance to Fort Portal was found to have any significant influence on measured illegal extraction.

**Illegal extraction data:** Resource extraction was measured by walking a 600 to 850m transect of the boundary between the village and the park. We recorded the number of harvested trees, entry trails, livestock seen grazing inside the park and poaching evidence (Olupot & Chapman, 2006). All park entry trails were followed to their terminus, with the exception of UWA patrol and research trails, to record the same information. UWA records of resource access agreements were used to remove legally harvested trees and sanctioned entry trails from the data. Measured extraction data from each village boundary were normalized by the length of the boundary sampled for that village, and treated as an independent data point to be compared with village aggregate survey data from the adjacent village.

In the household survey, respondents were asked if they ever illegally entered the park to collect resources. Given the illegal nature of the activity, this question was prone to non-response and social undesirability biases (St. John *et al*., 2010). Therefore, admitted extraction data was considered a lower bound to actual extraction; potentially useful to identify extraction trends, not absolute levels. Since the number of respondents admitting extraction of park resources was higher closer to the park boundary ($r_{Spearman}$=-0.322, $p<0.001$, n=573), the number of households within 1 km of the park, the typical upper bound distance to travel for forest resources (Naughton-Treves *et al*., 2007), was used as a variable to capture the number of households that might be opportunistically more prone to resource extraction.

### 4.2.3 Analysis

Since this study deals with illegal extraction, there is the potential for local villagers to be punished for illegal entry into the park should the study villages be explicitly identified (Robbins *et al*., 2006). To protect village identity, geographic masking (Leitner & Curtis, 2004; Armstrong & Ruggles, 2005), using interpolation to rasterize data to a 1,000m grid and then re-sampling to a resolution of 30m within the boundary measurement zone, was used for graphic representations of presented results.

All correlations quoted for the household scale are non-parametric (Spearman) because household scale data was not normally distributed. However, once aggregated to the village scale, most socio-economic variables, with the exception of sales to tourist facilities, were normally distributed, permitting parametric (Pearson) correlations to be used. Measured illegal extraction data was normally distributed for tree harvesting and entry trails, but not normally distributed for in-park grazing and poaching signs.

The variables chosen to be included in the village-scale regression modelling were based on findings reported in the literature and data from this study. Population density was found to be a predictor of tree harvesting in many countries in Africa (Cropper & Griffiths, 1994), exacerbated by in-migration (Newmark & Hough, 2000). However, people with more years of education tended to support conservation (Infield, 1988), potentially mitigating illegal extraction from the PA. Deforestation in Tanzania extended outward from large urban centres (Ahrends *et al*., 2010), a phenomenon represented by road distance between the village and Fort Portal in this study. Findings about the influence of wealth on conservation attitudes have been contradictory with higher capital asset wealth and higher income generating households favouring conservation (Infield, 1988; Holmes, 2003), but households with more livestock tend not to support PAs (Kideghesho *et al*., 2007). Therefore, all of the individual components of wealth (land, buildings, livestock, communication and transportation devices, employment rate, and income generating activities) were separately included in the regression model. Also, based on data from this study showing admitted extraction was higher near the park boundary, the number of households within 1 km of the park boundary was included as a regression variable.

Regression models based on these eleven variables were created for each measure of illegal extraction: number of trees harvested per km of boundary, number of illegal entry trails per km of boundary, number of livestock observed grazing in the park, and number of poaching signs found. Variables were then systematically removed from each model using backward step-wise regression, based on variable F-test significance ($p < 0.1$), leaving only the variables that significantly predicted the dependent illegal extraction variable. Spatial autocorrelation was not exhibited by the regression models (Moran's I, $p < 0.05$).

## 4.3 Results

### 4.3.1 Socio-economic landscape

**Population Density (Fig. 4-2a):** On average study households were home to six people, typically two to three adults and three to four children. The number of children under one year of age indicated a live birth rate of 53 per 1,000 people. Fifty-two percent of the population was under the age of 15, indicating a strong potential for natural population growth when coupled with the high birth rate (Bergman, 1995). Village population density ranged from 70 to 611/km$^2$ (Table 4-1). Fifty-six percent of household heads had moved to the study villages from elsewhere, many coming from the district of Kabale in southern Uganda.

Village population densities tended to be greater closer to FACE plantations ($r_{Pearson}$=-0.465, p=0.019, n=25), as was in-migration ($r_{Pearson}$=-0.569, p=0.003, n=25), suggesting people may be moving to the area in hopes of being employed. The shortest straight line distance from a village to either a FACE plantation, tourist facility, or research station, correlated with village population density ($r_{Pearson}$=-0.501, p=0.011, n=25), with denser population occurring closer to park-generated employment.

**Education (Fig. 4-2b):** Only 24% of the adults in our survey households had completed primary school. Men had completed an average of 5.4 years of school, while women were generally less educated, having completed a mean of 3.4 years of education. The village-mean years of education for adults was 4.3 years ranging from 2.8 to 6.4 (Table 4-1).

**Wealth (Fig. 4-2c):** A typical household compound contained one primary and one secondary dwelling, with the number of out-buildings varying from zero to seven. Ninety-four percent of survey households claimed ownership (customary and freehold) of land and 26% owned land both in the study village and elsewhere. The average household owned of 4.1 ha (median= 2 ha), but 6% of survey households owned greater than 10 ha, and 34% of the households owned less than 2 ha. To increase land available for cultivation, 39% of households rented or borrowed land from others. Larger households tended to own more land ($r_{Spearman}$=0.284, p<0.001, n=561), as did more educated households ($r_{Spearman}$=0.217, p<0.001, n=561).

Chickens were the most commonly owned livestock (82%), followed by goats (64%), pigs (46%), cows (20%), and sheep (11%). Households that owned more land tended to own more livestock (CEU: $r_{Spearman}$=0.461, p<0.001, n=567). Most households owned at least one radio (83%), and 39% had a cell phone even though none had electricity. For transportation, 53% of households owned a bicycle, but only 8% owned a motorcycle. Larger ($r_{Spearman}$=0.256, p<0.001, n=586) and more educated households ($r_{Spearman}$=0.308, p<0.001, n=580) tended to own more motorcycles, bicycles, radios and cell phones.

The mean capital asset wealth in our survey households was US$7,423 (median=US$5,033), but ranged from a low of US$119 to a high of US$132,541, indicating considerable wealth stratification within communities. Land holdings (58%) and the value of buildings (33%) contributed most to household capital asset wealth; livestock value (8%) and communication/transportation devices (2%) contributed far less.

**Income-Generating Activities (Fig. 4-2d):** Only 30% of households had members with off-farm employment. Most households earned income by selling excess food production (84%), growing cash crops (40%), owning woodlots (18%), owning a retail shop (13%), making honey (7%), selling food, fuel or crafts to tourism (4%), and by supporting other subsistence households by selling firewood, charcoal, bricks, or home brewed alcohol (12%). Larger ($r_{Spearman}$=0.330, p<0.001, n=580) and more educated households ($r_{Spearman}$=0.294, p<0.001, n=575) had more human capital and thus tended to be engaged in more income-generating activities. The average

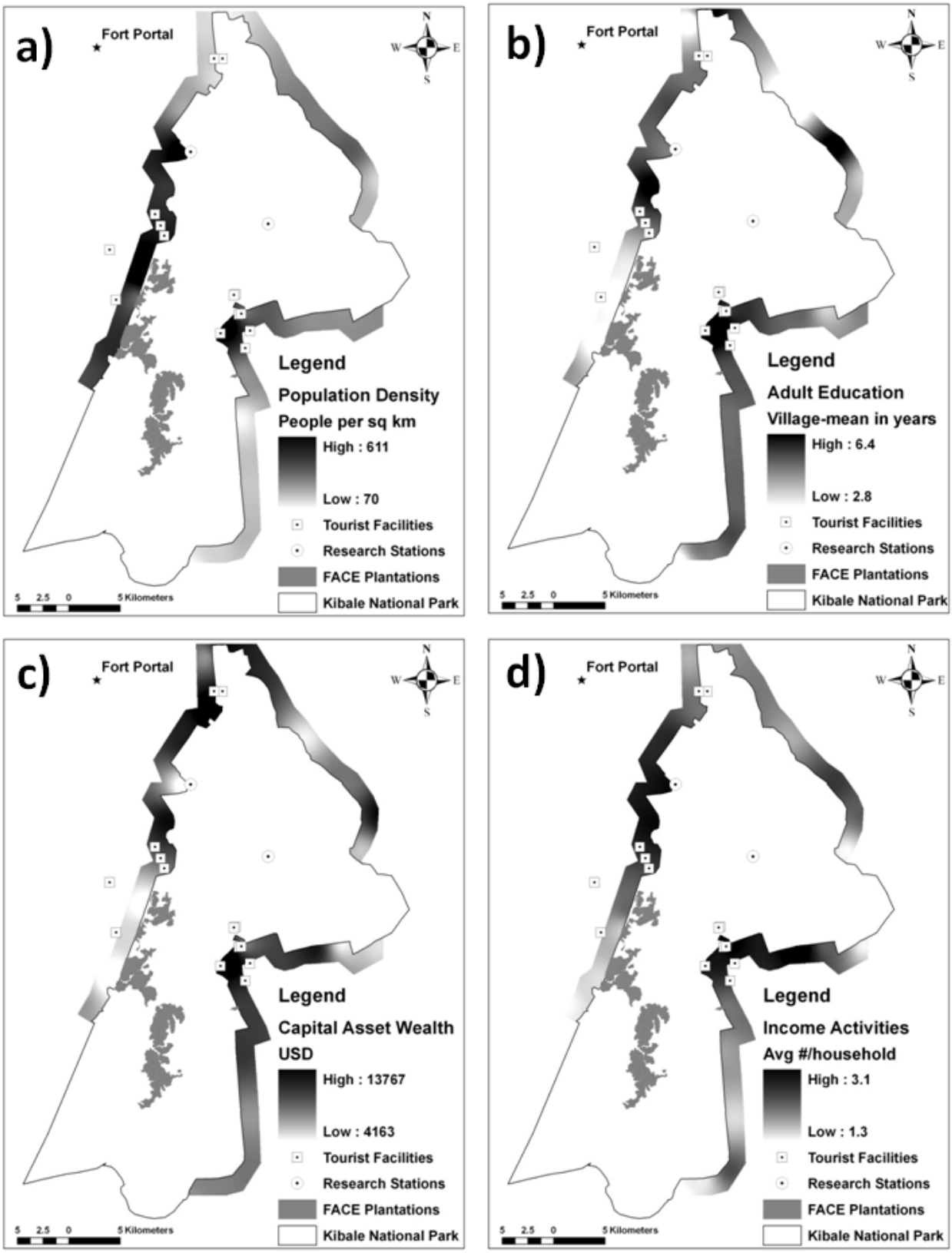



**Figure 4-2: Spatial distribution of socio-economic variables around Kibale National Park**

**Table 4-1: Socio-economic study variables for 25 villages around Kibale National Park, Uganda**

[1] for persons 15 and older

[2] Building Wealth Unit (BWU) = Value of all buildings in household compound divided by the value of a medium category primary dwelling

[3] Cattle Equivalent Unit (CEU) = Value of all livestock in a household divided by the market value of one cow

[4] Bicycle Equivalent Unit (BEU) = Summed value of radios, cell phones, bicycles and motorcycles divided by the market value of one bicycle

| *Indicator* | *Variable* | *Mean* | *Min* | *Max* |
|---|---|---|---|---|
| Population | Population density (people/ sq km) | 241 | 70 | 611 |
| | In-migration (% household heads who moved to KNP) | 56 | 21 | 91 |
| | Number of households within 1km of park boundary | 60 | 13 | 124 |
| Education | Village mean years of education for all adults | 4.3 | 2.8 | 6.4 |
| | Village adult literacy (% finishing primary school)[1] | 24 | 7 | 48 |
| | Years of education - men | 5.2 | 0 | 16 |
| | Years of education - women | 3.3 | 0 | 14 |
| Wealth | Village mean household capital asset wealth (USD) | 7777 | 4 163 | 13767 |
| | Household capital asset wealth (USD) | 7423 | 119 | 132541 |
| | Village mean building wealth (BWU)[2] | 1.9 | 0.9 | 3.2 |
| | Building wealth per household (BWU) | | | |
| | Village average land owned per household (ha) | 4.1 | 1.9 | 8.3 |
| | Land owned per household (ha) | 4.1 | 0 | 100 |
| | Village averaged cattle equivalent units per household (CEU)[3] | 2.1 | 0.3 | 11.8 |
| | Cattle Equivalent Units per household | 2.1 | 0 | 213 |
| | Village averaged bicycle equivalent units per household (BEU)[4] | 1.8 | 0.7 | 4.2 |
| | Bicycle Equivalent Units per household | 1.8 | 0 | 48 |
| Income Opportunities | Employment (% survey adults employed) | 22 | 2 | 51 |
| | Village mean number of income opportunities per household | 2.1 | 1.3 | 3.1 |
| | Percent of households in village: | | | |
| | Selling excess food crops | 84 | 54 | 100 |
| | Growing cash crops (tea, tobacco, coffee, vanilla, cocoa & Artemisia) | 41 | 8 | 91 |
| | Owning woodlots | 18 | 0 | 58 |
| | Owning a retail shop | 13 | 0 | 29 |
| | Producing honey | 7 | 0 | 26 |
| | Selling food, fuelwood/charcoal or crafts to tourism | 4 | 0 | 31 |
| | Selling firewood | 3 | 0 | 25 |
| | Making charcoal | 5 | 0 | 42 |
| | Making bricks | 3 | 0 | 13 |
| | Making alcohol | 4 | 0 | 22 |
| Market Access | Road distance to Fort Portal (Km) | 41.78 | 14.42 | 77.10 |

number of income-generating activities engaged in by households within a village increased with population density ($r_{Pearson}$=0.445, p=0.026, n=25).

Villages with higher mean years of adult education tended to have more income generating-activities ($r_{Pearson}$=0.557, p=0.004, n=25). As village employment rates rose, the percentage of households selling excess

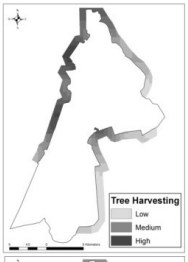
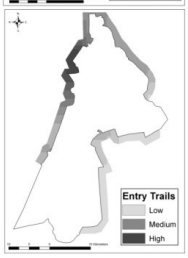
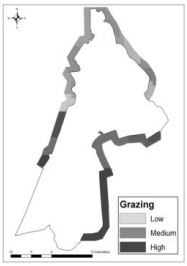
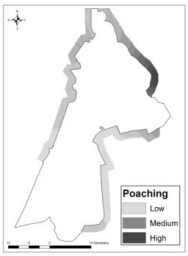
food crops dropped ($r_{Pearson}$=-0.670, p<0.001, n=25), and the prevalence of income-generating activities rose ($r_{Pearson}$=0.416, p=0.038, n=25), as money earned through employment was invested in small businesses.

**Market Access:** Most roads around KNP are dirt roads, some improved with a murram (laterite) layer similar to gravel. The best murram road runs from Fort Portal to Kamwenge Town (Fig. 4-1), with access to research, tourism, and carbon sequestration operations. There are two well-paved roads, one connecting Fort Portal to the capital, Kampala, and the other connecting Fort Portal to a southern town, Kasese (Fig. 4-1). Fort Portal is the administrative centre for the district of Kabarole and the seat of the Toro Kingdom, acting as a market hub for the region. The average road distance from a study village to Fort Portal was 42 km, but ranged from 14 km to 77 km. Villages closer to Fort Portal had higher employment ($r_{Pearson}$=-0.603, p=0.001, n=25), and more income opportunities were engaged in by households ($r_{Pearson}$=-0.424, p=0.035, n=25), presumably as households accessed urban markets.

### 4.3.2 Illegal extraction

Within the 128 km long boundary measurement zone (Fig. 4-1), 19.5 km of boundary were sampled. Tree extraction was found adjacent to all 25 study villages, with high levels of extraction clustered along the western boundary (Table 4-2). Seventy-five illegal entry trails were recorded, many found along the northwest border of the park where up to 20 trails were observed along one kilometre of park boundary. Herds of goats (<20 animals), and cows (100-200 animals) were observed grazing inside the park boundary near nine of the study villages, most in the

**Table 4-2: Illegal resource extraction from the boundaries of Kibale National Park, Uganda**

| | | | |
|---|---|---|---|
| **Number of Harvested Trees** | **Mean per village (± SD)** | 112 (± 99) |   |
| | **Total recorded** | 2794 | |
| | **Per km of boundary** | 143.6 | |
| | **Number of villages where disturbance was found** | 25 | |
| **Number of Entry Trails** | **Mean per village (± SD)** | 2.8 (± 3.7) |   |
| | **Total recorded** | 73 | |
| | **Per km of boundary** | 3.8 | |
| | **Number of villages where disturbance was found** | 15 | |
| **Number of Livestock Grazing in Park** | **Mean per village (± SD)** | 13.8 (± 41.9) |   |
| | **Total recorded** | 373 | |
| | **Per km of boundary** | 19.0 | |
| | **Number of villages where disturbance was found** | 9 | |
| **Number of Poaching Signs Found** | **Mean per village (± SD)** | 0.89 (± 2.8) |   |
| | **Total recorded** | 24 | |
| | **Per km of boundary** | 1.2 | |
| | **Number of villages where disturbance was found** | 6 | |

southern half of the park. Only 24 signs of animal poaching (pit-fall traps and snares) were found during boundary measurements, near six villages, with the highest density found along the northeast boundary of the park. Animal poachers have been known to travel deep into the park to hunt (PAWAR, 2009), so poaching along the boundaries of the park may not represent the true magnitude.

### 4.3.3 Socio-economic variables and measured extraction

Illegal tree harvesting was a function of demand, proximity to Fort Portal and wealth (Table 4-3). Tree extraction increased the closer a village was to the urban centre, suggesting off-take from the park near Fort Portal is not only for local consumption but also for gain by supplying Fort Portal. The demand for wood from the urban centre is supported by proportionally more households making charcoal, selling firewood and owning woodlots within 20 km of Fort Portal (Table 4-4). The number of households within 1 km of the park represents a more localized demand for wood. The influence of wealth variables was contradictory. High ownership of transport and communication devices lowering predicted illegal tree harvesting, while employment rate, higher near Fort Portal, predicted increased illegal tree off-take.

The number of illegal entry trails was strongly predicted by road distance from Fort Portal (Table 4-3), reinforcing that illegal off-take is augmented by urban market demand. Contrary to prior findings in the literature, higher years of adult education also predicted higher numbers of illegal entry trails (Table 4-3), but high in-migration to the village predicted lower numbers of entry trails. Capital asset wealth variables (land, building worth, and transportation and communication devices) consistently predicted fewer illegal entry trails.

The number of livestock grazing in the park was predicted by the average level of livestock ownership in a village, suggesting in-park grazing is primarily driven by local village demand (Table 4-3). A statistically significant model for poaching could not be generated, due to poaching signs only being found near six villages.

**Table 4-3: Linear regression analysis using socio-economic variables to model illegal resource extraction**

| *Resource* | *Model Variable* | *Standardized β* | *Model statistics* |
|---|---|---|---|
| Harvested | Road distance to Fort Portal | -0.434 | Adjusted $R^2$= 0.553 |
| Trees | Transport/communication devices BEU | -0.454 | F= 8.423, p< 0.001 |
| per km of | Employment rate | 0.317 | Moran's I= 0.129, p=0.235 |
| Boundary | Number of households within 1km of park | 0.300 | |
| Illegal | Road distance to Fort Portal | -0.529 | Adjusted $R^2$= 0.620 |
| Entry Trails | Migration rate | -0.403 | F= 7.529, p< 0.001 |
| per km of | Adult education | 0.628 | Moran's I= 0.316, p=0.054 |
| Boundary | Land owned | -0.364 | |
| | Building wealth | -0.499 | |
| | Transport/communication devices BEU | -0.385 | |
| Livestock | Cattle ownership CEU | 0.718 | Adjusted $R^2$= 0.494 |
| grazing | | | F= 24.425, p< 0.001 |
| in-park/ village | | | Moran's I= -0.155, p=0.284 |

**Table 4-4: Influence of distance to Fort Portal on tree extraction and wood based income activities**

| | *Road distance to Fort Portal* | | *t-test* |
|---|---|---|---|
| *Average of village-scale variables* | *≤ 20 km* | *> 20 km* | *p* |
| Number of illegally harvested trees/km of boundary | 261 | 134 | 0.087 |
| Percent of households making charcoal | 12.6 | 2.9 | 0.035 |
| Percent of households selling firewood | 8.4 | 1.7 | 0.015 |
| Percent of households owning woodlots | 29.4 | 15.2 | 0.057 |

## 4.4 Discussion

The extraction of woody biomass from PAs has been attributed to subsistence livelihood needs of local communities (Masozera & Alavalapati, 2004), with many studies supporting the finding that poor households are more dependent on forest products (Reddy & Chakravarty, 1999; Fisher & Shively, 2005; Mamo *et al*., 2007). However, wealthier households, although less dependent on forest products (Lepetu *et al*., 2009), actually want more access to forests (Holmes, 2003) and extract larger quantities of wood for timber sales and conversion to charcoal (Mamo *et al*., 2007); although this extraction is often dependent on access to urban markets (Fisher & Shively, 2005; McDonald *et al*., 2009). Our findings suggest that the households within 1 km of the park boundary represent local off-take, either to support their own subsistence livelihood or for sale, and that higher tree off-take is also driven by demand from the urban centre of Fort Portal.

A meta-analysis of case studies conducted by Geist and Lambin (2002) to identify the cause of deforestation in tropical countries found that no one cause could account for tropical deforestation, and in most locations, multiple causes contributed to forest loss. Our results add to the ever-growing number of studies citing the causes of deforestation and pressure on PAs in Africa as multifaceted. Although village demographics influenced tree harvesting, the effect was limited to households located close to the park, while demand from urban centres drove both tree harvesting and illegal entry into the park (Fisher & Shively, 2005; Jumbe & Angelsen, 2007; Mamo *et al*., 2007).

We did find increased capital asset wealth, excluding livestock ownership, was a mitigating factor for tree extraction and illegal entry trails (Cropper & Griffiths, 1994), but illegal in-park grazing was observed near villages with high livestock wealth (Infield *et al*., 1993; Kideghesho *et al*., 2007). These findings suggest an inconsistent influence of household wealth on illegal extraction of resources from PAs. Increased wealth may improve attitudes towards PAs (Infield, 1988; Gillingham & Lee, 1999), but these attitudes are only translated into lower illegal extraction if the source of wealth accumulation is not linked to resources available within the park, a finding consistent with a study in Tanzania (Gillingham & Lee, 1999). If household members are employed outside the home, then the redistribution of labour within the household may reduce the time available for wood collection, making the park an easier option for finding fuelwood. This might explain why higher employment rates predicted higher tree extraction from KNP.

We also found that population density was highest near tourism, research and carbon sequestration operations, supporting the argument that park-based employment may be contributing to increased anthropogenic pressure on PAs (Wittemyer *et al*., 2008), as a result of in-migration near park-based economic opportunities (De Sherbinin & Freudenberger, 1998). Although once attracted to KNP, the abundant rainfall and fertile volcanic soils allow households to rely on subsistence agricultural yield off relatively small plots of land. UWA and local governments therefore need to consider the potential migratory pull of park-based enterprises and plan to provide alternative sources of fuelwood, such as woodlots, for communities adjacent to these operations.

Our results highlight that illegal tree harvesting from KNP, charcoal making, firewood selling, and owning woodlots are significantly higher near the urban centre of Fort Portal, indicating that urban demand for fuelwood could be a threat to the forest habitat within KNP. A longitudinal study of deforestation around Dar-Es-Salam in Tanzania illustrated how demand for wood from a growing urban centre spatially extends deforestation (Ahrends *et al*., 2010). Although the boundaries of KNP appear to be remaining intact, the landscape around the park has become more fragmented with forest patches reducing in size or disappearing altogether (Hartter & Southworth, 2009). As Fort Portal grows, more of the boundary of KNP could be under pressure from tree poaching. Longitudinal analysis of remote sensing images to identify the spatial extent of forest fragment loss and growth of woodlots, as a function of distance from Fort Portal, could help estimate the spatial growth rate of the urban demand for wood. This information could be used by UWA, in partnership with local governments, to develop strategies to minimize tree harvesting within the park. The Kabarole district council has already issued an ordinance supplement stating “all land owners shall plant trees on at least 10% of the acreage of his or her land as advised by Council” (Kabarole District, 2006, p. 10),

suggesting the council is open to finding solutions to reduce wood extraction from KNP.

Although in-park grazing is illegal, there is a need for further research to understand the risks to conservation caused by cattle grazing in the park, especially in times of drought when livestock owners are most in need of grazing pastures (Neumann, 1998). Typically the most cited risk is the transmission of livestock disease to park protected animals (Goldberg *et al*., 2008), or the risk of fires set by livestock herders to stimulate grass growth. However, a study in KNP identified forest encroachment on in-park grasslands to be adversely affecting small antelope populations (Lwanga, 2006), and in-park livestock grazing could help maintain these grassland areas. The benefits and risks of allowing local people to graze livestock in the park need to be better understood to determine if negotiated access for communities near these grasslands could be considered.

## 4.5 Conclusions

Illegal extraction of trees and illegal entry trails into the park appear to be driven by subsistence demand within 1km of the park boundary, and for-profit market demand primarily from Fort Portal. In-park livestock grazing was also demand driven, being highly dependent upon the number of livestock owned. With the exception of livestock wealth and off-farm employment, village-mean household wealth tended to mitigate illegal resource extraction. Contrary to studies citing improved education as a driver of PA acceptance, villages with higher mean years of adult education tended to have more illegal entry trails into the park. Areas of high human population density were found close to enterprises providing park-based employment, highlighting a need for fuelwood supply near park-based employment to help mitigate illegal resource extraction from KNP.

Human pressure on forest PAs needs to be managed by understanding the local and regional drivers of resource extraction, and addressing, in partnership with local governments, both the rural and urban demand for natural resources, especially fuelwood. This understanding of the socio-economic drivers of extraction will support the development of conservation strategies that protect forest habitat, help meet the resource needs of local communities, and manage the threat of urban resource demand.

# 5 Trenches like Fences make Good Neighbours: Revenue Sharing around Kibale National Park

In the household survey, participants most often quoted ecosystem services (97%), illegal resource extraction from the park (25%), and the revenue sharing program (23%) as benefits of living next to KNP. The revenue sharing program is a conservation strategy implemented specifically to improve conservation attitudes while providing benefit to local communities. In this chapter, I describe the management process used to implement the program and distribute the revenue, and assess whether the program is having any influence on illegal resource extraction from the park. This chapter in a revised form was published in *The Journal for Nature Conservation* 20, 82-100 following the final submission of this dissertation.

**Summary:** Conservation compensation programs aim to balance the disadvantages people encounter living next to protected areas while fostering improved conservation behaviours. In Uganda, 20% of protected area entrance fees are shared with the local government to pay for projects to benefit communities adjacent to national parks. The process to distribute the funds and implement projects was investigated by interviewing Uganda Wildlife Authority wardens, local government and village members around Kibale National Park in Uganda. The perceived benefit of revenue sharing by officials and local communities was collected through interviews and a household survey, while the influence of the program on conservation objectives was assessed by measuring illegal tree extraction, entry trails, livestock grazing and poaching signs inside the boundary of the national park. It was found that the program is evolving into an effective mechanism for sharing benefits, but that better project management and increased accounting transparency could further improve the program. If the projects specifically dealt with the problem of crop raiding by park-protected animals, villagers did perceive substantial benefit and lower levels of illegal activity were found inside the park. Generally household perceived benefit was too low to affect conservation outcomes, however reduced in-park illegal activity was recorded where the village chairperson perceived higher benefit from the program implying that the village leadership may be influencing the conservation behaviours within the community. Broad implications derived from this study for revenue sharing program management are: to ensure community involvement in the project decision making process, to try to balance the monetary distribution against the losses incurred by local communities as a result of living next to the protected area and to provide program management training to those responsible for project implementation to avoid perceptions of mismanagement, and loss of trust in the program by local communities.

## 5.1 Introduction

In areas where species co-exist with the ever increasing pressures of human expansion and anthropogenic change, biodiversity conservation often relies on local perceptions of the economic benefits and losses associated with wildlife protection (Ninan *et al*., 2007). Setting aside land to protect species habitat is the primary focus of conservation strategies; however, local communities can perceive the loss of access to these lands as limiting their ability to survive and build economies based on natural resource extraction from, or agricultural development of, these protected areas (Adams & Infield, 2003).

Communities situated on the boundaries of protected areas often disproportionately bear the cost of conservation (Nyhus *et al*., 2005; Ninan *et al*., 2007). Protected animals may roam outside the park boundaries, trampling and eating crops, predating livestock and even maiming local residents (Nyhus *et al*., 2000; Treves & Karanth, 2003). The cost of these incursions can be high for subsistence farmers. Conservation compensation is a policy that attempts to partially off-set these losses for communities that reside next to protected areas. Such compensation can benefit the poor by providing economic benefit in exchange for good conservation behaviour, although since the compensation is usually less than the incurred loss, this can be perceived as "a form of economic coercion rather than a just resolution to resource management conflict" (Schroeder, 2008, p. 592). In sub-Saharan Africa, compensation if it exists is typically a percentage share of revenues from hunting (Lewis & Alpert, 1997) or eco-tourism (Alpert, 1996; Archibald & Naughton-Treves, 2001; Schroeder, 2008).

Ugandan conservation policy is dependent upon enforcement of park boundaries, yet the Uganda Wildlife Authority (UWA) has been moving towards a more participatory conservation strategy. UWA introduced conservation compensation in 1996 and is currently legislated to share 20% of park entrance fees with local governments for the benefit of communities located in parishes directly adjacent to protected areas (Uganda Wildlife Statute, 1996). The Ugandan program's objective is to improve relations with neighbouring communities, demonstrating that conservation can provide economic benefit, in the hope that local people will protect the park (UWA, 2000). To date, US$750,000 has been disbursed nationally to the local government councils around Uganda's National Parks.

Other countries, besides Uganda, have introduced revenue sharing. In Madagascar, a revenue sharing program was piloted in Ranomafana National Park where park revenues were equally shared between the conservation authority and local communities, creating a mechanism for local people to participate in conservation while simultaneously benefitting from community development (Peters, 1998). Revenue sharing has also been introduced in Kenya and Tanzania however the percent that is shared with local residents is typically less than 15% (Alpert, 1996; Honey, 1999). The effectiveness of these schemes has been dependent on having a local governance structure capable of executing the program with the necessary transparency so as not to be construed as coercion (Alpert, 1996; Schroeder, 2008).

Revenue sharing in Uganda was studied during the early days of program implementation, finding that the program improved relationships between park authorities and local communities and had the potential to positively influence conservation attitudes (Archibald & Naughton-Treves, 2001). However the focus on funding schools and health clinics, coupled with the small amount of money disbursed did not appear to reduce illegal extraction activities and benefits needed to be allocated to those who lost most due to the existence of the park (Archibald & Naughton-Treves, 2001; Mugisha, 2002; Chhetri *et al*., 2003). Conclusions on the effectiveness of the revenue sharing program were mixed, with generally positive opinions provided by UWA staff and government officials (Archibald & Naughton-Treves, 2001), while household surveys found little difference in conservation attitudes between those who benefitted from revenue sharing and those who did not (Mugisha, 2002). This lack of attitude change could have been due to a lack of knowledge that the funding source was the national park (Archibald & Naughton-Treves, 2001).

Focusing on the revenue sharing program implementation around Kibale National Park, this paper provides an update of the Ugandan revenue sharing program ten years after the first distributions occurred and answers the following questions: What projects have been funded?, How are the projects chosen and implemented?, Is the program perceived as beneficial by officials and communities near the park?, and Does the revenue sharing program influence conservation outcomes? All previous revenue sharing studies in Uganda (Archibald & Naughton-Treves, 2001; Mugisha, 2002; Adams & Infield, 2003), have used interview and survey based attitudes to assess the influence of the program on conservation outcomes. However, since attitudes do not necessarily translate into actions, I use measured levels of illegal resource extraction within the boundaries of the park to assess whether the revenue sharing program does support conservation objectives.

## 5.2 Methods

### 5.2.1 Study site

This study was conducted around Kibale National Park (KNP), located in western Uganda (Fig.5-1). Gazetted as a national park in 1993, the park covers 795 km² and is a rich area for primate biodiversity (Struhsaker, 1997) with habituated chimpanzees being the primary draw for foreign tourists. Although harvested for timber products until the mid 1970s, all commercial logging has stopped, with the exception of limited paid access agreements to extract exotic tree species. However, illegal extraction of resources from the park continues with evidence of animal poaching and harvesting of trees for building materials, fuel wood and charcoal production (Struhsaker, 2002; Naughton-Treves *et al*., 2007). Conservation policy is defined through national government legislation (Uganda Wildlife Statute, 1996) and is typically enforced through arrests and fines when local residents illegally enter the park. UWA also relies on access agreements for local communities to legally extract certain forest resources, community outreach and education, as well as the revenue sharing program to improve conservation behaviours.

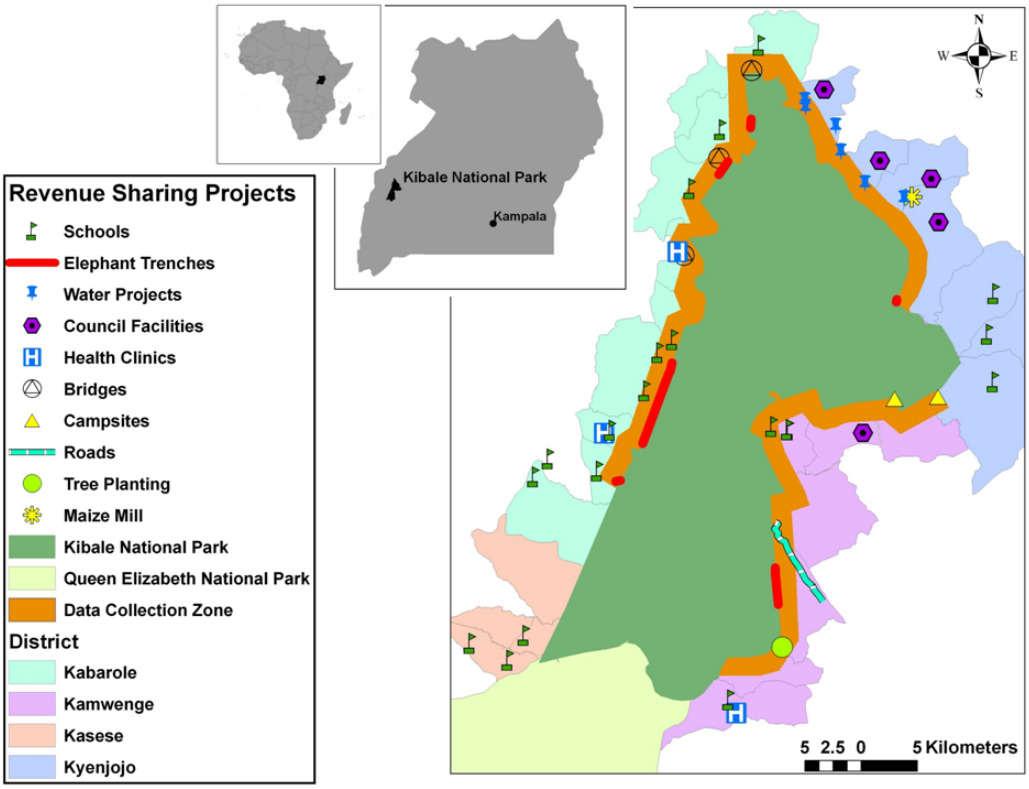



**Figure 5-1: Kibale National Park, adjacent parishes & revenue sharing locations**

*Source: Parish Shapefiles* (Hartter, 2007)

In Uganda, five council levels make up the elected hierarchical decentralized governance structure (Saito, 2003). The highest is the district council, followed by the county, sub-county, parish, and the village council. Four districts surrounding KNP benefit from the tourist proceeds through the revenue sharing program: Kabarole, Kamwenge, Kyenjojo and Kasese (Fig. 5-1). Kasese was excluded from this study based on proximity to Queen Elizabeth National Park and having only received 8% of the revenue sharing funds distributed by KNP. In the remaining three districts, 25 parishes border KNP within which 25 villages volunteered to participate in the study from May to August in both 2008 and 2009. Sixteen of the villages had benefitted from a revenue sharing project either in or near their village, while nine villages had not.

Local people live in villages of about 100 households, governed by the village council, which is led by a village chairperson, who manages the community and when necessary, disciplines village members. For this study, a village is defined by the spatial extent of households associated with a village name under the leadership of one village chairperson. Since benefits from the revenue sharing program are limited to parishes located next to the park, study villages were chosen based on some village members holding and/or cultivating land directly adjacent to the park boundary. Villages were located approximately every 5 km within the data collection zone (Fig. 5-1), but are not explicitly identified, since the reporting of illegal resource extraction might lead to retribution from the wildlife authority (Robins *et al.*, 2006). Most local inhabitants are subsistence farmers, although additional income is available from cash crops and off-farm work on tea plantations, planting trees for a carbon off-set program, as research assistants and in the tourism industry (Mulley & Unruh, 2004; Hartter, 2010).

### 5.2.2 Data collection and analysis

**Revenue Sharing Project Location & Valuation:** To map the spatial extent of the revenue sharing program, a global positioning system (GPS) waypoint was recorded for each revenue sharing project, located from a monetary disbursement record provided by UWA. The monetary benefit assigned to each study village was

based on use of the project by the village. So, if only 20% of the enrolled students at a funded school came from the study village, only 20% of the money was considered to have benefitted the village. Alternatively, if an elephant (*Loxodonta africana)* trench built between a study village and KNP to help keep the elephants from destroying village crops was funded, then the full cost of the trench was assigned to that village.

**Revenue Sharing Program Knowledge & Perceptions:** A semi-structured interview was conducted in English with four UWA wardens: a national community warden, the past and current chief wardens of KNP, and the community warden of KNP. Three levels of local government were included in the study by interviewing three district, 10 sub-county and 25 village chairpersons. Although UWA officials and district chairpersons spoke English, most of the sub-county and all village chairpersons had limited or no English fluency. Therefore sub-county and village chairperson interviews were usually conducted through a Ugandan interpreter with another assistant present to listen and participate in a post interview debriefing (Freed, 1988). A structured set of questions were asked of all respondents, to determine the knowledge of the program, operational process and perceived influence on conservation behaviours. The benefit of the program for local people, as perceived by the interviewed official, was quantified using a five point Likert scale with answers ranging from no benefit (1) to a lot of benefit (5).

A focus group was conducted in 15 of the study villages (60%) to understand the extent of knowledge, perceived benefit and influence on conservation attitudes of the revenue sharing program. The village chairperson, as per cultural norms, approved and organized the meetings, inviting men and women from a range of age groups representing a cross-section of the village. Since the meetings were held in the open, the size of the focus group often grew beyond those invited, as a result, the total number of people present ranged from 16 to 51. Women were in attendance at 14 of the focus groups, where they represented 6 to 65% of the participants. Discussion was facilitated by a senior field assistant in the local tribal languages, translating the responses verbally during the meeting, while another assistant recorded all that he heard from the villagers, allowing triangulation of notes for a more complete record of the meeting. The meeting facilitator tried to ensure that all members of the meeting had a voice, creating a good rapport with male and female participants and asking the women for their opinions if they did not initially speak up.

Qualitative thematic analysis was used to understand the perceptions about the revenue sharing program from the five groups interviewed: UWA, district, sub-county and village chairpersons and village focus groups, while content analysis was used to quantify structured interview transcripts related to knowledge of the revenue sharing program structure and process.

**Household Awareness & Perceived Benefit of Revenue Sharing:** In each of the 25 study villages, 24 household surveys were conducted to answer the following questions: 'Are you aware of any projects in your region that have been funded by the revenue sharing program (Yes/No)?', 'How much does your household benefit from revenue sharing projects (not at all, a little, some, reasonably or a lot)?', 'Since you live next to the park, do you consider the park to be a very good, good, neutral, bad or very bad neighbour?'

A total of 596 surveys were collected using a stratified random sample, wealth stratified in each village based on house construction categorization, where mud and wattle construction indicated a poorer household and brick construction a richer household (Hartter, 2009). The survey was administered by four Ugandan field assistants, three men and one woman, in two local languages. Translation/back-translation (Werner & Campbell, 1973) was used, thereby highlighting interpretational discrepancies and ambiguous interpretations of the questions.

Survey data were averaged at the village level for comparison with project monetary value and observed illegal extraction, since these variables could not be attributed to any one household. Since the type of project implemented might influence the household perceived benefit of the program, the household perceived benefit of revenue sharing was compared using a Kruskal-Wallis test (non-parametric analogue of ANOVA), based on the type of project implemented in the village. Only projects attributable to more than 30 households were compared, and households in villages with multiple completed projects were excluded. Since groups were of unequal size, as a post-hoc test, all groups were compared pair-wise using Mann-Whitney rank sum

testing. However, since this involved 15 pair-wise tests (6 project type groups), a Bonferroni correction was applied, tightening the criteria for significance to $p<0.003$.

**Illegal Extraction:** To assess the influence of the revenue sharing program on conservation behaviours, data were collected on illegal resource extraction. Using a method developed from an assessment of human disturbance in Bwindi Impenetrable National Park (Olupot *et al*., 2009), the boundary of KNP was accessed adjacent to each village and an observational walk of 600 to 850 m was conducted with the aid of skilled field assistants. Along the boundary the following data were recorded: number of trees completely or partially harvested, number of human entry trails, number of livestock seen grazing inside the park, and number of poaching signs (*e.g.*, pit traps and snares). Each entry trail was followed to its terminus to record the same information, unless the trails were created for researchers or UWA patrols (as confirmed by UWA rangers), then only the first 200 m was checked for illegal activities.

Illegal trails and tree harvesting were found on all observational walks, however, higher concentrations were observed along the western boundary of the park. Livestock grazing inside the park was primarily in the southern half of the park, while animal poaching was highest along the north-eastern boundary. Therefore, to represent an overall measure of illegal access for each village, the illegal resource extractions from the boundary walk for that village were combined into a single index of human disturbance for each village.

$$HDI_i = \frac{TH_i - \overline{TH}}{\sigma_{TH}} + \frac{ET_i - \overline{ET}}{\sigma_{ET}} + \frac{IPG_i - \overline{IPG}}{\sigma_{IPG}} + \frac{P_i - \overline{P}}{\sigma_P} \qquad (4\text{-}1)$$

Where:

$HDI_i$ = human disturbance index for a given village

$TH_i$ = number of trees illegally harvested per km of boundary for a given village

$\overline{TH}$ = mean number of trees illegally harvested per km of boundary for all villages

$\sigma_{TH}$ = standard deviation of the number of trees illegally harvested per km of boundary for all villages

$ET_i$ = number of illegal entry trails per km of boundary for a given village

$IPG_i$ = number of domestic animals seen grazing inside the park for a given village

$P_i$ = number of poaching signs found near a given village

## 5.3 Results

### 5.3.1 Funded revenue sharing projects

Fifty-five revenue sharing projects were implemented between 1999 and 2008 with the US$150,000 distributed around KNP to date (Fig. 5-1). Schools have been the most frequent beneficiaries, building classrooms, student latrines and teacher housing. Seven communities received money to build elephant trenches to reduce crop raiding, some receiving multiple disbursements to extend the trenches. Five council facilities have benefited from the program, upgrading and furnishing four parish halls, and constructing one sub-county headquarters. One health clinic has been constructed and two other health clinics have received rain water collection systems. The remaining projects include: five protected water wells, three bridges, two tourist camps, one road, one tree planting project, and one maize mill. The latest disbursement was released in July 2009 and plans include eight elephant trench projects and six income generation projects to provide livestock, coffee seedlings, and a beekeeping operation to frontline villages. All projects implemented to date have been located within 7km of the park boundary and 36% were within 1km of the park, the area most affected by wild animal crop raiding (Naughton-Treves, 1998).

### 5.3.2 Monetary distribution and project implementation process

The money available for revenue sharing is a function of the number of visitors attracted to KNP and the visitor fees charged. Visitors pay a park entrance fee (US$25 in 2008) and activity fees for chimpanzee trekking (US$80), but only 20% of the entrance fee is transferred to the revenue sharing program. Most chairpersons knew that the amount shared was 20%, but that the revenue was restricted to entrance fees was less well known and least clear when speaking with village chairpersons (Table 5-1). Not everyone is satisfied with the amount reaching local communities, believing UWA could afford more; “The 20% that reaches them is very small. It is like when you are

very hungry and you come across a person with a pan full of cooked bananas and he just gives you one piece…will you really be happy?" (Village Focus Group, 6/7/08).

Thirty percent of the district and sub-county chairpersons interviewed stated they would like to see more revenue shared and villagers asked to understand the accounting of the remaining 80% of the fees. Nationally, UWA revenues still do not cover their operating expenses as the Ugandan government and international donors still provide 48% of the operating budget (UWA, 1999-2007). Thus, there appears little potential for increasing the percentage of money shared with local communities.

If distributed annually, the amount of money is too small to do substantial projects, so for KNP UWA adopted a policy that the disbursement would occur when the funds collected exceeded US$54,000, resulting in disbursements about every two years. The money is distributed based on each parish bordering the park receiving an equal sum; however with 28 parishes bordering KNP, the 2009 distribution translates to about $US2,000 per parish, insufficient to fully fund any meaningful project. Therefore, the sub-counties rotate the benefit between their parishes. This management of the distribution coupled with increased park visitor numbers has resulted in the average funds per project increasing from US$690 in 2003 to US$4,600 in 2009.

In discussions with UWA, local government chairpersons and villagers, the process to decide on what to spend these funds was structured as follows: UWA provide the funds to local government, district level leadership distributes to the sub-counties and audits the process, sub-county leadership make the project decisions, while villagers and their village and parish chairpersons make project proposals for the sub-county's consideration. Although this is the general process, the consistency of implementation varied.

By legislation, UWA has no legal mandate to decide how the money is spent; "UWA can only lobby for pro-conservation projects. We have no right to reject a project decided upon by local government" (UWA Official, 24/7/08). In some communities, UWA attends meetings where project proposals are discussed, however, when asked who influences the project decision only a few village chairpersons responded UWA (Table 5-1). In August 2007, an auditing role was defined for the district level leadership, "due to a lack of accountability at the sub-county level" (UWA Official, 2/6/08). All three district chairpersons interviewed provided a different

**Table 5-1: Knowledge of revenue sharing program by chairpersons**

[1]correct answer as per Uganda Wildlife Statute, 1996

| | *District Chairperson Responses (n=3* | *Sub-County Chairperson Reponses (n=10)* | *Village Chairperson Responses (n=25)* |
|---|---|---|---|
| **Source of Funds** | | | |
| 20% is the amount shared[1] | 100% | 100% | 32% |
| Source is entrance fees[1] | 100% | 40% | 8% |
| Source is all UWA revenues | 0% | 40% | 28% |
| Don't know source of funds | 0% | 20% | 64% |
| **Who Influences Project Decision?** | | | |
| UWA | 0% | 0% | 20% |
| District Chairperson | 67% | 20% | 8% |
| Sub-county Chairperson | 67% | 70% | 68% |
| Parish Chairperson | 33% | 50% | 28% |
| Village Chairperson | 100% | 80% | 56% |
| Villagers | 33% | 40% | 20% |
| **Mandate for Revenue Sharing** | | | |
| Legislation[1] | 100% | 55% | 24% |
| UWA or KNP | 0% | 36% | 40% |
| Negotiated by local government | 0% | 0% | 12% |
| Negotiated by local people | 0% | 0% | 9% |
| Don't know | 0% | 9% | 16% |

understanding of their role, ranging from auditing how the money is spent to approving/vetoing the projects chosen by the sub-counties. Only in the district where the District chairperson said he had "veto power if they do not choose a good project" (5/6/09), did village chairpersons say that the district had any decision making power in the process.

A majority of chairpersons acknowledged that deciding on the project was the role of the sub-county leadership (Table 5-1). However, the level of autonomy in the sub-county decision varied. In most communities, "village chairmen are called to a meeting with the sub-county to give ideas and then decide on priorities" (Village chairperson, 10/6/08), however, in some sub-counties "parish level councillors get together with the sub-county and village chairmen are not involved except for proposing what to spend the money on" (Village chairperson, 14/7/08). Limiting the decision process to the parish and sub-county leadership led to resentment from villagers because "those who decide about the revenue sharing project live far from the park and they do the planning without living with the problems of being next to the park" (Village Focus Group, 23/6/08). Although the village was usually represented by the village chairperson, in a few sub-counties a majority decision was taken in an open meeting where all residents bordering the park attended. This approach appears more prevalent in recent decisions; "Before, the money stopped at the sub-county or parish and they decided. This is why we got a parish hall. Now, the locals are consulted and they decide what the money should be used for" (Village chairperson, 8/6/09).

Many people were concerned that the funds were being mismanaged, a perception fuelled by a lack of visibility of the funds, poorly managed projects and a sense that sub-county leadership lacked empathy for the local people. Ironically, the concern over mismanaged funds partially results from an accounting practice of local government requiring all construction projects over US$540 be tendered. The Chief Administrative Officer of one district clarified that "if local leadership can mobilize the community then money should go straight to tools and paying the community, if not then they must go through the tender board to ensure competition" (4/8/08). One project highlighted just how quickly the money can be reduced if tendered and contracted, as only 100m of elephant trench was excavated instead of the 2 km that could have been excavated using local village labour.

It was primarily in villages where unfinished projects were located where resentment towards the sub-county's handling of funds was present. Sixteen of the 55 revenue sharing projects were found to be unfinished, abandoned, or not yet operational in 2009. Construction of one health clinic had been progressing since 2003, but was still not open prompting villagers to say they had received no benefit from the revenue sharing program. Incomplete projects mainly resulted from funds being too small for the planned project. However, even with substantial funds, as in one parish where US$7,300 was provided for a maize mill, machines were damaged during delivery, did not fit in the building constructed and required electrical power in a village with no electricity, prompting household respondents to comment that they would "never benefit from the maize mill" (21/7/09).

Local people felt that the sub-county leadership could not understand the needs of the frontline villages, "the district and sub-county leadership have never even seen an elephant" (Village Focus Group, 29/6/08) and so could not empathize with the community's need for defences against crop raiding. The decision to use US$6,500 of the revenue sharing funds for a sub-county headquarters so angered local residents, they wrote a petition to complain. UWA acknowledged that leaders "may divert the money to projects that may not reflect the needs of the communities" (UWA Official, 10/8/09) and that is why UWA are now attending village meetings making sure people understand the program. However, this has not yet removed the undercurrent of mistrust in higher levels of government; "Park money is handled by untrustworthy poor people who first have to fill their pockets before they serve the people" (Village Focus Group, 1/7/08). The legislation specifies that the project decision rests with the local government, however many, including UWA, "believe there is a need to provide UWA with some legal authority to ensure the funds are used efficiently and effectively for the intended purpose" (UWA Official, 10/8/09). Based on the current mistrust, it appears that a more active role for UWA or the district leadership is warranted.

UWA strongly believe they "need to solve the human-wildlife conflict and stop crop losses, so trenches are a priority" (UWA Official, 24/7/2008). However, district leadership believe the money should be spent on "a combination of local community needs and government priorities" (District chairperson, 12/6/2008). Although one district chairperson agreed that elephant trenches were the best projects to fund, other district chairpersons claimed that "building trenches to protect the people is the duty of UWA and that the trench construction should not be paid for by the revenue sharing program" (8/7/2008). Many people perceive that wild animals belong to the government and therefore should be controlled by UWA, just as local farmers are expected to control their livestock (Naughton-Treves, 1998). One district chairperson (5/6/2009) said that "using the revenue sharing money to fund trenches is as if I buy cattle then expect my neighbours to pay for putting up the fence that keeps them on my land". This opinion is reflected in the projects undertaken with only 32% of the revenue sharing money spent on trenches where trenches are seen as UWA's responsibility, while 81% of revenue funds went to trenches where the district chairperson supported using revenue funds for crop raiding defences.

Early revenue sharing disbursements went to schools, health clinics, water projects, and parish halls; however, this needed to change, as "old benefits were not benefiting front line farmers enough" (Sub-county chairperson, 13/6/2008). "Trenches create friendship between the national park … and the people but animals eating people's crops makes them poor" (Sub-county chairperson, 20/6/2008), so by addressing crop raiding they address development for local communities and "funding for schools and health clinics is easier to get from other sources" (Sub-county chairperson, 11/6/2008). Twenty of the 25 village chairpersons interviewed identified "trenches or any other methods that will stop crop raiding" (Village chairperson, 18/6/2008) as the preferred use for revenue sharing funds, adding as a second or third choice that they would like to see the money used for health units, schools, roads, water projects, or a community hall. However, use of funds for government facilities was not seen as acceptable; "There were sub-counties before the park existed and they had their headquarters paid for, why should UWA fund government buildings?" (Village chairperson, 27/6/2008).

The priority for village focus groups was also stopping crop raiding. Elephant trenches were seen not only as a defence but also as a means of generating income in the village; "Let the local people who face crop raiding be given a chance to excavate the trench" (Village Focus Group, 13/7/08). Other infrastructure projects were seen as a lower priority for most villagers, although villagers emphasized that "people pay taxes into the education fund and that is what should pay for schools" (Village Focus Group, 23/6/08).

Although the mandate for the revenue sharing program comes from national legislation, this knowledge deteriorated as one moved down the leadership ladder (Table 5-1), with some village chairpersons telling us that UWA and specifically KNP had decided to share the money with them or that local government or the people themselves had negotiated with the park to share revenues. The changing understanding of the mandate led to different interpretations of the program. District chairpersons saw revenue dispersal as a right, whereas village chairpersons saw the money as a favour from the park and as a result "they cannot destroy the park because they get benefit" (28/7/08).

### 5.3.3 Perceived benefit of revenue sharing

The mean perceived benefit tended to decrease as one moved from the authorities to the villagers (Table 5-2). In villages where revenue sharing projects had been implemented, the mean perceived benefit of poorer households was less than richer households, but the difference of means was not significant (Kruskal Wallis, $p=0.203$). Although generally lower, the village averaged household perceived benefit did correlate with the benefit of the revenue sharing program as perceived by the village chairperson ($r_{Spearman}=0.440$, $p=0.028$, $n=25$).

A correlation between the monetary allocation visible to each UWA warden and chairperson (Table 5-2), against their perceived benefit of the revenue sharing program, was significant ($r_{Spearman}=0.330$, $p=0.035$, $n=41$), indicating that people with greater visibility of program funds perceive more benefit. Dividing the village distribution by the number of households in a village equates to about US$10 per household over the last 10 years. As a result, the villagers often considered the benefit to be very little because "problems are much greater than benefits" (Village Focus Group, 16/7/08). The percent of households in a village with knowledge of a revenue

**Table 5-2: Perceived benefit of the revenue sharing program**

[1] Perceived Benefit Scale: 1=no benefit, 2=a little benefit, 3= some benefit, 4= reasonable benefit, 5= a lot of benefit

[2] An exchange rate of 1860 Ugandan Shillings to the US Dollar was used

[3] Although the disbursement is announced by UWA on the radio, most villagers are not aware of the cost of the projects.

| | | *Perceived Benefit of the Revenue Sharing Program*[1] | | | | |
|---|---|---|---|---|---|---|
| ***Group*** | ***N*** | ***Mean*** | ***Min*** | ***Max*** | ***Variance*** | ***Average Fund Visibility US$***[2] |
| *UWA Wardens* | 4 | 3.8 | 2.0 | 5.0 | 1.60 | $150,000 |
| *Local Government Chairpersons* | | | | | | |
| District | 3 | 3.2 | 2.5 | 4.0 | 0.58 | $37,097 |
| Sub-County | 9 | 3.3 | 2.0 | 5.0 | 1.32 | $11,290 |
| Village | | | | | | |
| All study villages | 25 | 2.6 | 1.0 | 4.0 | 1.15 | $585 |
| Only villages with a project | 16 | 2.8 | 1.0 | 4.0 | 1.10 | $914 |
| *Community Household* | | | | | | NA[3] |
| All study villages | 588 | 1.48 | 1.0 | 5.0 | 0.96 | |
| Only villages with a project | 353 | 1.5 | 1.0 | 5.0 | 0.99 | |
| Low wealth | 133 | 1.38 | 1.0 | 4.0 | 0.72 | |
| Medium wealth | 175 | 1.57 | 1.0 | 5.0 | 1.11 | |
| High wealth | 45 | 1.62 | 1.0 | 5.0 | 1.29 | |

sharing project also correlated with the monetary value of the projects done in or near each village ($r_{Pearson}$=0.448, p=0.025, n=25), and the village mean perception of the park as a neighbour was better as monetary value rose ($r_{Pearson}$=0.415, p=0.039, n=25). This increase in program visibility and attitude towards the park supports the procedures adopted to do fewer but more substantial projects.

Household respondents, aware that the project had been done by the revenue sharing program, had significantly higher perceived benefit of the program (Mann-Whitney U, p<0.001, n=588). Even though 88% of the interviewed chairpersons confirmed they held meetings to announce that a project had been funded by the park, only 40% of the household survey respondents were aware that projects had been funded by revenue sharing. This indicates that the information does not reach all village members or that the knowledge fades with time; "Happy about projects at the time that they happen, but crop raiding is continuous and this makes them angry" (Village Focus Group, 11/6/08). Some chairpersons were insistent that the projects had to happen directly in their village to benefit, and the village chairperson's perceived benefit marginally decreased the further from the village a revenue sharing project was located ($r_{Pearson}$=-0.396, p=0.050, n=25).

The highest mean household perceived benefit of the revenue sharing program was found in villages receiving income generating projects (maize mills and tourist camps), followed by medical clinic projects and elephant trenches (Table 5-3). The projects with the least perceived benefit were council facilities renovations, and tree planting. Household perceived benefit of revenue sharing did vary significantly by project (Kruskal-Wallis, $\chi^2$ (6 project groups)=53.23, p<0.001). Elephant trenches and medical clinics were the only projects considered significantly more beneficial than no project (both Mann-Whitney U, p<0.001), while elephant trenches were also perceived as more beneficial than schools (Mann-Whitney U, p<0.001), and council facilities (Mann-Whitney U, p<0.001). The preference for elephant trenches was emphasized by one of the village chairpersons; "Yes, the children go to the schools, but if going on an empty stomach because of crop raiding then they need trenches more" (1/7/08).

### 5.3.4 Influence on conservation

Even though the perceived benefit by villagers is low, UWA is convinced that revenue sharing is having a positive influence on conservation; "There has been an improvement in the attitudes of the local communities towards the parks, less illegal activity, improved community livelihoods and increased participation in park management where the program exists" (UWA Official, 10/8/09).

Local government and villagers alluded to the moral obligation that resulted from having received the revenue sharing benefit, while others wanted to ensure that revenue sharing benefits would continue to be received; "If a goat gives you milk you have to keep the goat well to keep getting milk, so if park benefits us we would conserve it to make sure the benefits still came" (Village Focus Group, 16/7/08). This sense of duty to the park was particularly strong in villages where elephant trenches had been built or were planned to be built in the coming dry season; "We need to work together with UWA, it is our duty to protect the park if UWA gives us money for the trench" (Village Focus Group, 29/6/08).

When asked in the focus groups how conservation behaviours had improved as a result of the revenue sharing program, villagers explained that "they do not kill animals" (28/7/08), and "they make sure that they don't start fires and report poachers or people cutting trees" (10/7/08). However, a strong theme emerged from focus group discussions that "only if the animals stay in the park can they become good conservationists" (23/6/08). This again highlighted that elephant trenches would be valued by the villagers and would have the greatest influence on conservation outcomes; "We would love the park better if we had trenches. Building schools and facilitating other programs other than the trench does not change the conservation attitudes of the people" (Village Focus Group, 1/7/08).

Some chairpersons felt that the benefit was "not enough because crop raiding losses far exceed the gains from the park" (10/6/08). They thought the park owed the people and therefore people took from the park; "Some hate the park because crop raiding held them back from exams and graduating school" (2/7/08), "people have already lost a lot to the animals, so locals go to the park to get meat" (8/6/09). However, this resentment was not widespread, as 69% of households ranked the park as a good or very good neighbour.

The measured illegal extraction of resources adjacent to each of the study villages was used to test if the revenue sharing program was effectively contributing to improved conservation in the park. The village-mean household perceived benefit of revenue sharing, village chairperson perceived benefit of the program, levels of illegal tree extraction, illegal entry trails, number of domestic animals seen grazing in the park, number of poaching signs found and the combined human disturbance index are summarized in Table 5-4. The village average household perceived benefit had too little variance to correlate with any of the illegal extraction measures. However, the perceived benefit of the revenue

**Table 5-3: Household perceived benefit of revenue sharing**

[1] Perceived Benefit Scale: 1=no benefit, 2=a little benefit, 3= some benefit, 4= reasonable benefit, 5= a lot of benefit

| | | *Perceived Benefit of Revenue Sharing*[1] | | |
|---|---|---|---|---|
| ***Project*** | ***N*** | ***Mean*** | ***Min*** | ***Max*** |
| Maize Mill | 24 | 1.79 | 1 | 4 |
| Tourist Camps | 24 | 1.79 | 1 | 4 |
| Medical Clinic | 46 | 1.74 | 1 | 5 |
| Elephant Trench | 141 | 1.68 | 1 | 5 |
| School | 187 | 1.57 | 1 | 5 |
| Roads & Bridges | 70 | 1.56 | 1 | 5 |
| Water Projects | 48 | 1.40 | 1 | 4 |
| Council Facilities | 94 | 1.23 | 1 | 4 |
| Tree Planting | 24 | 1.17 | 1 | 4 |

sharing program by the village chairperson significantly correlated with the number of animals grazing in the park ($r_{Spearman}$=-0.488, p=0.013, n=25), the number of poaching signs found ($r_{Spearman}$=-0.406, p=0.044, n=25) and the human disturbance index ($r_{Spearman}$=-0.459, p=0.021, n=25), indicating that when the chairperson perceived higher benefit the human disturbance in the park adjacent to their village was less.

The only two projects that had been implemented in more than six villages and could therefore be tested for equality of means were schools and elephant trenches. However only the elephant trench realized a significantly higher village-averaged household perceived benefit (t-test, p=0.002), higher village chairperson perceived benefit (t-test, p=0.028), and a lower human disturbance index (t-test, p=0.016). In fact, no signs of poaching or livestock grazing were found along park boundaries where trenches had been built. However, it is recognized that trenches may not only be a physical barrier to the elephants raiding crops but also a barrier to humans and livestock; "We never get resources from the park because we can no longer cross the trench" (Household survey comment, 7/7/09). So, if the revenue sharing program aims to improve conservation behaviours and benefit local people, the building of crop raiding defences in the form of elephant trenches appears to be the most effective allocation of the money.

## 5.4 Discussion

In many areas of the world, the primary critique of revenue sharing programs is that the distribution of community projects is either too sparse or too homogeneous to account for the unequal distribution of household losses incurred due to the existence of protected areas (Archibald & Naughton-Treves, 2001; Adams & Infield, 2003; Spiteri & Nepal, 2006). The revenue sharing projects around KNP have been spatially distributed close to the park with a third of projects located within the area of highest incurred losses due to crop raiding (Naughton-Treves, 1998). The significantly higher perceived benefit in villages where defences have been built to deal with crop raiding losses supports the conclusion that revenue sharing needs to be targeted towards those who bear the greatest cost of conservation.

Prior studies have also concluded that community incentive benefits are too small to improve conservation attitudes or reduce illegal activities (Archibald & Naughton-Treves, 2001; Kaltenborn *et al*., 2008; Spiteri & Nepal, 2008). For the revenue sharing program around KNP, UWA and local leadership believed the program benefits local people and improves conservation attitudes, a finding consistent with Archibald and Naughton-Treves (2001), and reduced human disturbance was recorded where the village chairperson perceived higher benefit from the program.

However, all of the focus groups indicated that losses incurred by living next to the park far outweighed the benefit of the revenue sharing program resulting in very low household perceived benefit, a finding consistent with studies around other protected areas in East Africa (Mugisha, 2002; Kaltenborn *et al*., 2008; Schroeder, 2008). Since, the decision to poach or take wood illegally from the park is made within the individual household, the lack of correlation between human disturbance and benefit perceived by the household suggests that most

**Table 5-4: Village-scale perceived benefit and illegal resource extraction**

[1] Perceived Benefit Scale: 1=no benefit, 2=a little benefit, 3= some benefit, 4= reasonable benefit, 5= a lot of benefit

| | *N* | *Mean* | *Min* | *Max* |
|---|---|---|---|---|
| Village-mean household perceived benefit[1] of revenue sharing | 25 | 1.48 | 1.00 | 2.87 |
| Village Chairperson's perceived benefit[1] of revenue sharing | 25 | 2.62 | 1.00 | 4.00 |
| Number of trees illegally extracted per km of boundary | 25 | 143.6 | 10 | 614 |
| Number of illegal entry trails per km of boundary | 25 | 3.8 | 0 | 27 |
| Number of domestic animals seen grazing in the park per village | 25 | 19 | 0 | 200 |
| Number of poaching signs found per village | 25 | 1.2 | 0 | 14 |
| Village Human Disturbance Index | 25 | 0 | -2.32 | 5.07 |

community benefits may not be sufficiently influencing the conservation behaviours of the individual. However, since the chairperson is the arbiter of discipline in the local villages, the level of perceived benefit by the chairperson may be translating, at least partially, into the actions of the people. Additional research asking villagers what causes or deters them from entering the park would confirm this. However if true, the chairperson could be instrumental in shaping the conservation attitudes and behaviours within their village and could become a conservation advocate for UWA.

Inclusion of local communities in the project decision process is critical to conservation effectiveness (Agrawal & Gibson, 1999; Honey, 1999; Spiteri & Nepal, 2006; Kaltenborn *et al*., 2008). The most important finding of this research is that the type of project chosen for implementation with the revenue sharing funds does matter. Generally, the villagers perceived that schools and health clinics were the responsibility of the government and that using the revenue sharing funds to pay for them meant they lost more localized benefits. The only type of project that was preferentially desired by villagers, showed significantly higher perceived benefit of the program by the household and the village chairperson, as well as lower levels of human disturbance inside the park was the building of elephant trenches to reduce crop raiding. It is therefore recommended that fund distribution be allocated based on local community preferences and for the communities around KNP this means the funds should be spent on the building and maintenance of elephant trenches or other crop raiding defence projects.

Based on the correlation of perceived benefit and the amount of money locally distributed, increasing the revenue sharing distributions should improve the perceived benefit of the program. However this increase is impractical to expect while UWA's operating budget is still subsidized by the Ugandan government and international donors. In lieu of increased funds, perceptions of the revenue sharing program could be improved through better accounting transparency, project implementation and oversight. Data on fund accumulation, distribution and accounting of the funds as they distribute through the district and sub-county governments should be shown to village chairpersons and the villagers.

Weak institutions to manage incentive based compensation schemes have been reported to lead to mistrust of conservation authorities (Archibald & Naughton-Treves, 2001; Spiteri & Nepal, 2006; Kaltenborn *et al*., 2008) and in Kenya to down-sizing of the revenue sharing program (Honey, 1999). Around KNP, significant project management issues were encountered in 30% of the projects implemented to date, including incomplete projects, missing materials and poor contractor oversight. Program management training for sub-county leadership, inclusion of the village chairpersons in the implementation process and the use of local village labour to legally by-pass the tendering process could improve the program implementation in the eyes of frontline villagers.

Finally, the intent of the revenue sharing program is to forge better relationships between UWA, local governments and local people, while improving conservation attitudes and behaviours. The distribution of the funds is improving relationships, but UWA lacks the mandate to ensure that project implementation is directly linked to conservation. A change in the revenue sharing legislation to provide either UWA or the district leadership with greater oversight of fund allocation and project implementation is recommended.

Based on the finding that the perceived benefit of the revenue sharing program increases with the amount of money distributed for a given project, the practice of rotating the benefit between parishes to retain sufficient funds to provide tangible benefits to the local communities should be continued. Trying to gain consensus with villagers and village chairpersons about the projects chosen and choosing projects that provide crop raiding defences are project management practices that should be continued because they ensure the needs of local people are included in the project decision making process, resulting in more positive perceptions of the program by local communities.

## 5.5 Conclusions

The implementation of the Ugandan revenue sharing program has evolved over the decade since its inception. UWA and local leadership have contributed to this evolution demonstrating a desire to progressively improve the structure, implementation process, and effectiveness of the program to address the needs of frontline villagers. This study has demonstrated that the revenue sharing program around KNP does provide real

benefit to the local people and improve conservation behaviours if the projects specifically deal with the villagers' primary problem of crop raiding by park protected animals.

The results of this study can guide the implementation of revenue sharing programs in other locations, because the lessons learned speak to general management practices. Most importantly, revenue sharing programs need to ensure community involvement in the project decision making process, so they can focus the benefits to best deal with the losses incurred as a result of living next to the protected area. If possible, the benefit distributed should balance the losses incurred and the monetary collection and distribution process needs to be accountable and transparent to community members. If funds are small, don't try to give a small benefit to many communities. Focus the funds on a substantial project in one community, choosing a different community to benefit from each distribution. Although corruption within the distribution process is often identified as an issue with these financially based programs, the results of this study indicate that what might be perceived as corruption, was more often than not, just poor project management. Therefore, those responsible for project implementation should be trained to properly scope and manage the projects being undertaken. Attention to these management details will produce a more effective revenue sharing program.

# 6 Elephants in the Garden: Financial and Social Costs of Crop Raiding

Seventy-three percent of survey respondents reported that their household experienced crop raiding by park-protected animals, and in the focus groups, participants claimed that many side-effects of crop raiding were detrimental to human development in villages next to KNP. In this chapter, I describe the extent of crop raiding around KNP, and test the claims recorded in the focus groups to quantify the financial and social costs of crop raiding. This chapter in a revised form, co-authored by Peter Ahabyona, was published in *Ecological Economics* 75, 72-82 following the final submission of this dissertation.

**Summary:** Residents near protected areas disproportionately bear the costs of conservation, in part due to crop raiding by protected animals. These costs increase as conservation efforts lead to the recovery of animal populations, and human population growth increases the proportion of land outside the parks used for agriculture. Financial and social costs associated with crop raiding were studied in 25 villages around Kibale National Park, Uganda. Perceptions about crop raiding were collected using focus groups and household surveys, while damage was evaluated based on physical monitoring of crop raiding incidents over six months. The average financial loss for farmers around the park was US$74 (1.5% of median household capital asset worth) and damage was particularly high within 0.5 km of the park boundary. Households experiencing crop raiding were more prone to food insecurity, and higher rates of self-reported human and livestock diseases, while children from villages bordering the park tended to have poorer scholastic achievement than children from other villages. Compensation is not affordable for the wildlife authority, nor is it sustainable as crop raiding is escalating. To mitigate costs for local communities, funding is needed to develop defences, and development activities need to be preferentially delivered closer to the park.

## 6.1 Introduction

When people live next to protected areas (PAs) there will be conflicts between people and wildlife. Local people argue that wild animals are trespassing on their lands (Sifuna, 2005; Laudati, 2010) while conservationists highlight that ever-growing human population encroaches upon wild animal habitat and migration routes (Thouless & Sakwa, 1995; Sitati *et al*., 2003). Close to PAs, crop damage caused by park protected animals is a significant risk for farmers. Studies in Tanzania and Uganda found 86-88% of farmers living adjacent to national parks reported crop raiding (Newmark *et al*., 1994; Naughton-Treves, 1997; Weber *et al*., 2007). As a result, local communities can retaliate, killing animals or aiding the efforts of poachers (Nyhus *et al*., 2005; Sifuna, 2005). Human-wildlife conflict creates barriers to cooperation between communities and conservation authorities (Tweheyo *et al*., 2005; Treves *et al*., 2006), while deteriorating support for conservation (Tchamba, 1996; Naughton-Treves, 1998; Gillingham & Lee, 1999; Gadd, 2005; Chelliah *et al*., 2010).

Near PAs the traditional crop defence of killing raiding animals is restricted by conservation policy (Naughton-Treves, 1997; Weber *et al*., 2007; Laudati, 2010). Farmers in Africa are typically not compensated for losses caused by protected animals (Naughton-Treves, 1997), however, some countries, such as Botswana, do have partial compensations programs (Jackson *et al*., 2008). Although smaller losses may be tolerated, the loss of an entire season's crops in one night can be devastating (Naughton-Treves, 1998). This economic injustice at the boundary of PAs has lead to calls for compensation programs (Tchamba, 1996; Naughton-Treves, 1998; Laudati, 2010); but most conservation authorities lack the revenues to finance such programs without external support (Tchamba, 1996). Others believe compensation simply will not work (Bulte & Rondeau, 2005; Warren *et al*., 2007), and recommend funding be directed towards the development of crop raiding defences to mitigate losses (Thouless & Sakwa, 1995; Sitati & Walpole, 2006; Warren *et al*., 2007).

Farmers do try to protect their crops. Human guarding, chilli rope, trenches, and dogs are effective against crop raiders (Newmark *et al*., 1994; Sitati *et al*., 2005; Sitati & Walpole, 2006), while non-electrified fences and live traps are generally ineffective (Sitati *et al*., 2005; Weber *et al*., 2007). Passive solutions like fences and trenches need to be maintained to remain effective, as raiding animals learn to destroy or move around them

(Thouless & Sakwa, 1995; Osborn & Parker, 2003). Guarding is labour intensive, often restricting the household from participating in income generating activities and leads to other social costs (Naughton-Treves, 1998; Hill, 2000; Osborn & Parker, 2003; Hill, 2004). All household members guard crops, including children who are held out of school (Haule *et al*., 2002; Kagoro-Rugunda, 2004). Therefore crop raiding has negative implications for development in communities adjacent to PAs, resulting in poor childhood education (Haule *et al*., 2002), food insecurity (Hill, 1997), and physical injury from raiding animals (Sifuna, 2005).

The research reported here describes and values the crop raiding losses incurred by farmers around Kibale National Park in Uganda. We then performed a preliminary assessment of the perceived social costs of crop raiding, as reported in village focus groups, by testing the perceived costs against household survey and scholastic achievement data to determine whether the social costs were generally realized around the park.

## 6.2 Methods

### 6.2.1 Study site

Kibale National Park (KNP), located in southwest Uganda (Fig.6-1), is a 795km$^2$ mixed forest and savannah protected area that provides habitat for 13 primate species (Chapman & Lambert, 2000), of which baboon (*Papio cynocephalus*), redtail monkey (*Cercopithecus ascanius*), vervet monkey (*Cercopithecus aethiops*), black and white colobus (*Colobus guereza*), L'hoesti monkey (*Cercopithecus lhoesti*), and chimpanzees (*Pan troglodytes*) have all been documented to raid crops adjacent to the park (Naughton-Treves, 1998). Other identified crop raiding species in KNP (Naughton-Treves, 1998; Chiyo & Cochrane, 2005) include elephants (*Loxodonta africana*), bushpigs (*Potamochoerus procus*), civets (*Civettictis civetta*), porcupines (*Hystrix africae-australis*), duiker (*Cephalophus* spp.) and bushbuck (*Tragelaphus scriptus*).

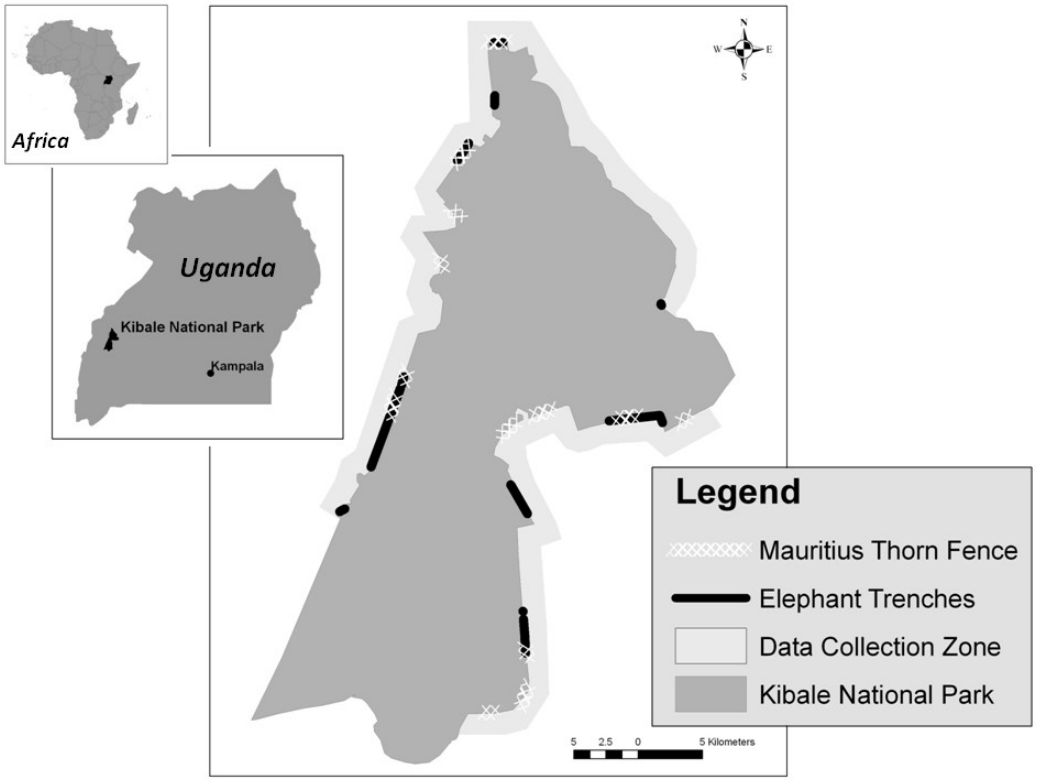



**Figure 6-1: Kibale National Park and passive crop raiding defences**

In 1992 a 23 month crop raiding study covering six villages and 93 households within 500 m of the park boundary (Naughton-Treves, 1998) found the most frequent wild animal raiding species were redtail monkey and baboon, with primates in general responsible for 71% of crop raiding events and 48% of the damage. More total area was damaged by baboons than elephants, but elephants caused more damage per event. In 1999 a short follow-up study in three of the same villages found redtail monkeys and baboons remained the most frequent raiders, but baboons had usurped the red-tail monkey as the species that damaged the greatest number of farms (Naughton-Treves & Treves, 2005). Proximity to the forest was the strongest predictor of damage (Naughton-Treves & Treves, 2005), with 90% of damage located within 160 m (1992-94) and 200 m (1999) of the park boundary respectively. Farmers lost on average 4-7% of their crops in 1992-94 and 6-9.4% of their crops in 1999 (Naughton-Treves & Treves, 2005).

As per local tradition, neighbours provide compensation if livestock damage crops; however, the Ugandan government does not compensate for wild animal crop raiding (Naughton-Treves, 1997). Since people consider the animals in the park to belong to the government, they consider the park to be a bad neighbour (Naughton-Treves, 1999). However the Ugandan Wildlife Act (Uganda Wildlife Statute, 1996) does provide for 20% of park entrance fees to be shared with local communities for development projects through a revenue sharing program (Archibald & Naughton-Treves, 2001). Most communities want this money to be directed toward providing crop raiding defences (Chapter 5). Some of this money has been used to construct elephant trenches (3 m deep by 2-3 m wide), along portions of the boundary (Fig. 6-1). These trenches can deter elephants from crop raiding, as long as the trenches are continuous, extend well beyond the length of a village and are well maintained. Additional trenches have been funded by the Uganda Wildlife Authority (UWA), the FACE the Future Foundation (FACE), and the International Union for Conservation of Nature (IUCN). IUCN also funded the planting of Mauritius thorn (*Caesalpinia decapetala*) to deter crop raiding animals from exiting the park (Chhetri *et al*., 2004).

### 6.2.2 Data collection

A combination of interview based surveys and physical verification of damage by trained field assistants were used to collect data on crop raiding damage, and focus groups were used to learn about the perceived social problems caused by crop raiding. Combining these methods is the most common approach to crop raiding investigation in Africa (Tchamba, 1996; Hill, 1997; Naughton-Treves, 1998; Kagoro-Rugunda, 2004; Sitati & Walpole, 2006). Damage caused by livestock was recorded in both the household survey (4.1% of estimated crop production lost) and the physical verification study (6.2% of the total area damaged), because prior studies in Africa (Naughton-Treves, 1998; Warren *et al*., 2007; Weber *et al*., 2007) found crop damage due to livestock accounted for 11-36% of lost crop yield. Damage caused by livestock was removed from the total damage and all raiding data reported in this paper includes only damage caused by wild animals.

Local people live in villages of about 100 households, governed by a village council, which is led by a village chairperson. For this study, a village is defined by the spatial extent of households associated with a village name under the leadership of one village chairperson. Twenty-five study villages were chosen, located approximately five kilometres apart in the data collection zone (Fig. 6-1), based on village members holding and/or cultivating land directly adjacent to the park.

**Household survey:** All village households were Global Positioning System located while noting the construction standard of primary dwellings. Households were located between 15 and 3,300 meters from the park boundary. The number of households per village ranged from 41 to 242. Within each village 24 households were surveyed (10% to 59% of the village households) representing 24% of households in all study villages. The households were chosen by random stratified sampling, with stratification based on the house construction standard (Ellis & Bahiigwa, 2003), a proxy for household wealth. The survey was administered in July and August 2009 by four Ugandan field assistants, three men and one woman, in two local languages (Rutooro and Rukiga).

During the survey, respondents were asked whether or not their household experienced crop raiding, how troubled they perceived their household to be by this raiding and to estimate the fraction of five staple crops (maize (*Zea mays*), matoke (*Musa* spp), cassava (*Manihot esculenta*), sweet potato (*Ipomea batatas* L) and yams (*Dioscorea* spp)) that had been lost in the last growing

season and how much of that loss was due to livestock. Matoke is a starchy, green cooking banana similar to plantain. Beans (*Phaseolus vulgaris* L) were not included in the survey as a prior study near Budongo Forest Reserve in Uganda found beans were not as vulnerable to crop raiding (Hill, 1997). Using picture cards of both wild and domestic animals, they were asked to identify which species raided their crops and to rank the animals by frequency of raids and damage caused (Hill, 1997). They were then asked to identify months of the year that were worse for crop raiding and whether they used any crop raiding defences or planting strategies to reduce crop raiding. If they identified human guarding as a defence method, they were asked how many days per week men, women and children spent guarding in both high and low crop raiding seasons. Crop raiding tends to be worse at the end of the dry seasons (Naughton-Treves *et al*., 1998), when the crops are harvested (May-July and November-January).

Two survey variables were used for statistical analysis: 1) the perceived trouble to the household as a result of crop raiding measured on a 5 point scale (1=none, 2=a little, 3=some, 4=considerable, 5=a lot), and 2) the estimated fraction of crop production lost, calculated as the mean estimated loss of the five staple crops, with the fraction of loss attributed to domestic animals removed. Neither of these two variables was normally distributed, so non-parametric statistics were used.

Demographic and socio-economic data was also collected in the survey. Thirty-three percent of households relied solely on subsistence agriculture for their livelihood, while 67% were engaged in other income generating activities such as making honey, charcoal, bricks, and gin, and cash crop production. Thirty percent of households had members employed outside the household. Village population densities ranged from 70 to 611 people/km$^2$.

**Six month study:** The frequency of visits by field assistants to verify damage was identified in the literature as critical, because greater than two weeks between visits allowed for evidence to be destroyed through replanting and further crop raiding (Hill, 2000). Due to the logistic challenge of reaching all 25 villages every one to two weeks, a local assistant was trained in each village (Naughton-Treves, 1997; Sitati *et al*., 2005) to physically verify damage, identify which crops were damaged, measure the damaged area, and count the number of damaged plants. Using a standardized identification guide, the assistant also attempted to identify which species of animal caused the damage based on tracks, scat and bit marks (Naughton-Treves, 1998; Kagoro-Rugunda, 2004). Physical verification by the local assistant continued weekly for six months (August 2009 to January 2010), capturing parts of both dry seasons and a wet season. For quality assurance/control, Peter visited each village at least once per month to audit, review, and collect data. Unfortunately, data from one village had to be discarded due to apparent data fabrication.

Six households participated in the physical verification in each study village, including households along the border of the park and households along a village track travelling away from the park boundary (up to 1314m). This allowed for variations in damage along the boundary and away from the boundary into the village to check if shielding by other households or distance from the park reduced crop losses (Hill, 1997; Naughton-Treves, 1998). Ninety of the physical verification households had also participated in the survey, allowing survey data to be linked to the six month study.

The statistical analysis was based on three variables: 1) the number of wild animal crop raiding incidents experienced by the household, 2) the area damaged, and 3) the valued financial loss incurred. A crop raiding incident was defined as a single foray by a single crop raiding species to a single household. Six month study variables were not normally distributed so non-parametric statistics were used.

Financial losses were calculated as the value of the crop/m$^2$ multiplied by the area of crop damaged (Naughton-Treves & Treves, 2005). Crop value was equal to yield multiplied by local market price (Kagoro-Rugunda, 2004). The yield of each crop was based on the average of farmer estimates from at least six locations around the park to allow for variation in geographic location or farming methods. The average calculated yield was checked against national yield statistics (European Commission, 2010), and was typically within 10%. The national yield was used for rice (*Oryza sativa*) and Artemisia (*Artemisia* L) as these crops are grown in very limited areas around the park. Local market prices, collected early in 2009, were derived from six to 18 trading centres around the park, depending on how widely the crop was cultivated. Loss value was converted

from Ugandan shillings to US dollars using an exchange rate of 2250 Ugandan shillings/US Dollar.

**Focus groups:** To understand the perceived problems of crop raiding, focus groups were conducted in 15 study villages (60%) in June and July 2008. The village chairperson, as per cultural norms, approved and organized the meetings, inviting men and women from a range of age groups representing a cross-section of the village. Since the meetings were held in the open, the size of focus groups often grew beyond those invited, resulting in the total number of people present ranging from 16 to 51. Women, the primary cultivators of food crops (Naughton-Treves, 1997), were in attendance at 14 of the focus groups, where they represented 6 to 65% of the participants. Discussion was facilitated by Peter in the local languages, translating the responses verbally during the meeting, while another assistant recorded all villager comments, allowing triangulation of notes for a more complete record of the meeting.

Comments associated with crop raiding were coded based on whether the problem related to direct financial loss, opportunity costs, food insecurity, education, health, or fear of physical injury. These six themes formed the basis to understand if socio-economic implications of crop raiding were systemic problems or isolated issues being exaggerated for the meeting. Group discussions about crop raiding have tended to result in more complaints and over estimation of damage relative to physical studies (Naughton-Treves & Treves, 2005). The socio-economic implications of crop raiding raised by focus groups were statistically tested against the household survey using Chi-squared analysis, comparison of means and correlation testing to check for statistically significant relationships that might warrant further investigation.

**Supplementary scholastic achievement data:** Concerns about children missing school to guard crops and the inability to pay school fees due to crop raiding were raised in 60% of the focus groups. Since children's scholastic achievement had not been collected in the household survey, a supplementary study was conducted with 12 primary schools located adjacent to study villages from October to December 2009. The catchment area of these schools included the study village, adjacent to the park, as well as other villages farther from the park. In each of the school years, P1 to P7, the average achieved grade for students from the study village was compared with the average achieved grade for all other students. This comparison was done separately for boys and girls. Pairing data by school year and gender permitted the scholastic achievement of students from the study villages to be directly compared with their peers without having to separately control for age, gender, or quality of school. The difference in grades achieved between village students and their peers for all years and both genders were averaged for each primary school and compared with the mean perceived trouble of crop raiding in the adjacent study village.

## 6.3 Results

### 6.3.1 The crop raiding landscape

In all focus groups, crop raiding was identified as the biggest problem of living next to KNP and 73% of survey respondents said their households experienced crop raiding. Respondents felt crop raiding troubled their household considerably and on average reported having lost 30% of their production of the five staple crops recorded. The perceived burden of crop raiding and the estimated fraction of production lost from the survey were strongly correlated ($r_{Spearman}$=0.621, p<0.001, n=591) and both varied around the park boundary (Fig. 6-2a&b). During the six month physical verification study, 77% of households were raided, on average experiencing eight raids causing 1,638 $m^2$ of damage, valued at US$74, and like the survey data, this varied around the park boundary (Fig. 6-2c&d). The survey and six month study were compared in the overlapping 90 households, finding that on average 20% of the area cultivated by the household had been damaged during the six month study, suggesting the 30% loss of production crops estimated by the survey respondents was overstated. The number of baboon raids recorded in the six month study and the perceived trouble of crop raiding correlated ($r_{Spearman}$=0.217, p=0.040, n=90), indicating frequent baboon raiding may be shaping the opinions of local farmers.

Crops were primarily raided by baboons and elephants, but redtail, vervet, and colobus monkeys (not distinguished by subspecies) also raided crops, as did bush pigs, chimpanzees, and civets (Table 6-1). Spatial distribution of raiding incidents by species varied around the park (Fig. 6-3). Baboon raiding was prevalent around much of the park, while elephant and small primate raiding occured less along the southern boundaries. Bush pig raiding was localized to the south and western

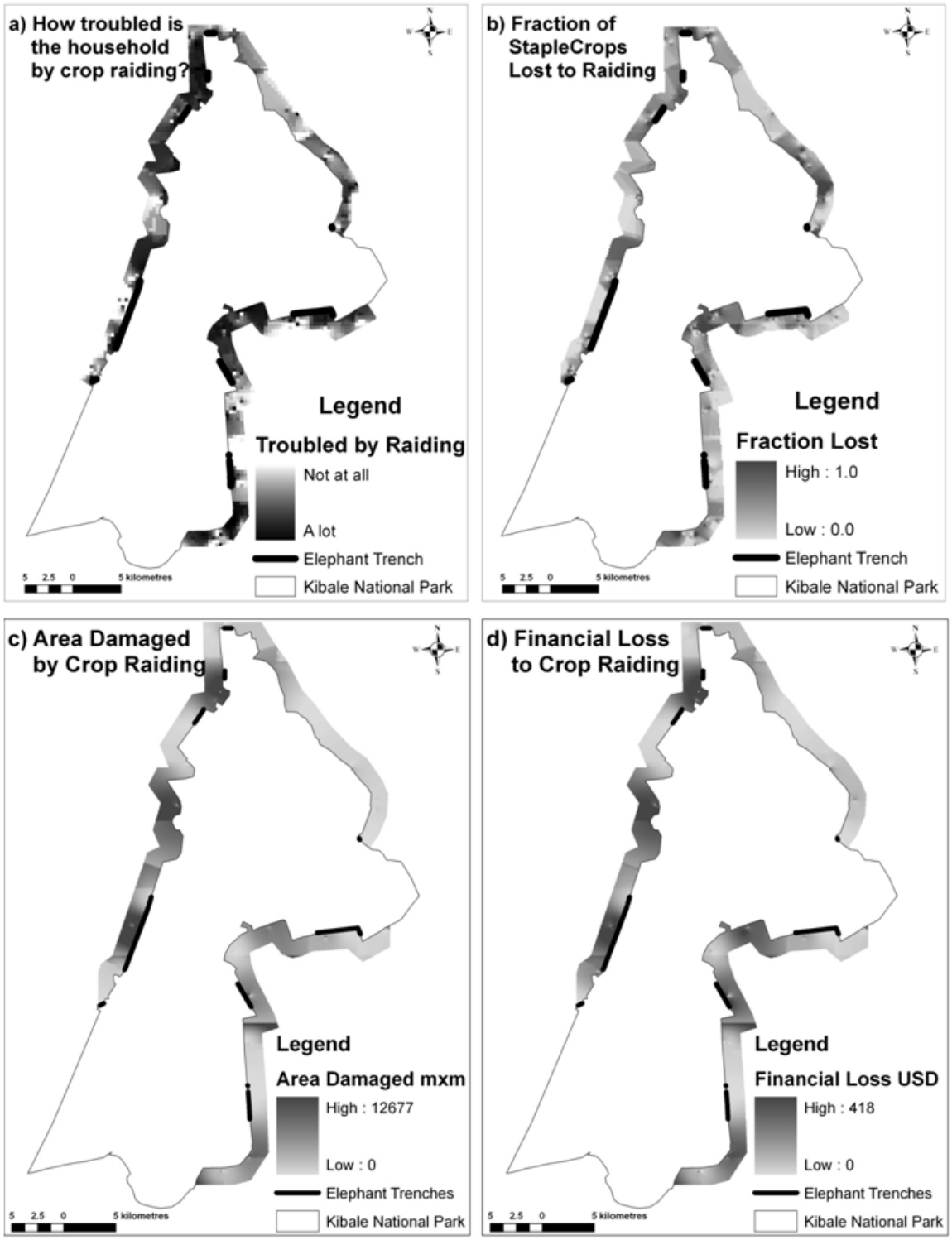



**Figure 6-2: Spatial distribution of crop raiding around Kibale National Park**

(a) Perceived trouble, (b) Fraction of production lost, (c) Area damaged, (d) Financial Loss

(a) & (b) Survey data n=596, (c) & (d) six month physical verification data n=143

**Table 6-1: Crop raiding damage by species around Kibale National Park, Uganda and its estimated financial cost**

| | Household Survey | | | Six Month Physical Validation Study | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Percent | Frequency | Damage | Percent | Incidents | | | Area damaged ($m^2$) | | | Financial loss (USD) | | |
| Raiding Species | reporting | Rank | Rank | raided | Total | Mean | Max | Total | Mean | Max | Total | Mean | Max |
| Baboon (*Papio cynocephalus*) | 85% | 1 | 2 | 60% | 470 | 3.3 | 25 | 78714 | 550 | 8996 | $3,809.20 | $26.24 | $312.67 |
| Elephant (*Loxodonta africana*) | 79% | 2 | 1 | 50% | 190 | 1.3 | 12 | 87183 | 610 | 23396 | $3,515.21 | $24.58 | $603.25 |
| Redtail Monkey (*Cercopithecus ascanius*) | 69% | 3 | 3 | 21% | 77 | 0.5 | 10 | 3052 | 21 | 722 | $163.49 | $1.14 | $84.64 |
| Vervet Monkey (*Cercopithecus aethiops*) | 42% | 5 | 5 | 14% | 45 | 0.3 | 8 | 3008 | 21 | 894 | $175.93 | $1.23 | $32.77 |
| Bushpig (*Potamochoerus procus*) | 42% | 4 | 4 | 9% | 53 | 0.4 | 10 | 4153 | 29 | 2238 | $250.91 | $1.75 | $97.61 |
| Colobus Monkey (*Colobus* spp) | 10% | 6 | 7 | 9% | 21 | 0.1 | 3 | 151 | 1 | 62 | $70.26 | $0.49 | $34.19 |
| Chimpanzee (*Pan troglodytes*) | 15% | 7 | 6 | 7% | 14 | 0.1 | 3 | 1102 | 8 | 338 | $134.25 | $0.94 | $88.99 |
| African Civet (*Civettictis civetta*) | 0% | NR | NR | 2% | 8 | 0.1 | 3 | 18 | 0.1 | 7 | $0.53 | $0.00 | $0.21 |

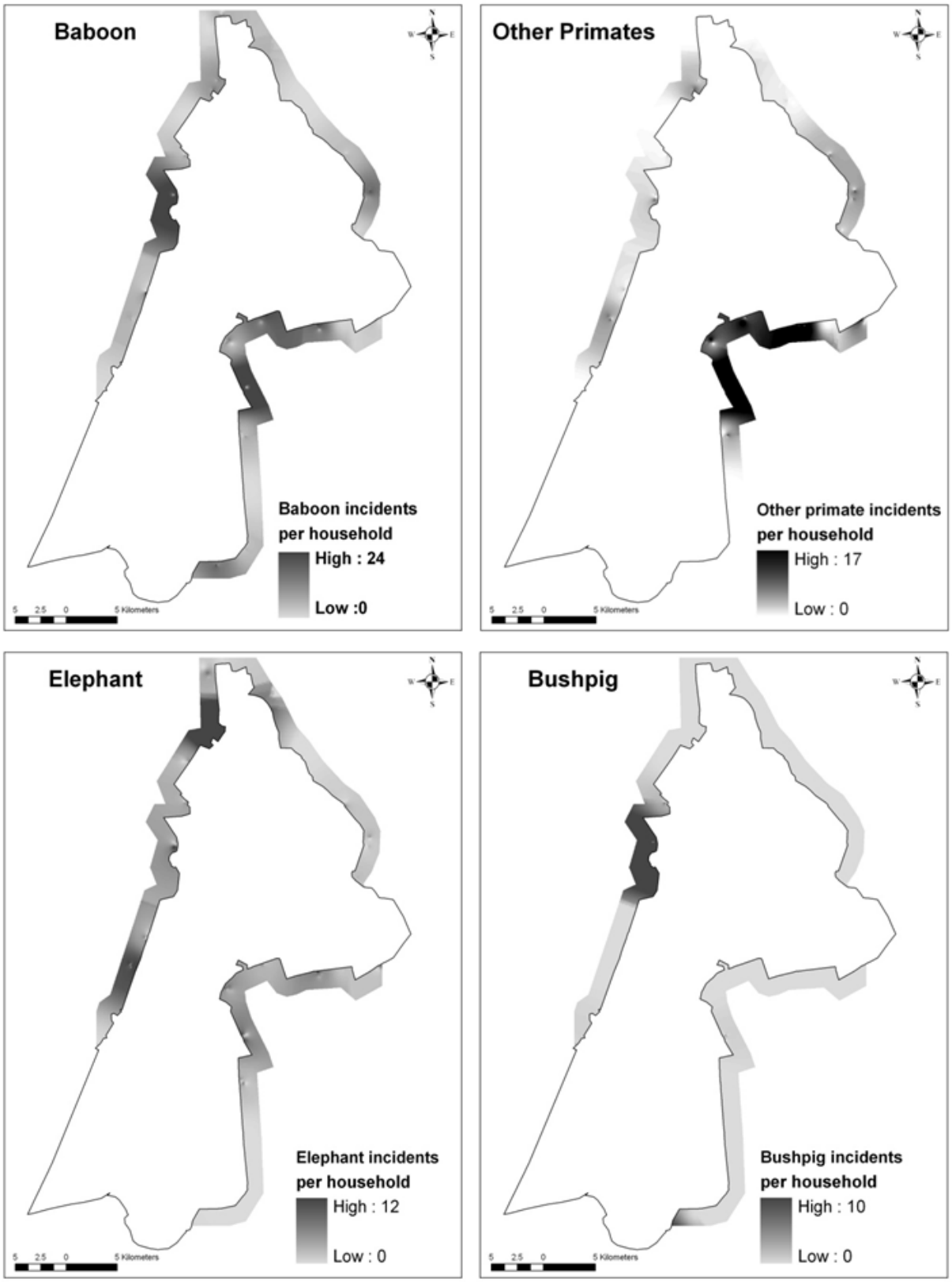



**Figure 6-3: Spatial distribution of crop raiding by species around Kibale National park, Uganda**

borders. Chimpanzee raids were recorded in villages close to chimpanzee troops that had been habituated for research and tourism. The most frequent crop raider was the baboon, followed by elephant and small primates (Table 6-1). Elephant raiding caused more area to be damaged overall and per household; however, due to the frequency of baboon raids on higher value crops, the financial loss was highest for baboons.

Two annual crop raiding cycles were identified by survey respondents (Fig.6-4), with peak crop raiding months occurring just before or during harvest times. The six month study covered one of these temporal cycles and confirmed a peek in November when crops are ready for harvesting, with a steep decline after crops were harvested. Baboon raiding was consistent with this temporal pattern; however, elephant raiding steadily declined from the beginning of the six-month study.

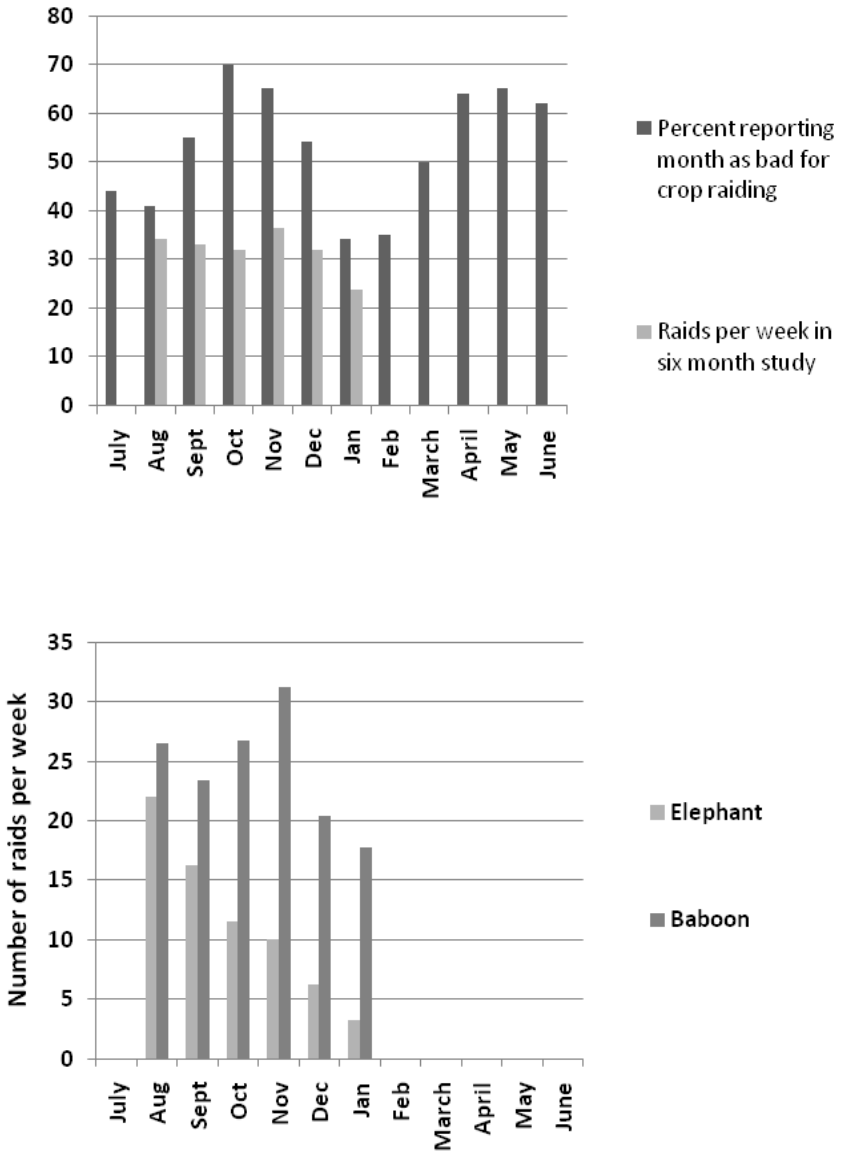



**Figure 6-4: Temporal variation in crop raiding**

### 6.3.2 Barriers to crop raiding

The location around the boundary of the park influenced how much a household would be troubled by certain species (Fig. 6-2 & 6-3). Villages experiencing more elephant raiding had significantly more area damaged ($r_{Spearman}$=0.493, p=0.014, n=24). The percent of households reporting crop raiding increased closer to the park (Fig. 6-5), with 90% of damage occurring within 530m of the park boundary. Closer to the park, households reported a higher fraction of crop production lost ($r_{Spearman}$=-0.413, p<0.001, n=596) and perceived that they were more troubled by crop raiding ($r_{Spearman}$=-0.377, p<0.001, n=596). The verified area damaged by crop raiding animals also increased closer to the park boundary ($r_{Spearman}$=-0.282, p=0.001, n=143).

Households with more than two houses between them and the park boundary experienced significantly fewer raiding incidents (Mann-Whitney U, p=0.011), had less area damaged (Mann-Whitney U, p=0.006) and incurred lower financial losses (Mann-Whitney U, p=0.024). However, overall village population density did not correlate with crop raiding levels ($r_{Pearson}$=0.022, p=0.920, n=24). As per Batooro custom, immigrants to the village, typically Bakiga, are allocated land closer to the park (Mann-Whitney U, p=0.001), creating a crop raiding buffer for Batooro farmers (Aluma et al., 1989; Naughton-Treves & Treves, 2005). Therefore Bakiga households lost more to crop raiding than their Batooro neighbours (Mann-Whitney U, p=0.023).

Twenty-four different crops were raided by wild animals during the six month study. The most frequently raided crops were maize (39%), cassava (14%), beans (10%), matoke (10%), and sweet potato (8%). These crops are grown by a majority of households (Table 6-2). To mitigate crop raiding losses, some households employed crop planting strategies; planting closer to the house (41%), planting as far from the park as possible (27%), stopped planting crops animals prefer (17%), planting crops they knew animals did not like (10%), planting tea as a buffer crop (not palatable to animals, 9%), while 10% of households reported abandoning land near the park as a result of crop raiding.

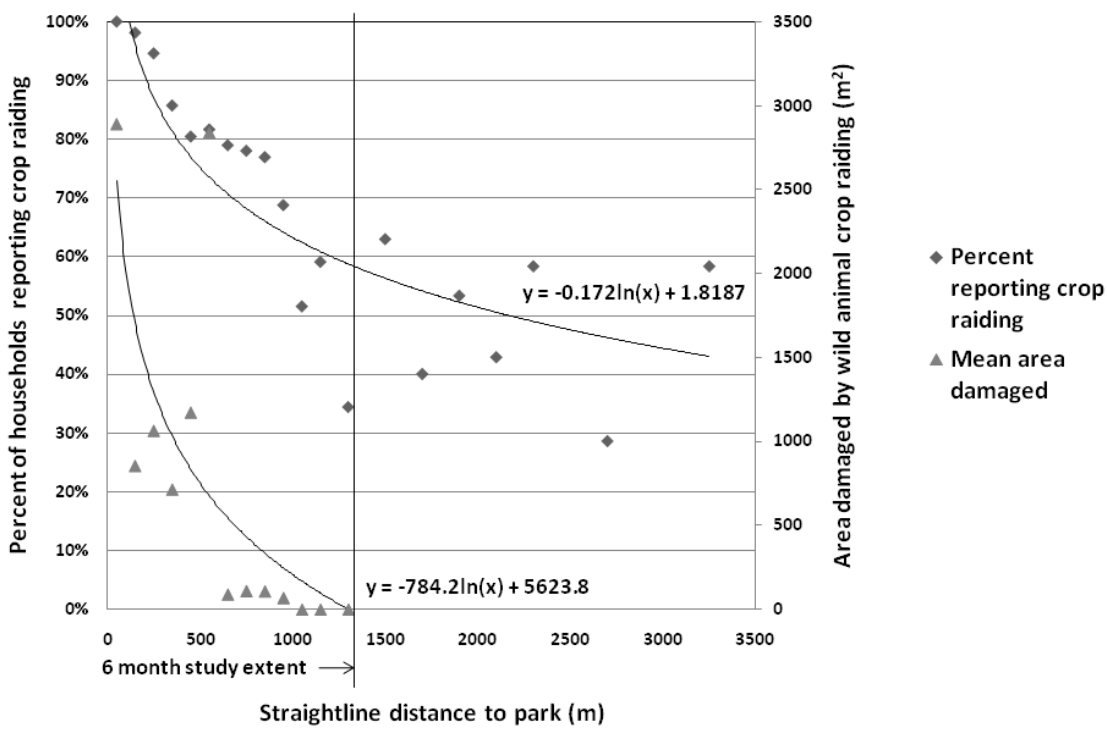



**Figure 6-5: Crop raiding as function of distance from park**

## Table 6-2: Crops grown by study households and raider crop preferences near Kibale National Park, Uganda

Crops representing more than 10% of raids

[a] Not consumed by elephants, but was destroyed by trampling.

[b] Tea was not consumed by baboons. They picked the seedling out of the ground to look for food below ground level before throwing the seedling away.

| *Crop* | *Percent of households growing crop* | *Percent all crop raiding incidents* | *Percent Elephant incidents* | *Percent Baboon incidents* | *Percent Redtail Monkey incidents* | *Percent Vervet Monkey incidents* | *Percent Bushpig incidents* | *Percent Colobus Monkey incidents* | *Percent Chimpanzee incidents* | *Percent African Civet incidents* |
|---|---|---|---|---|---|---|---|---|---|---|
| Beans (*Phaseolus vulgaris* L) | 97% | 9.5% | 8.4% | 10.4% | 6.6% | 12.5% | 13.6% | 9.5% | 0.0% | 0% |
| Sweet potato (*Ipomea batatas* L) | 97% | 7.8% | 5.1% | 8.9% | 6.6% | 12.5% | 13.6% | 14.3% | 6.7% | 0% |
| Maize (*Zea mays* L) | 96% | 39.4% | 30.7% | 40.9% | 61.8% | 47.9% | 40.7% | 71.4% | 6.7% | 100% |
| Matoke *(Musa spp*) | 91% | 10.0% | 13.6% | 9.4% | 2.6% | 10.4% | 0.0% | 0.0% | 20.0% | 0% |
| Cassava (*Manihot esculenta*) | 90% | 14.4% | 19.6% | 12.6% | 6.6% | 6.3% | 22.0% | 0.0% | 0.0% | 0% |
| Fruit (*Passiflora* spp, *Persea Americana*) | 82% | 1.3% | 2.0% | 1.1% | 0.0% | 0.0% | 0.0% | 4.8% | 0.0% | 0% |
| Irish potato (*Solanum tuberosum* L) | 82% | 2.1% | 0.9% | 3.5% | 1.3% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| G nuts (*Arachis hypogeal*) | 76% | 2.1% | 1.6% | 2.9% | 2.6% | 2.1% | 0.0% | 0.0% | 0.0% | 0% |
| Doodo (*Amaranthus dubious*) | 72% | 0.2% | 0.7% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Yams (*Dioscorea spp*) | 71% | 2.7% | 2.4% | 2.2% | 1.3% | 8.3% | 8.5% | 0.0% | 0.0% | 0% |
| Millet (*Eleusine corocana* L) | 71% | 2.7% | 5.3% | 1.1% | 5.3% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Eggplant (*solanum spp*) | 71% | 0.1% | 0.2% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Pumpkin (*Cucurbita spp*) | 69% | 1.1% | 0.4% | 1.9% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Sorghum (*Sorghum bicolor* L) | 61% | 0.8% | 0.4% | 1.3% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Sugarcane (*Saccharum officinarum* L) | 41% | 3.0% | 3.3% | 1.8% | 5.3% | 0.0% | 0.0% | 0.0% | 60.0% | 0% |
| Pineapple (*Ananas comosus*) | 37% | 0.4% | 0.2% | 0.6% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Coffee (*Coffea* spp) | 29% | 0.1% | 0.2% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Green Pepper (*Capsicum* spp) | 24% | 0.2% | 0.2% | 0.2% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Tomato (*Lycopersicon esculentum*) | 20% | 0.3% | 0.9% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Cabbage (*Brassica oleracea* L) | 19% | 0.2% | 0.4% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Tobacco[a] (*Nicotiana tabacum*) | 13% | 0.1% | 0.2% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Tea seedlings[a,b] (*Camellia sinensis*) | 9% | 0.5% | 0.2% | 1.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |
| Rice (*Oryza sativa*) | 7% | 0.6% | 1.1% | 0.3% | 0.0% | 0.0% | 1.7% | 0.0% | 0.0% | 0% |
| Artemisia[a] (*Artemisia* L) | 1% | 0.5% | 1.6% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0% |

## Table 6-3: Crop raiding defence mechanisms used around Kibale National Park, Uganda

Effectiveness rating: 3=works well, 2=works sometimes, 1=does not work

Troubled rating: 5=a lot, 4=considerably, 3=somewhat, 2=a little, 1=not at all

Other includes: Shouting or other noise (9), throwing burning firewood at animals (3), burning plastic or bicycle tires (3), hanging clothes to scare animals (2), burning chili and cow dung (1), burning peppers in the fire (1), hire Bakonjo tribe to kill baboons (1), grow tobacco (1), scare shooting by a local (1).

Survey data collected July-August 2009 and six month study data collected August 2009-February 2010

| | ***Household Survey (n=596)*** | | | | ***Six Month Study: Overlap households with survey (n=90)*** | | | |
|---|---|---|---|---|---|---|---|---|
| ***Defence Mechanism*** | ***Percent of Households using defence*** | ***Mean Effectiveness Rating*** | ***Mean Rating for 'Troubled by Crop Raiding'*** | ***Mean Fraction of Production Lost*** | ***Percent of Households using defence*** | ***Mean number of Incidents*** | ***Mean Area Damaged ($m^2$)*** | ***Mean Financial Loss (USD)*** |
| Human Guarding | 64.9% | 2.08 | 4.62 | 0.43 | 83.0% | 7.6 | 1624 | $79.88 |
| Fire | 53.5% | 2.10 | 4.66 | 0.45 | 71.1% | 7.9 | 1771 | $85.19 |
| Beating drums | 53.4% | 2.08 | 4.61 | 0.44 | 71.1% | 8.2 | 1878 | $92.24 |
| Scare shooting by UWA rangers | 27.2% | 2.17 | 4.66 | 0.43 | 44.4% | 9.4 | 2229 | $106.47 |
| Elephant Trench | 14.4% | 2.07 | 4.56 | 0.45 | 20.0% | 9.3 | 2543 | $95.63 |
| Dogs | 10.9% | 2.11 | 4.28 | 0.42 | 16.7% | 11.0 | 2855 | $137.76 |
| Spread dung on maize | 8.1% | 2.27 | 4.4 | 0.41 | 13.3% | 14.0 | 2558 | $146.50 |
| Live fence (Mauritius thorn) | 5.7% | 2.29 | 4.59 | 0.49 | 12.2% | 8.0 | 1638 | $73.02 |
| Set traps for animals | 1.2% | 1.86 | 4.57 | 0.52 | 2.2% | 18.0 | 5757 | $204.70 |
| Fences | 0.8% | 2.20 | 5 | 0.54 | 2.2% | 16.5 | 1304 | $57.38 |
| Chilli rope | 0.3% | 1.50 | 4.5 | 0.48 | 0.0% | - | - | - |
| Shallow trenches around gardens | 0.2% | 1.00 | 5 | 0.60 | 0.0% | - | - | - |
| Other | 4.4% | 2.00 | 4.31 | 0.42 | 6.7% | 8.0 | 1471 | $56.37 |
| Do not try to stop crop raiding | 6.2% | - | 4.38 | 0.31 | 6.7% | 3.3 | 431 | $25.45 |

Households used numerous methods to defend their crops against wild animals (Table 6-3); the most common were human guarding, burning fires and beating drums to scare animals away. The average time spent guarding per week by men (5.2 days), women (3.9 days) and children (1.5 days) during high crop raiding months was twice that spent at other times of the year. The person days invested in guarding by a household was proportional to the estimated fraction of crop production lost ($r_{Spearman}$=0.333, p<0.001, n=448), the number of raiding incidents ($r_{Spearman}$=0.238, p=0.024, n=90), area damaged ($r_{Spearman}$=0.265, p=0.012, n=90) and financial loss incurred ($r_{Spearman}$=0.250, p=0.017, n=90), suggesting that households were investing human resources based on the level of crop raiding risk to which the household was exposed. Elephant raiding incidents were similar in villages with and without elephant trenches (Mann-Whitney U, p=0.786). However, the trenches appear to have been built in the areas most troubled by elephants, potentially reducing elephant raids in those villages to background levels for the park. The small number of households protected by Mauritius thorn fence (5.7%) experienced the same number of crop raiding incidents as households without thorn fence (Mann-Whitney U, p=0.684).

The most physically threatening crop raiders, elephants and bushpigs, tend to raid at night, when the men sleep in the fields and guard the crops. Therefore the observations of previous studies, indicating female headed households should be more vulnerable to crop losses (Hill, 1997), would be expected to also apply to this area. In both the survey and the six month study 18% of the households were headed by a woman. However, contrary to expectation, female headed households reported less of their crop production lost to wild animals (Mann-Whitney U, p=0.030) and considered their household to be less troubled by crop raiding than male headed households (Mann-Whitney U, p=0.003). In the six month study, female headed households experienced slightly higher mean numbers of raids and financial losses, but the difference was not significant (Mann-Whitney U, p>0.4).

### 6.3.3 Social implications

Household financial losses averaged US$74 over the period of the six month study, a substantial loss given the median capital asset wealth of study households, was US$5,033. In 87% of the focus groups, participants wanted individual cash compensation for lost crops. If cash could not be given, they suggested compensation should be in the form of: access to the park to harvest trees, provision of tree/coffee/tea seedlings, or maize flour and beans to feed their families. Some villagers went so far as to ask “Can we sue in civil court to get compensation from the government?” (Village Focus Group, 11/6/08). The lack of individual compensation left many expressing a desire for revenge; “If a thief pays for his sins, then animals should be speared and killed if there is no compensation” (Village Focus Group, 16/7/08).

UWA does not individually compensate for crop losses (Uganda Wildlife Statute, 1996). However, the park does share 20% of entrance fees collected from visitors with local government councils to benefit communities affected by crop raiding. Although some local farmers see this shared revenue as a substitute for direct compensation, “because the animals eat our crops, UWA decided to give us money from the entrance fees being paid” (Village Focus Group, 19/6/08), there is still confusion whether the shared revenue will come as cash to the village or as a community project that many fear will not benefit those who actually lose their crops. The revenue disbursed to the study villages through the revenue sharing program over the last 10 years, equated to about US$1 per household per year; far less than the crop raiding losses incurred.

Crop raiding also lead to lost opportunity costs; “Men and women are guarding so they have no opportunity to do income generating activities” (Village Focus Group, 3/7/08). In an average household, ten working days per week, throughout the year, are spent by adults crop guarding. At the minimum local wage of US$1/day, paid for agricultural labour, the household is forgoing US$10/week. However, the number of income activities in which the family was engaged was actually higher for households experiencing crop raiding (Mann-Whitney U, p=0.002). Although income generation could potentially be higher if people did not have to guard crops, crop raiding does not seem to be restricting households from engaging in income generating activities.

Focus group participants claimed; “Unemployment is high because we cannot guard our crops all night and then work all day” (Focus Group, 6/7/08). No correlation was found between the percent of adults employed in the household and how much the household was troubled by

crop raiding ($r_{Spearman}$=0.049, p=0.229, n=594) or the fraction of crops lost ($r_{Spearman}$=0.000, p=0.995, n=593). Since employment opportunities in the area are few, the lack of employment is more probably linked to the regional economy than crop raiding. However, local people definitely saw crop raiding as a barrier to development; "How can we make money with crop raiding going on?" (Village Focus Group, 11/6/08). Claiming they had to abandon land near the park and telling us that "local farmers take out loans to invest in their farms but then lose it all when they are crop raided and then are imprisoned for defaulting on the loans" (Village Focus Group, 22/6/08).

Crop raiding does lead to financial hardship for some households and this has broader implications for development. Since many households make their living solely from subsistence farming, loosing crops to wild animals can have serious consequences for food security. In 60% of the focus groups participants said that, "when crops [are] raided, there is not enough food" (Village Focus Group, 22/6/08) and that, "sometimes due to crop raiding we fail to feed our children" (Village Focus Group, 11/6/08). In the survey, 19% of household respondents reported they had experienced a period when their household had no food in the prior year. Food insecure households had lost higher fractions of their staple crops to wild animals than households that had not reported food insecurity (Mann-Whitney U, p=0.002), specifically losing more maize (Mann-Whitney U, p=0.024), yams (Mann-Whitney U, p=0.005) and cassava (Mann-Whitney U, p=0.004).

Although primary education is now free in government schools, private schools are perceived to provide a better education and charge fees to attend (US$130 to over US$500 per year). Even in government primary schools, parents have to pay for uniforms, school supplies, examination fees and school lunches. Focus group participants asked "How can parents send children to school and have to pay for fees, books and uniforms, when they cannot even feed their children?" (Village Focus Group, 22/6/08). Failure to pay fees or provide food for school lunches typically results in a child being turned away from school. In addition, many households hold their children back from school; "Our children cannot spend time in school because they have to guard the crops" (Village Focus Group, 1/7/08). Sixty percent of survey households reported children under the age of 18 guarding crops. Some survey respondents said their children were only allowed to guard crops on holidays and weekends, but 25% of households reported children guarding crops for three to seven days a week during high raiding seasons (May-July & November-January). For the twelve primary schools involved in the scholastic achievement study, children from study villages had lower grade averages than their peers from other villages (paired t-test, p<0.001). Study village students had increasingly poorer school grades relative to their peers as the village-mean perceived trouble to the household caused by crop raiding increased ($r_{Pearson}$=-0.650, p=0.022, n=12).

Focus group participants perceived other negative influences of crop raiding; "Crop raiding brings diseases that spread to humans and livestock" (Village Focus Group, 1/7/08). Survey respondents were asked to self-report if anyone in their household had suffered from certain health problems within the last five years. Households reporting crop raiding were more likely to report malaria ($\chi^2$=9.53, p=0.002), worms and parasites ($\chi^2$=13.64, p<0.001), dysentery ($\chi^2$=8.11, p=0.004), and pneumonia ($\chi^2$=7.64, p=0.006). Focus group participants explained that, "disease is transmitted from animals in the park to people by insects" (Village Focus Group, 11/6/08), but that more "malaria is contracted due to spending nights outside guarding the crops" (Village Focus Group, 3/7/08). The survey respondents also told us which diseases had been suffered by their livestock. Among households where livestock was owned, the following diseases were more likely to be reported in households experiencing crop raiding: tick-borne disease in goats ($\chi^2$=4.37, p=0.037), lumpy skin disease in goats ($\chi^2$=6.77, p=0.009), east coast fever in goats ($\chi^2$=4.48, p=0.034) and pigs ($\chi^2$=7.55, p=0.006), and Fascioliasis in cows ($\chi^2$=5.01, p=0.025).

In one third of the focus groups people expressed their fear of elephants; "We are scared of being killed when guarding against elephants, because they charge and kill" (Village Focus Group, 1/7/08). Almost one quarter of survey households (23%) reported a household member being attacked by wild animals while guarding crops. During the course of the six month study alone, one man lost the use of a hand as a result of a baboon attack and a juvenile elephant chased and injured one woman. Households reporting being attacked by wild animals tended to have lost more crop production to wild

animals (Mann-Whitney U, p=0.002) and invested more person days per week to guard crops (Mann-Whitey U, p=0.001), exposing themselves to a higher potential for injury.

## 6.4 Discussion

Crop raiding has been reported to be escalating around PAs due to increased human settlement and recovering animal populations (Thouless & Sakwa, 1995; Tweheyo *et al*., 2005; Weber *et al*., 2007). This escalating trend appears to also be occurring in KNP, when the results of this study are compared with prior KNP crop raiding studies (Naughton-Treves, 1998, Naughton-Treves & Treves, 2005). However it should be noted, that although there is some overlap between the specific study villages in this and Dr. Naughton-Treves' research, this study extends over considerably more of the park boundary. The spatial extent of crop raiding by park protected animals, defined as the distance within which 90% of the damage had occurred, is extending 300 m farther from the park boundary than prior studies. The percentage of farmers raided by elephant has increased by 66%. However, raiding by bushpig has decreased 55% and redtail monkey raiding has decreased by 59%. Baboons are now the most frequent primate raider. Baboon and elephant combined now account for 93.5% of all area damaged by wild animals. The percent of crop loss, based on the measured area of damage, is also increasing; up over 10%.

As found in other crop raiding studies, the distance to the park boundary was the primary predictor of crop raiding frequency and damage (Hill, 1997; Naughton-Treves, 1998; Naughton-Treves & Treves, 2005; Linkie *et al*., 2007). Therefore, farmers restricted to cultivating directly adjacent to the forest boundary are the most vulnerable. This study also confirmed that having other households between a given household and the park reduced the frequency and area damaged by crop raiding (Hill, 1997). Consistent with other studies, baboon raiding increased just before the harvest (Naughton-Treves *et al*., 1998; Hill, 2000), and guarding investment increased as the risk of raiding rose (Hill, 2000). Elephant raiding in this study may have been under reported as no data was recorded for the peak month of July, as elephant raids tend to decrease from July to December (Chiyo *et al*., 2005).

Human guarding, drums, and fire, the most effective means of reducing crop raiding losses (Sitati *et al*., 2005), were the most frequently employed defence strategies around KNP. Although people who live next to the park want elephant trenches, we found trenches did not significantly reduce elephant raiding below background levels. However, trenches may limit the locations where human guarding is required, as trenches were described as 'like the wall of a house', where guarding now only had to be done at the windows and doors, areas where elephants were more likely to cross. Although new trench excavation is well done, older trenches were too short, incomplete, crossed wetlands, or were poorly maintained. Elephants are smart and learn to cross or go around these obstacles (Osborn & Parker, 2003; Sitati & Walpole, 2006). There was also a perception in villages without trenches that trenches would be effective against baboons. This is not so and could lead to expectations not being met when trenches are excavated. Adult baboons have been seen jumping across trenches and juveniles bend trees above the trench until they can jump off the tree and over the trench. Focus group participants in villages with trenches said the trenches helped if they were maintained, estimating the trenches were 65% effective at stopping elephants. Given the high labour investment required to guard crops, we believe further development and improvement of passive defences should be pursued.

Residents local to KNP definitely wanted to be compensated for their crop raiding losses, as they would if livestock had caused the damage. Lack of compensation by UWA has been called unjust, especially since mitigation efforts by local farmers are restricted by conservation policy (Laudati, 2010). However, in Uganda, like many other African countries, compensation for everyone affected is not possible (Tchamba, 1996; Weber *et al*., 2007). While compensation is cautiously recommended by some (Tchamba, 1996; Naughton-Treves, 1998), such programs would not be sustainable in the long run, since conservation efforts attempt to recover animal numbers and human population growth results in higher numbers of farmers near PAs, leading to ever increasing human-wildlife conflict (Nyhus *et al*., 2005). Even in African countries where compensation exists (Jackson *et al*., 2008), the cost of operating the compensation program is expensive, and compensated farmers still incur an 11% income loss relative to farmers who are not raided.

Compensation schemes are complicated to implement and are fraught with pitfalls. Programs have to guard against fraudulent claims (Bulte & Rondeau, 2005), must have the means to verify that damage was caused by a protected animal (Nyhus *et al*., 2005), and guard against 'moral hazard'; a situation where the farmer looses the incentive to protect their crops, because all are equally compensated for losses (Bulte & Rondeau, 2005; Nyhus *et al*., 2005). Setting up compensation schemes can also run counter to conservation objectives by providing incentive to put more land into agriculture and promote in-migration by people seeking access to crop loss compensation (Bulte & Rondeau, 2005). Given the issues involved with compensation programs and their inherently unsustainable nature, we recommend that any funding found to address the losses incurred by local farmers should be put towards mitigating the loss mechanism by developing better crop raiding defences, an activity that typically is underfunded (Thouless & Sakwa, 1995).

Direct costs have been valued in crop raiding studies (Kagoro-Rugunda, 2004; Naughton-Treves & Treves, 2005) and many authors have acknowledged the social implications of having to guard crops against raiding animals (Tchamba, 1996; Naughton-Treves, 1997; Hill, 2000; Haule *et al*., 2002; Kagoro-Rugunda, 2004). Reducing the levels of crop raiding could also address some of the indirect costs incurred, such as labour opportunity costs, loan defaults, food insecurity, poor educational performance, health problems, and fears of injury from park protected animals. Without exception, participants in the focus groups held around KNP stated that the costs of living near the park far outweighed any benefits they received as a result of conservation. Although the opportunity cost incurred by guarding crops might not be recovered due to few employment opportunities in the area, many of the other indirect costs could be mitigated by directing development aid to households closest to the park.

Data collected from schools near Kilombero Game Controlled Area in Tanzania, found 88.4% of pupils reported that they guarded the family's crops and 60% had missed classes at least once to guard (Haule *et al*., 2002). Around KNP 60% of households admitted their children guarded crops and many focus group attendees voiced concern about having to choose between food to eat and their child's education. The preliminary study at the twelve schools around KNP shows children living in villages directly adjacent to the park have poorer scholastic achievement and that this might be tied to their parent's perceptions of crop raiding risk. Non-Government Organizations (NGOs), linked to conservation, operate around KNP and focus on improving childhood education, literacy and conservation education. Lack of school fees due to crop raiding losses and the need to guard crops instead of going to school could be addressed by these NGOs providing extra support to those living closest to the park.

Given the self-reported nature of the health data in the survey, a more rigorous study is required to confirm the relationship between disease and levels of crop raiding. However, if true, whether or not a household is troubled by crop raiding will need to be considered when planning health programs, particularly malaria prevention. Providing bed-nets for the household will not be effective if people live in fields instead of their house in high guarding season. Bacterial genetic similarity has also been found between humans, livestock and non-human primates in and near forest fragments adjacent to KNP (Goldberg *et al*., 2008). The primary determinant of bacterial transmission was a spatial and ecological overlap between human, livestock, and non-human primate habitats due to human disturbance inside the forest fragments, crop raiding by primates and livestock grazing in and near the fragments. Most of the livestock diseases that were more prevalent for households experiencing crop raiding are transmitted by ticks, and not animal to animal transmission. Therefore transmission is either due to crop raiding animals leaving ticks in the farmers' fields or to livestock grazing on the low brush vegetation inside the boundary of the park where crop raiding animals hide prior to raiding and where ticks thrive (T.L. Goldberg, personal communication), or when livestock are intentionally grazed inside the park as frequently occurs in the south of KNP. The potential for transmission coupled with higher disease reporting rates in households experiencing crop raiding supports the need for further research to fully understand crop raiding implications for human and livestock health.

## 6.5 Conclusions

Crop raiding around KNP is getting worse as conservation initiatives lead to the recovery of animal populations and human population growth and in-migration in search of

land increases agriculture along the park boundary. Given the escalating human-wildlife conflict and the financial and social costs incurred, a compensation scheme would be unsustainable around KNP. Rather mitigation of losses is needed by reducing animal egress from the park.

It is evident that crop raiding is imposing direct and indirect costs on households next to KNP. These direct losses and the indirect social costs suffered by households experiencing crop raiding impede community development. This requires additional attention to be paid to crop loss mitigation by conservation organizations. Funding needs to be allocated for research to improve deterrents to crop raiding, as well as development programs to lessen the social burden borne by local residents while trying to meet global conservation goals. Lowering the levels of crop raiding for local residents could also improve community support for conservation.

# 7 Accruing Benefit or Loss from a Protected Area: Location Matters

In addition to revenue sharing, local people around KNP benefit from ecosystem services, park-based employment, NGO activities, and resource access agreements. Although crop raiding is the most prevalent loss for local residents, they also accrue losses due to livestock predation by park animals, and having to pay fines to UWA when caught illegally inside the park. In this chapter, I describe the value and spatial extent of these benefits and losses around KNP, and determine if the value of these benefits and losses are related to attitudes towards the park or illegal resource extraction from the park. This chapter, in a revised form, was accepted for publication by *Ecological Economics* following the final submission of this dissertation.

**Summary:** Communities neighbouring protected areas disproportionally accrue the costs of conservation, but they also benefit. The spatial distribution of protected area benefits and losses were mapped for twenty-five villages around Kibale National Park, Uganda. Benefits included park-based employment, tourism, integrated conservation and development projects, and resource access agreements. Losses were primarily caused by park-protected animals raiding crops and preying on livestock. Eight villages accrued an annual net benefit as a result of the park, while 17 villages accrued a net loss. These benefits and losses were compared with perceived costs and benefits of living next to the park, and with measured illegal extraction from the park near the study villages. Perceived benefit of living next to KNP was driven by resource access agreements and park-based employment, while perceived costs were driven by crop raiding. Only resource access agreements and tourism revenue sharing were associated with lower illegal resource extraction from the park, while predation losses were associated with higher animal poaching. Attitudes towards the park appear to be shaped by loss aversion, suggesting loss mitigation, rather than benefit provision, would more effectively increase positive perceptions of the park, while meeting the conservation objective of lowering illegal resource extraction.

## 7.1 Introduction

The benefits of protected areas (PAs) tend to accrue globally, while their creation results in a complex web of gains and losses for local people (West *et al*., 2006; Adams & Hutton, 2007). Recreation opportunity, existence and bequest value (Tisdell, 2005), and the utility of forested PAs to sequester carbon (Grieg-Gran & Bann, 2003; Naughton-Treves *et al*., 2005) are most often realized by people in more developed countries. Communities situated on the boundaries of PAs disproportionately assume the costs of conservation (Nyhus *et al*., 2005; Ninan *et al*., 2007), as a result of eviction (Brockington & Igoe, 2006), crop and livestock raiding by park-protected animals (Nyhus *et al*., 2000; Naughton-Treves & Treves, 2005), and exclusion from resources (Ferraro, 2002). However, local people may also benefit from PAs through the preservation of ecosystem services (Millennium Ecosystem Assessment, 2005), tourism (Archibald & Naughton-Treves, 2001), implementation by non-governmental organizations (NGOs) of integrated conservation and development programs (ICDPs; Barrett & Arcese, 1995) and through resources access agreements (Chhetri *et al*., 2003).

Exchange theory dictates that rational humans base their behavioural choices on maximizing gains and minimizing costs (Shogren *et al*., 1999), implying that if local people benefit from the existence of a PA, they will be more likely to support conservation and the continued existence of the PA. Therefore it has been proposed that certain material benefits can offset accrued costs, while also making conservation economically beneficial to communities near PAs (McNeely, 1988; Ferraro & Kiss, 2002). Offsetting costs by providing benefits may support biodiversity conservation while attempting to limit negative consequences for local livelihoods (Adams *et al*., 2004).

Assessment of the influence of benefits has been cautiously optimistic (Archibald & Naughton-Treves, 2001; Spiteri & Nepal, 2008); although the recognition of benefits can be limited and may not sufficiently accrue to those carrying the greatest conservation costs. A number of studies on the negative influences of PA creation have been written in recent years (Naughton-Treves & Treves, 2005; Brockington & Igoe, 2006; Brockington *et al*., 2006), focusing on issues such as eviction, exclusion, and crop raiding. Balanced assessments of the benefits and losses

accrued by local communities have been reported by Dixon and Sherman (1991) and Spiteri and Nepal (2008), but in general there is a dearth of research upon which to base the development of conservation incentive strategies that will protect biodiversity and minimize negative externalities assumed by local communities (Wilkie *et al*., 2006; Igoe, 2006; Brockington *et al*., 2008). This paper aims to provide an assessment of the benefits and losses accrued by communities living next to Kibale National Park, Uganda, and to investigate whether benefits and losses accrued as a result of the existence of the park affect attitudes and behaviours of local residents, where behaviours are represented by measured illegal resource extraction from the park.

## 7.2 Methods

### 7.2.1 Study site

Kibale National Park (KNP) is a 795 km$^2$ mixed evergreen forest and savannah grassland PA located in south western Uganda (Fig. 7-1). The park provides protected habitat to chimpanzees (*Pan troglodytes*), 12 other primate species, elephants (*Loxodonta africana)*, and a high level of biodiversity in other taxa (Chapman & Lambert, 2000). The Uganda Wildlife Authority (UWA) manages KNP and uses a 'Park and Neighbour' conservation strategy (Jones, 2006), including seven specific components: 1) strict monitoring and enforcement of boundaries, 2) forest restoration by a carbon sequestration program (FACE the Future, 2011), 3)ecological conservation research, 4) community and education outreach, 5) negotiated resource access for community associations (Chhetri *et al*., 2003), 6)encouraging tourism, and 7) sharing 20% of park entrance fees with local communities (Archibald & Naughton-Treves, 2001).

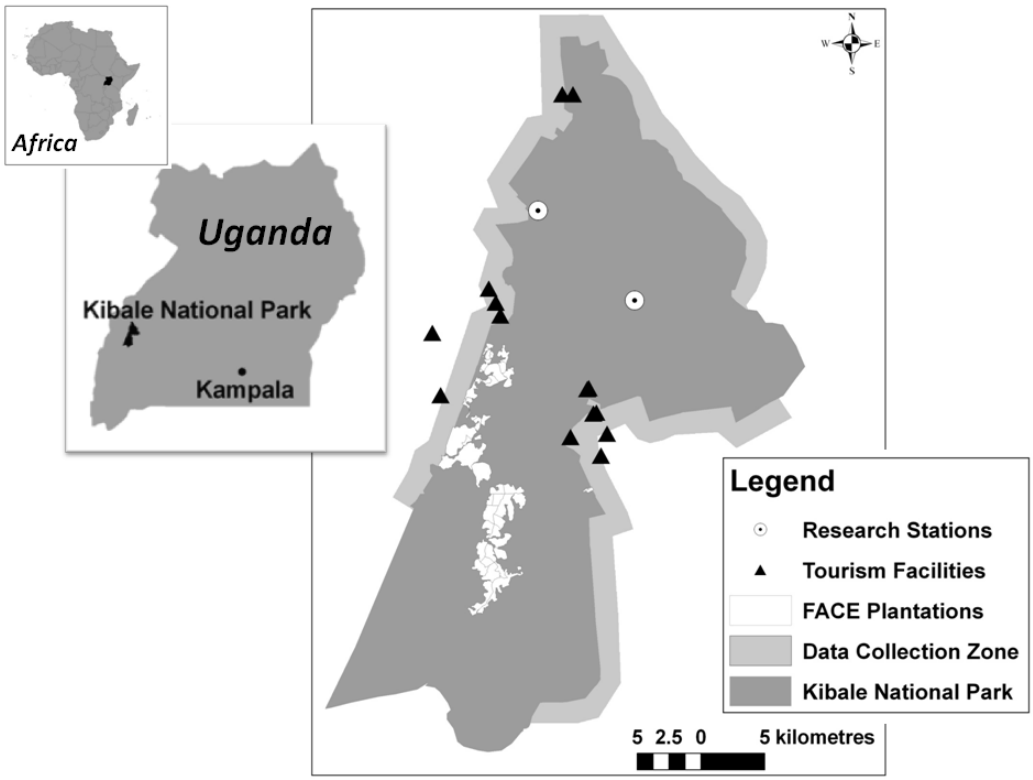



**Figure 7-1: Kibale National Park, Uganda and data collection zone**

The ability to access resources inside the area that is now KNP has been restricted throughout the area's recorded history. In the 1800s, the tribal Toro king placed restrictions on hunting that were further tightened by the British colonial government in the early 1900s (Naughton-Treves, 1999). The post-colonial government maintained the forest reserves and game corridors set-up by the British. However, these reserves were overrun and settled by rebels and people seeking refuge during the 1971-1987 civil war (Hamilton, 1984). It is estimated that up to 50,000 people settled in the area now covered by KNP (Aluma *et al*., 1989). Following the war, the Ugandan government reclaimed Kibale forest reserve and game corridor, evicting approximately 35,000 people (Feeney, 1998; Naughton-Treves, 1999), although estimates of the actual number evicted vary from 8,000 to 170,000 (Chapman & Lambert, 2000). KNP was established as a national park in 1993, and people continued to be excluded from accessing park resources, with the exception of contractually negotiated access to non-threatened resources (Chhetri *et al*., 2003).

Since the area had restricted use since the 1800s, and 56% of household heads in villages surrounding KNP migrated to the borders of the park within the last generation, it can be argued that a majority of local residents never could lay claim to land inside the park. Therefore the opportunity cost associated with the inability to cultivate inside the park is not a valid cost accrual, and has not been included in this study. The lost opportunity to extract resources from the park has also not been valued in this study, because the volume of permitted extraction prior to the creation of KNP is unknown (Ferraro, 2002). Restrictions on access to firewood, construction poles, in-park grazing, water and medicinal plants existed even when the park was a forest reserve. Therefore, it is debatable whether loss of access to these resources is a valid opportunity cost.

African rural communities are heterogeneous (Agrawal & Gibson, 1999), and benefits and losses differentially accrue among households (Brockington *et al*., 2008). However, the valued benefits and losses presented in this paper are accounted at the village scale for three reasons: 1) revenue sharing and many NGO projects are provided at the village scale, 2) to capture a broader geographic perspective of the spatial distribution of benefit and loss accrual around the park, and 3)because illegal resource extraction, measured inside the park near study villages, is a village-aggregate value that cannot be attributed to any one household. In this study, a village is defined by the spatial extent of households associated with a village name under the leadership of one village chairperson. Twenty-five villages participated, chosen based on village members holding and/or cultivating land next to the park. Villages were located approximately five kilometres apart along the park boundary, within the data collection zone (Fig. 7-1), but are not explicitly identified, since reporting illegal resource extraction might lead to retribution from the wildlife authority (Robbins *et al*., 2006). Therefore, inverse distance weighted interpolation to rasterize data to a 1,000 m grid, followed by re-sampling to a resolution of 30 m within the data collection zone has been used to visualize village-level data.

### 7.2.2 Data collection

**Focus groups:** To understand the problems and benefits of living next to KNP, focus groups were conducted in 15 villages (60%) in June and July 2008. Focus groups often grew beyond those invited, resulting in the number of participants ranging from 16 to 51. Women were in attendance at 14 of the focus groups, representing up to 65% of the participants. Focus group participants were asked to list all problems of living next to the park and then all benefits. At the end of the meeting we asked if the disadvantages of living near the park were offset by the benefits.

**Household survey:** In July and August 2009, a household survey was conducted in all 25 villages. Within each village 24 households were surveyed[1]. A total of 596 surveys were collected, representing 24% of all households in the 25 villages (2480), with a median of 28.6% of households surveyed per village. Households were chosen by random stratified sampling, with stratification based on house construction standard (Ellis & Bahiigwa, 2003), a proxy for household wealth. The survey was administered by one female and three male, Ugandan field assistants, in the two local languages (Rutooro and Rukiga).

Based on self-identification of benefits and problems of living next to the park collected during focus groups, survey respondents were asked to rate their perceived

[1] The number of households in a study village ranged from 41 to 242, median=84.

benefit or loss associated with each benefit or problem on a five-point Likert scale ranging from no benefit or loss to a lot of benefit or loss. They were then asked to describe specific benefits their household or village had received. They were also asked to estimate the number and type of domestic animals lost to predation and the fraction of crop yield lost to wild animal crop raiding, and to report if they had been fined by UWA for illegal activity inside the park. At the end of the survey, the respondent was asked to rate the overall benefit or loss of living next to KNP on the same five-point scale.

During the focus groups, people described the park as a good or bad neighbour based on the impact the park had on their lives. They also described themselves as either a good or bad neighbour to the park based on whether they helped to conserve the park or engaged in illegal activity in the park. To gauge attitudes toward the park, the survey included three final questions, rated on a five-point scale from very bad to very good with a neutral mid-point: 'how do you rate your relationship with UWA?'; 'what type of neighbour do you consider the park to be?'; and 'what type of neighbour are you to the park?'

**Crop raiding study:** Crop raiding was the largest loss and since people tend to overestimate losses due to crop raiding in the hopes of receiving compensation (Tchamba, 1996), data were collected to verify the damage incurred. The type of crop and area damaged by park-protected animals was recorded weekly for six households in each village from August 2009 to January 2010.

**Interviews:** Interviews were conducted from May 2008 to January 2010 with UWA wardens, district, sub-county and village chairpersons, resource access agreement (RAA) association chairpersons, and managers of tourist facilities, research operations, NGOs and the FACE the Future Foundation carbon offset project (FACE the Future, 2011). Enterprises providing park-based employment (tourism, research, FACE, and UWA) were asked about the number of employees, the average pay rate, and to identify the home village of their employees. UWA and district, sub-county, and village chairpersons were asked to summarize their experience with the revenue sharing program and to state the money distributed or received from 1999 to 2009. RAA chairpersons were asked to describe their agreement with UWA, the number of association members and their home villages, and to estimate the value or resource yield of the agreement for the average association member. NGO managers were asked to describe the nature of the benefits they provided and to provide a summary of the financial investment made, and the timeframe they had been active around KNP.

**Illegal extraction data:** Resource extraction from KNP was quantified by walking a 600 to 850 m transect along the park boundary neighbouring the village, recording the number of harvested trees, entry trails, livestock seen grazing inside the park, and poaching evidence (Olupot & Chapman, 2006). Data were collected along 19.5 km of the park boundary, which represents 15.2% of the data collection zone (Fig. 7-1). Park entry trails were followed to record the same information. UWA records of RAAs were used to remove legally harvested trees and sanctioned entry trails from the data. Measured extraction data from each village boundary were normalized by the length of the boundary sampled for that village, and treated as an independent data point to be compared with data from the adjacent village. Data were collected from May to August 2008 and in June 2009.

### 7.2.3 Analysis

**Valuation:** The village-value of park-based employment was calculated by multiplying the number of people employed by a given enterprise in a village by the average salary paid by that enterprise. The annual budget for each NGO was divided between all villages where the NGO was active, and then the benefit from each NGO was summed to represent the financial value of NGO activity for each village.

The monetary value of the revenue sharing program assigned to each village was based on project use. For example, if a school with 600 students received a funded project but only 120 students came from the study village, only 20% of the monetary value was allocated to the village. Alternatively, if the project was an elephant trench built between a village and the park to stop elephants from destroying village crops, then the full cost of the trench was assigned to that village.

The value of RAAs was equal to the number of people who were members of an RAA association in the village (typically three to 30 households), multiplied by the value of the RAA per person and was based on information supplied by the RAA chairperson. For example,

beekeeping RAA valuation was calculated as the mean volume of honey produced by each association member per year, multiplied by the market price of honey.

Crop raiding losses were calculated as the local market value of the crop/$m^2$ multiplied by the area damaged. The yield of each crop was based on random farmer estimates from different locations around the park to allow for variation in geographic location and farming methods. The mean value of crop loss recorded for the six crop raiding households was applied to the proportion of village households reporting crop raiding (survey data), then summed for each village.

In the survey, respondents reported that their livestock had been predated by lion (*Panthera leo*), leopard (*Panthera pardus*), serval cat (*Felis serval*), civet cat (*Civettictis civetta*), hyena (*Crocuta crocuta*), jackel (*Canis adustus*), baboon (*Papio cynocephalus*), and mongoose[2] (*Herpestidae* spp.) The reported number of cows, goats, sheep, pigs and chickens lost to wild animals was multiplied by the market value of each domestic animal and summed for each household. The village-mean household predation loss was scaled by the proportion of households reporting predation in that village for a village-scale predation loss. The value of UWA fines was equal to the total value of fines reported by survey respondents in a village, scaled by the ratio of all households to survey households in the village.

All monetary benefits and losses were annualized and the exchange rate to convert from Uganda Shillings to US Dollars was dependent upon the timeframe over which the benefit or loss was accrued[3].

**Statistical analysis:** All village-scale variables were normally distributed with the exception of park-based employment benefit, RAA benefit, the number of livestock seen grazing in the park and poaching signs. Pearson correlations were used if both variables were normally distributed. If one or both variables did not exhibit a normal distribution, a Spearman correlation was used.

Household-scale data were categorical and not normally distributed. Therefore the gamma association test was used to compare household-scale survey responses to see if the answers statistically aligned between questions (Healey & Prus, 2010). As gamma approaches one or minus one, the answers of survey respondents to two different questions are positively or inversely aligned. The gamma association test was used to study how the perceived benefit or loss of specific benefits or problems, aligned with the overall perceived benefit or loss of living next to KNP, household attitudes about having the park as a neighbour, and the household's relationship with UWA.

## 7.3 Results

### 7.3.1 Benefits

**Ecosystem Services:** Focus group participants said: "The trees keep it cooler and make more rain" (10/7/08), "we no longer face serious drought because we are always showered with rainfall" (13/7/07), and "other places are hotter and drier because they are away from the park" (13/7/08). Ninety-seven percent of survey respondents said they benefitted from more rainfall, and 94% said the soil fertility was better closer to the park. The perceived benefit of rainfall plus soil fertility was higher on the eastern side of the park (Fig. 7-2a). Unfortunately, due to a lack of data on rainfall and soil fertility around and moving away from the park, this benefit could not be valued.

**Park-based Employment:** The existence of the park does lead to employment opportunities for local people. The FACE the Future Foundation is reforesting the areas cleared by the evicted settlers. Seasonally, they employ up to 367 people, 96% of whom live close to the park (Fig. 7-2b). The Makerere University Biological Field Station (MUBFS) attracts domestic and foreign researchers who

[2] Mongoose is a common predator in Uganda, however, households were more likely to report mongoose predation closer to the park (Mann-Whitney p=0.022)

[3] Park-based employment benefit and crop raiding loss were converted using the mean exchange rate for August 2009 to February 2010 = 2,250 Ugandan shillings/US$, revenue sharing benefit used the mean exchange rate from 1999 to 2008 (the distribution period) = 1,600 Ugandan shillings/US$, NGO activities were active around KNP from 1998 to 2009 and averaged annualized benefits were converted using an exchange rate of 1,681 Ugandan shillings/US$, finally predation losses and UWA fines collected in the survey referenced a 10 year period prior to the survey (July 1999 to July 2009) and used a mean exchange rate of 1,772 Ugandan shillings/US$.

hire assistants from local villages. In January 2010, long-term research projects employed 93 Ugandans, 91% from villages near the park (Fig. 7-2c). Habituated chimpanzees attract 7,650 visitors annually (UWA, 2009), creating full and part-time employment in tourist facilities for over 250 people, 87% from near the park (Fig. 7-2d). UWA employs eight wardens, 54 rangers and 36 support staff (clerks, porters and trail cutters). All support staff come from villages near the park (Fig. 7-2e), while wardens and many rangers come from other areas of Uganda. Park-based employment tended to be near carbon reforestation, research, tourism, and UWA operations, clustered toward the centre of the park. The annual financial value of park-based employment salaries to residents of the districts surrounding KNP totaled US$570,839 (FACE US$132,463, Research US$127,176, Tourism US$271,200, UWA US$40,000). The value of park-based employment to study villages ranged from nothing to over US$27,000 per year.

**Revenue Sharing Program:** The revenue sharing program has been law in Uganda since 1996, and requires that 20% of park entrance fees be distributed for community projects in villages next to the park. From 1999 to 2009, 55 projects had been implemented with the US$150,000 distributed. Initially, money was spent on schools and health clinics, but more recently is spent on crop raiding defences and income generation projects. The projects are distributed around the park, within 7 km of the park boundary (Fig. 7-2f). The financial benefit of the revenue sharing program to study villages averaged between 0 and US$457 per year.

**Non-Governmental Organizations:** Most NGOs around the park were initiated by researchers to help communities near their research locations: The Kasiisi project supports primary and secondary education (Kasiisi Project, 2008), Books Open the World (BOW) created libraries and supports literacy education (Books Open the World, 2007), Kibale Fuel Wood Project provides tree seedlings to improve wood availability and teaches people to build energy saving stoves to conserve firewood (Kibale Fuel Wood Project, 2011), and the Kibale Health and Conservation Centre has been built to support the medical needs of six communities (Kibale Health & Conservation Project, 2011). Independent ICDP projects have also been initiated by the International Union for Conservation of Nature (IUCN), to develop crop raiding defenses and improve income generating opportunities (Chhetri *et al*., 2004), and by Uganda and North Carolina's International Teaching for the Environment program (UNITE) to improve conservation education in Ugandan primary schools (Uganda UNITE, 2007). The Kibale Association For Rural and Environmental Development (KAFRED) is a community-based conservation organization (Lepp & Holland, 2006), supporting development projects through community-based tourism. NGO activity is primarily clustered along the northwest and central eastern boundary (Fig. 7-2g), where most of the research for KNP is done. The annual financial value of all NGO activities totaled over US$135,490, with study village financial value ranging from 0 to over US$4,800 per year[4].

**Resource Access Agreements:** Community associations can negotiate with UWA for access to non-threatened resources inside the park. In return for access, the community associations promise to report illegal activity inside the park, manage their activities so as not to endanger park animals, and help educate other villagers about conservation (Chhetri *et al*., 2003). Current agreements permit keeping beehives, collecting craft materials, harvesting exotic tree species, and fishing in two lakes inside the park (Fig. 7-2h). In the past there were also agreements for firewood collection, watering cattle and picking wild coffee, discontinued due to resource exhaustion, non-compliance with park rules, or collapse of the community association. The financial value of RAAs to study villages ranged from nothing to over US$2,500 per year, with the highest value accruing to villages with beekeeping RAAs.

**Total Valued and Perceived Benefit:** The summation of village-valued benefits ranged from nothing to US$28,460 per year (Fig. 7-2i); with the highest benefit accruing near park-based employment. The village-mean perceived benefit of living next to KNP correlated with the village-valued total benefit ($r_{Pearson}$ =0.540, p=0.005, n=25), even though the valued benefit did not include ecosystem services.

The highest perceived benefit rating by survey respondents was for ecosystem services, followed by park based employment (Table 7-1). Household response for the overall perceived benefit of living next to KNP most

[4] Valuation includes all NGOs except UNITE, for which financial data was not made available.

closely aligned with the perceived benefit of RAAs and park-based employment, but was not related to the perceived benefit of ecosystem services (Table 7-1); suggesting people may take ecosystem services for granted while RAAs and employment are perceived as more tangible benefits.

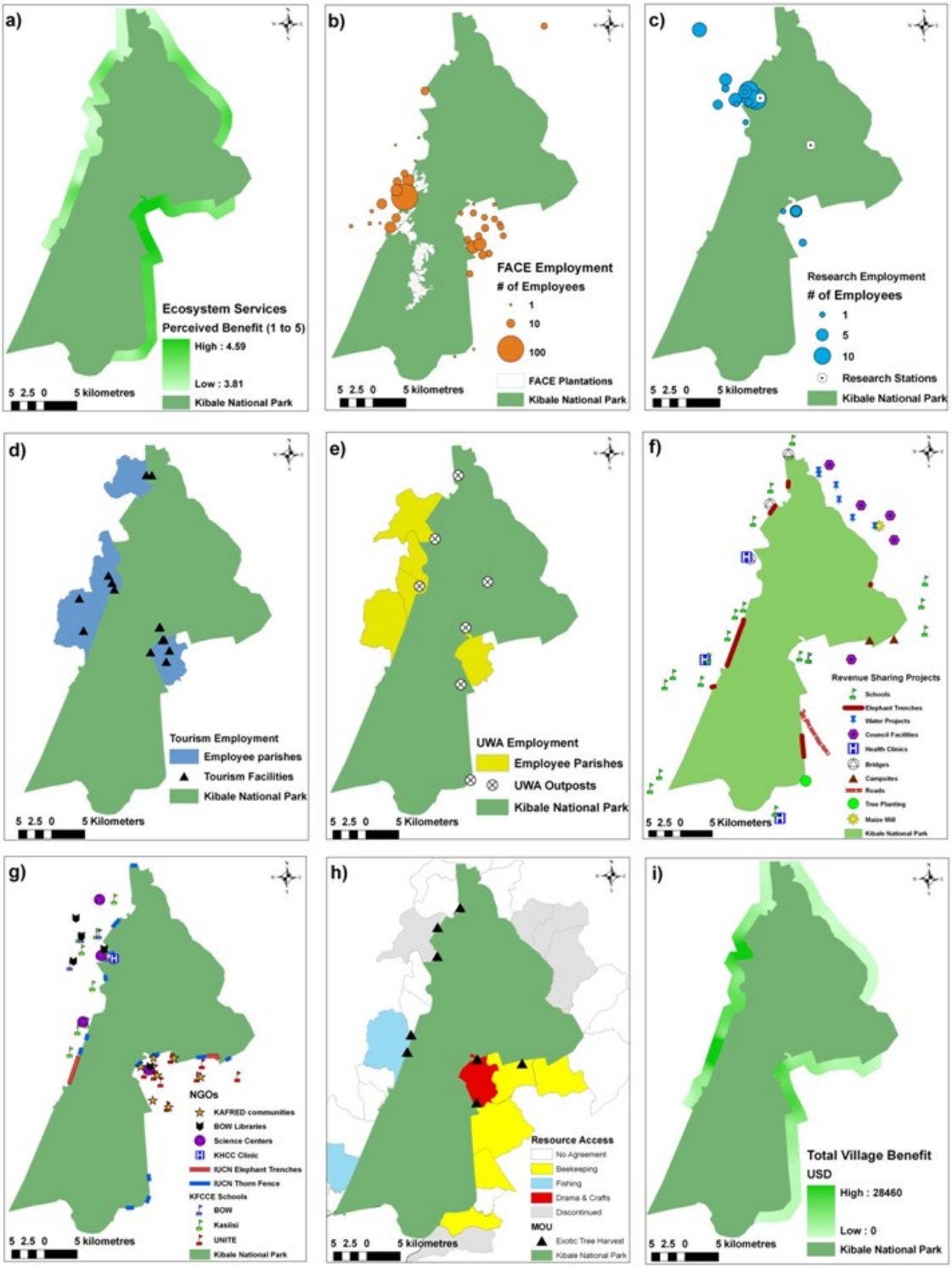



**Figure 7-2: Spatial distribution of benefit sources around Kibale National Park**

a)Ecosystem services, b)Carbon sequestration employment, c)Research employment, d)Tourism employment, e)UWA employment, f)Revenue sharing projects, g)NGO activities, h)Resource access agreements and i)Total village-valued annual benefit

**Table 7-1: Perceived benefit of specific benefits against overall perceived benefit of living next to KNP**

| *Benefit (n=596)* | *Mean Perceived Benefit* | *Gamma* | *p* |
|---|---|---|---|
| **Ecosystem services** | 4.53 | 0.081 | 0.206 |
| **Park-Based Employment** | 1.79 | 0.335 | <0.001 |
| **Revenue Sharing** | 1.66 | 0.162 | 0.006 |
| **NGO Benefit** | 1.64 | 0.284 | <0.001 |
| **Illegal Extraction** | 1.50 | 0.223 | 0.001 |
| **Resource Access** | 1.23 | 0.368 | <0.001 |

### 7.3.2 Losses

**Evictions:** Only three of 596 survey respondents (0.5%) stated their household had been evicted from the park. Since so few evictees remained in the study villages, this loss was not valued.

**Crop Raiding:** Seventy-three percent of survey households reported crop raiding, with 90% of the damage occurring within 0.5 km of the park boundary. The average crop raiding study farmer lost US$148 per year to park-protected animals eating or damaging their crops. Losses to crop raiding were not uniform (Fig. 7-3a), with village-valued loss ranging from nothing to over US$24,000 per year.

**Predation:** The average household in the survey had lost US$16 per year to predation. Predation losses were higher in households with more livestock ($r_{Spearman}$=0.238, p<0.001, n=594) and tended to be higher closer to the park ($r_{Spearman}$=-0.129, p=0.002, n=594). Some villages were more prone to predation than others (Fig. 7-3b). The village-valued loss to predation ranged from US$426 to US$9,154 per year.

**Risk of Injury**: Twenty-three percent of survey respondents reported they had been attacked by park-protected animals while trying to protect their crops and livestock. The impact of these attacks ranged from severe physical injury, one man had lost the use of one hand after a baboon attack, to psychological distress, "people are scared of elephants so cannot harvest crops" (Village Focus Group, 3/7/08). Given the wide range of personal injury outcomes, financial valuation of these attacks was beyond the scope of this study.

**Arrested and Fined by UWA:** Only 4.7% of survey households had had a household member arrested for poaching resources in the park, and only 6.4% had been fined by rangers for being in the park illegally. Reported fines ranged from US$5 for collecting firewood to US$100 for grazing livestock. The valued loss to UWA fines per village ranged from nothing to US$140 per year (Fig. 7-3c). The likelihood of arrests/fines was higher within 8 km of a UWA outpost (t-test, p=0.003).

**Total Valued and Perceived Loss:** The village-valued total loss ranged from just over US$1,000 per year to over US$25,000 per year (Fig. 7-3d). The village-mean perceived loss of living next to KNP did not correlate with valued loss ($r_{Pearson}$=0.134, p=0.522, n=25). Household ratings of loss caused by living near the park tended to be high, with any loss considered to be a major burden for the household. The highest perceived loss rating was for crop raiding (Table 7-2). Responses for the household perceived loss of living next to KNP was strongly associated with perceived loss to crop raiding, but was only weakly tied to the risk of injury from park animals, predation of livestock and loss of access to park resources (Table 7-2). Arrests and fines did not influence the overall perceived loss of living next to KNP.

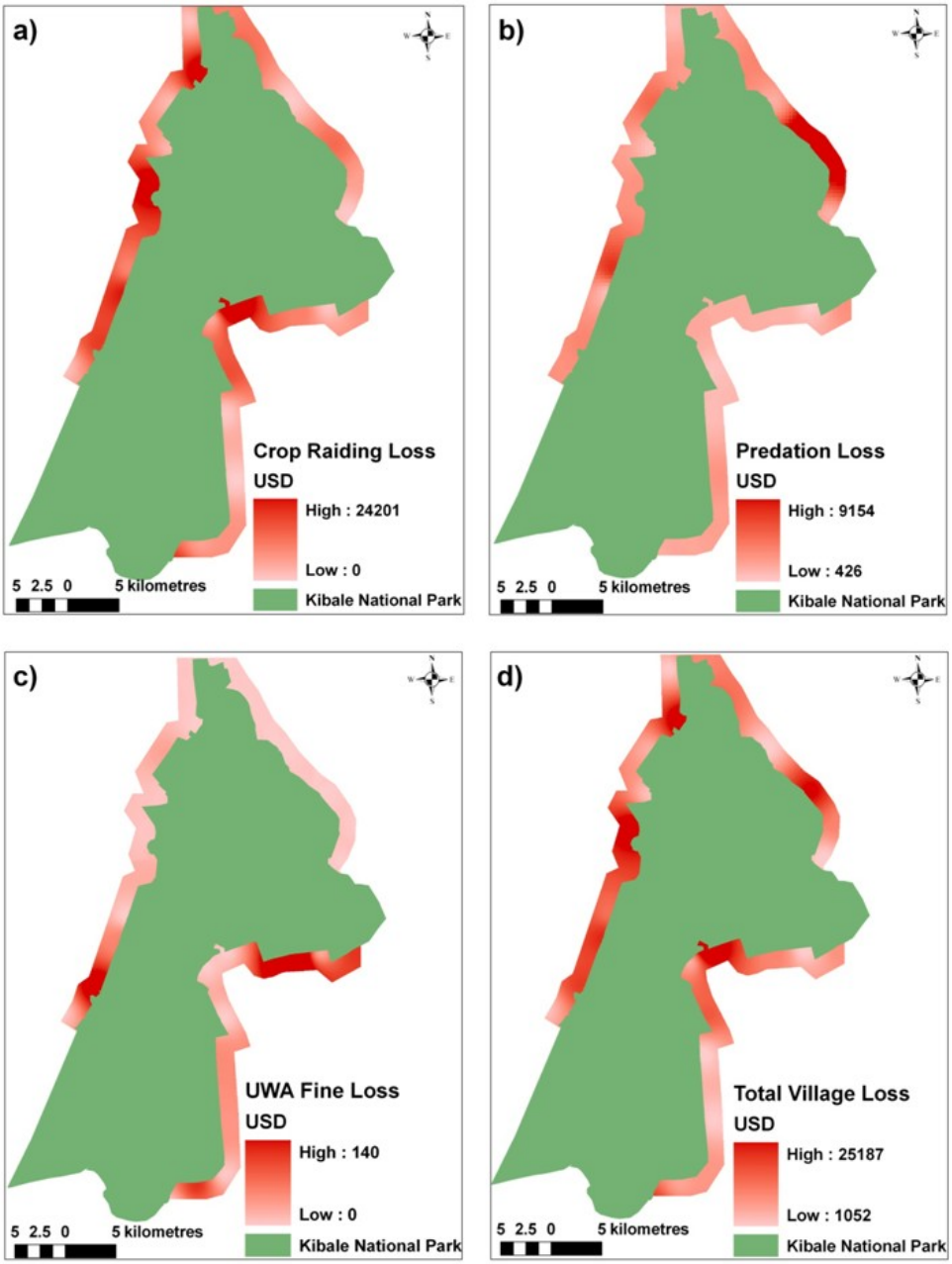



**Figure 7-3: Spatial distribution of losses around Kibale National Park**
a)Crop raiding, b)Predation, c)UWA fines and d) Total village-valued annual loss

**Table 7-2: Perceived loss of specific losses against overall perceived loss of living next to KNP**

| *Loss (n=596)* | *Mean Perceived Loss* | *Gamma* | *p* |
|---|---|---|---|
| **Crop Raiding** | 3.87 | 0.767 | <0.001 |
| **Lost Access to Resources** | 3.58 | 0.155 | 0.002 |
| **Predation** | 2.87 | 0.220 | <0.001 |
| **Injury from Wild Animals** | 1.56 | 0.289 | <0.001 |
| **Fines & Arrests by UWA** | 1.19 | 0.109 | 0.356 |

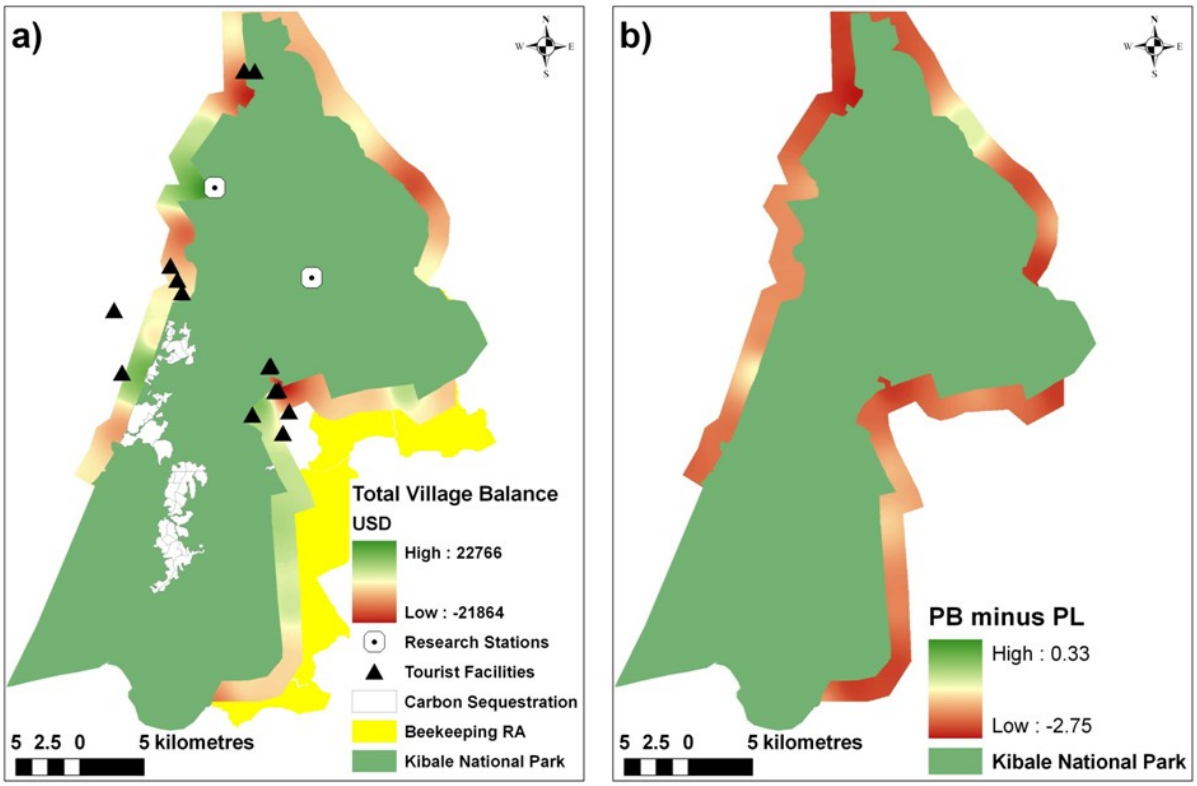



**Figure 7-4: Benefit-loss balance around Kibale National Park**
a) Village-valued benefit minus loss and b) Village-mean perceived benefit minus perceived loss

### 7.3.3 Village-scale balance of benefits and losses

The village-valued total loss subtracted from total benefit (Fig. 7-4a) resulted in eight villages summing to a net annual benefit, while 17 villages summed to a net annual loss, illustrating that benefits and losses are spatially inequitably distributed. Villages with a net benefit were located near park-based employment or were villages with beekeeping RAAs. The village-mean perceived benefit minus the village-mean perceived loss (Fig. 7-4b) did correlate with the village-valued net balance ($r_{Pearson}$ =0.446, p=0.026, n=25), but showed a definite bias towards perceived loss, even in net benefitting villages. This was supported by all focus group participants responding that park-based problems were much bigger than park-based benefits[5].

### 7.3.4 Measured illegal extraction from KNP

Village-valued benefits, losses, and balance, were compared with each of the illegal extraction measures: trees harvested, entry trails, livestock grazing in the park and signs of animal poaching (Table 7-3). Only RAA village-valued benefit related to a reduction in illegal activity, where illegal entry trails were lower near villages with higher valued RAAs ($r_{Spearman}$=-0.492, p=0.013, n=25). The RAA providing the most financial benefit is beekeeping and a village chairperson explained that, "beekeepers guard the park and stop others entering"(21/7/08).

[5] Five focus group villages were net benefitting, while ten accrued a net loss.

**Table 7-3: Illegal resource extraction found along the boundary of Kibale National Park**

| *Illegal Extraction Measure* | *Village-Mean Value (± SD)* | *Percentage of villages where disturbance was found (n=25)* | *Total recorded for 19.5km of boundary* | *Per km of boundary measured* |
|---|---|---|---|---|
| **Harvested trees/km boundary** | 159.7 (± 147.6) | 100% | 2794 | 143.6 |
| **Illegal trails/km boundary** | 4.6 (± 6.6) | 60% | 73 | 3.8 |
| **Number of poaching signs** | 0.89 (± 2.8) | 24% | 24 | 1.2 |
| **Number of grazing livestock** | 13.8 (± 41.9) | 36% | 373 | 19.0 |

Park-based employment ($r_{Spearman}$=0.437, p=0.029, n=25) and NGO ($r_{Pearson}$=0.573, p=0.015, n=25) village-valued benefits correlated with higher illegal tree harvesting, potentially due to demand for these resources being higher near park-based employment and NGO activities. Pitfall traps and snares for poaching were more prevalent near villages with high village-valued predation loss ($r_{Spearman}$=0.655, p<0.001, n=25), one village chairperson suggested people may extract resources as compensation for losses from the park: "People have lost a lot ... so locals go to the park to get meat" (8/6/09). No other specific benefit or loss, total benefit, total loss, or net balance correlated with any illegal extraction measures.

### 7.3.5 Conservation attitudes toward UWA & KNP

The household relationship with UWA was on average rated as good, and was inversely related to the perceived benefit of illegal resource extraction, which I interpret as being due to the risk of being caught by UWA (Table 7-4). The household relationship with UWA improved as the perceived benefit of ecosystem services rose. However, high perceived loss to livestock predation and personal injury caused by wild animals were related to low ratings of the relationship with UWA, since people stated that UWA should be protecting them; "I see this as UWA's responsibility to keep animals in the park"(5/6/09).

The mean rating of the park as a neighbour was 3.62, falling between neutral and good on the five-point scale. The rating of the park as a neighbour increased with increased perceived benefit of ecosystem services, but decreased with perceived benefit of illegal extraction from the park, and the perceived losses of crop raiding and personal injury caused by wild animals (Table 7-4). The village-mean rating of the park as a neighbour also decreased as the village-value of crop losses rose ($r_{Pearson}$=-0.449, p=0.024, n=25). Higher village-value of revenue sharing projects correlated with higher village-mean ratings of the park as a neighbour ($r_{Pearson}$=0.482, p=0.015, n=25), suggesting the program improves perceptions of the park at the community level. Also, the type of project chosen to implement with revenue sharing funds was important, as illegal extraction was less near villages with elephant trenches to help reduce crop raiding losses (t-test, p=0.015).

The household respondent's perception of how well they treated the park (*i.e.*, whether they were a good or bad neighbour), was generally rated high with a mean rating of 4.26, between good and very good. They recognized that illegal extraction of resources meant they were a bad neighbour, as high perceived benefit from illegal extraction associated with low ratings of themselves as a neighbour to the park (Table 7-4). At the village-scale, a high valued loss as a result of UWA fines correlated with low village-mean rating of the household as neighbour to the park ($r_{Spearman}$=-0.407, p=0.044, n=25). Also high ratings of the household as a park neighbour were associated with high household perceived benefit from NGO activities (Table 7-4), presumably because NGOs often educate people about the need to conserve KNP.

**Table 7-4: Conservation attitudes, perceived benefit & perceived loss**

Attitude rating: 1=very bad, 2=bad, 3=neutral, 4=good, 5=very good
Perceived benefit rating: 1=none, 2=a little, 3=some, 4=considerable, 5=a lot
Perceived loss rating: 1=none, 2=a little, 3=some, 4=considerable, 5=a lot
N=596

| ***Attitude*** | ***UWA relationship*** | | ***KNP as neighbour to you*** | | ***You as neighbour to KNP*** | |
|---|---|---|---|---|---|---|
| Mean Attitude Rating | 3.96 | | 3.62 | | 4.26 | |
| ***Perceived Benefit of:*** | ***Gamma*** | ***p*** | ***Gamma*** | ***p*** | ***Gamma*** | ***p*** |
| Ecosystem services | 0.478 | <0.001 | 0.480 | <0.001 | 0.595 | <0.001 |
| Park-Based Employment | -0.075 | 0.281 | -0.051 | 0.434 | -0.021 | 0.787 |
| Revenue Sharing | -0.113 | 0.092 | 0.041 | 0.531 | -0.123 | 0.107 |
| NGO Benefit | -0.276 | <0.001 | -0.116 | 0.098 | -0.255 | 0.002 |
| Illegal Extraction | -0.483 | <0.001 | -0.291 | <0.001 | -0.421 | <0.001 |
| Resource Access | -0.141 | 0.211 | 0.015 | 0.886 | -0.033 | 0.796 |
| Living next to KNP | 0.013 | 0.802 | 0.158 | 0.003 | -0.052 | 0.387 |
| ***Perceived Loss of:*** | ***Gamma*** | ***p*** | ***Gamma*** | ***p*** | ***Gamma*** | ***p*** |
| Crop Raiding | -0.018 | 0.761 | -0.204 | <0.001 | 0.088 | 0.196 |
| Lost Access to Resources | 0.064 | 0.248 | 0.097 | 0.068 | 0.101 | 0.097 |
| Predation | -0.162 | 0.002 | -0.086 | 0.104 | -0.089 | 0.144 |
| Injury from Wild Animals | -0.404 | <0.001 | -0.269 | <0.001 | -0.062 | 0.472 |
| Fines & Arrests by UWA | -0.287 | 0.051 | -0.202 | 0.182 | -0.238 | 0.135 |
| Living next to KNP | -0.021 | 0.706 | -0.188 | <0.001 | 0.115 | 0.059 |

Attitudes toward UWA and the park were not significantly associated with the perceived loss of access to resources from the park (Table 7-4); suggesting lost opportunity costs do not strongly influence local people's perceptions about KNP. Also, with the exception of ecosystem services, perceived benefits did not improve household attitudes toward UWA or KNP. Conversely, perceived losses did negatively affect attitudes towards UWA and the park, suggesting losses are shaping people's attitudes.

## 7.4 Discussion

Some villages did benefit from the existence of the park while other villages experienced a net loss. This spatial inequity of benefit and loss distribution has implications for conservation research, as conclusions drawn about the interaction between PAs and local livelihoods could be different depending on whether the benefits in a village are larger or smaller than the losses. The greatest losses accruing as a result of the park occurred within 0.5 km of the park boundary, and yet benefit accrual extended up to 15 km from the park. Therefore, at the scale of the PA, those living directly next to the park are disproportionately carrying the costs of conservation relative to those living only slightly farther away; analogous to PAs benefiting the global community while costing the local community (Wells, 1992), but at a much smaller scale. Organizations having discretion over the location of benefit provision, such as UWA and local governments in the case of revenue sharing projects, and NGOs in the case of development activities, need to consider focusing their benefits closer to the park boundary to support those who lose most from the existence of the park.

Perceptions about living next to KNP were biased toward losses, even in villages with a net benefit. Net benefit may be overstated since opportunity costs were not included. However, given the tendency for loss ratings to be high even when actual losses were relatively low, and the lack of association between the perceived loss of living next to KNP and the loss of access to park resources, the dominance of perceived loss is more likely an example of loss aversion, where the disutility of losing something is valued higher than the utility of getting something of equal value (Thaler *et al*., 1992). This

suggests that benefits need to be disproportionately higher than losses to influence conservation behaviour. Therefore, improving the perceptions of local people about the park would be better served by mitigating losses, rather than providing benefits.

The local benefit most often associated with a PA is the preservation of ecosystem services (Millennium Ecosystem Assessment, 2005). Appreciation for ecosystem services was high in all villages and strongly associated with positive attitudes toward having the park as a neighbour. However, the household perceived benefit of ecosystem services was not related to the perceived benefit of living next to KNP. Ecosystem services tend to be social benefits, and although appreciated, are not easily valued at the household scale (Dixon & Sherman, 1991). Conversely, employment and RAAs were aligned with the perceived benefit of living next to the park, but these are private benefits, easily accounted as financial returns to the household.

The introduction of incentives in the form of financial or service benefits is done to "alter peoples' perceptions of what behaviour is in their self-interest" (McNeely, 1988, p. 125). This strategy is based upon the assumption that the individual will do whatever maximizes their own profit and that reciprocity is contingent upon receiving benefits (Sobel, 2005). However this also dictates that if people lose as a result of the PA they may take resources from the park to balance their losses. Although predation loss did correlate with poaching signs, the far greater crop raiding loss did not correlate with illegal resource extraction, suggesting that retaliation may be limited. The reason for the lack of retaliation for crop raiding losses may be cultural, as African societies show less retaliation in ultimatum and dictator games than western societies (Gowdy *et al*., 2003). The effectiveness of benefits to reduce illegal extraction was also limited. Resource access agreements did appear to reduce illegal entry trails, but these agreements do have reciprocity explicitly built into the contract.

High levels of illegal tree harvesting near villages with significant benefits, primarily from park-based employment, suggest benefits do not improve conservation behaviours. However, other factors may also be influencing this relationship. The villages near park-based employment tended to have high population densities, so demand for resources may be higher in these villages. Firewood is the only fuel source for cooking and heating, most houses are constructed with wooden poles, and much of the local landscape outside the park has been denuded of tree cover (Southworth *et al*., 2010). Therefore a resource market failure exists, because there is a lack of substitutable resource available for the benefiting household to procure, with the park remaining a convenient source for wood. This supports the theory that a resource market failure near PAs can result in higher illegal resource extraction if benefits are provided (Muller & Albers, 2004), because marginal earnings and labour are directed to generating more income rather than time consuming wood gathering farther from the park.

The revenue sharing program did have a positive influence on village-mean attitudes, with villagers looking more favourably upon the park as a neighbour when revenue sharing benefits were high. Also, illegal extraction of resources was lower near villages with elephant trenches, supplied by the revenue sharing program to help protect crops against park-protected animals. In one village a focus group participant said; "It is our duty to protect the park if UWA gives us money for the trench" (29/6/08). Therefore this benefit shows promise as a strategy to improve conservation attitudes and behaviours as long as monetary values increase and project choice is directed towards mitigating losses.

Compensation has been recommended as a means to redress the losses carried by local people as a result of the existence of PAs (World Bank, 2001; Cernea & Schmidt-Soltau, 2006; Adams & Hutton, 2007; Laudati, 2010), however, it may not be a sustainable solution to redress losses accrued due to crop raiding and predation. Most wildlife authorities in developing countries do not have the funds to cover crop raiding and predation losses (Tchamba, 1996; Weber *et al*., 2007), and compensation programs require additional transactions costs to cover program management to guard against fraudulent claims and moral hazard (Bulte & Rondeau, 2005), and to confirm the damage was caused by park-protected animals (Nyhus *et al*., 2005). In many African countries, recovering animal populations and increased human settlement along the borders of PAs is leading to more crop raiding (Tweheyo *et al*., 2005; Weber *et al*., 2007), inflating the cost of any proposed compensation scheme. Although mitigating losses through the implementation of crop raiding and predation defences would incur up-front

and maintenance costs, I believe this presents a more sustainable option; an option repeatedly requested by focus group participants.

## 7.5 Conclusions

Village benefits and losses accrued as a result of the existence of KNP were spatially inequitably distributed with eight villages benefitting but many more accruing a net loss. Losses were highest within 0.5 km from the park, while benefit distribution extended up to 15 km from the park, requiring the provision of benefits, where possible, to be better targeted to households closer to the park.

Household perceived benefit of living next to KNP was driven by legal resource access and park-based employment, while perceived loss was driven by crop raiding. Overall attitudes towards the park appear to be shaped by loss aversion. To improve perceptions about the park, conservation funds would be better spent on projects to mitigate crop raiding and predation losses than on providing conservation benefits through community development projects; although development projects may still be justified for poverty alleviation. To this end, revenue sharing funds should continue to be focused on crop loss mitigation and additional funds should be raised by the conservation community to support these loss mitigation efforts.

Village-valued benefits did not appear to reduce illegal extraction from the park with the exception of resource access agreements, and the revenue sharing program, particularly when money was directed towards crop raiding defences. Retaliation for losses through extraction of resources from the park appears to be limited. Other factors may also be influencing these results, so a more complex analysis that includes population density, household wealth, education, market access and potential market failure, is recommended.

While the global community benefit from the continued protection of biodiversity in PAs, the communities living next to these PAs do incur losses. To address this inequity, more inventories of financial and social costs and benefits of PAs are needed. These inventories can support the development of conservation strategies to minimize the burden of PAs on local communities while preserving biodiversity.

# 8 Conclusions

The overarching question being answered by this dissertation is: *For the people living in communities that directly border Kibale National Park, Uganda, do benefits and losses, accrued as a result of the existence of the national park, affect their conservation behaviours towards the park?* Statistical relationships have been identified between benefits and losses, accrued as a result of the park, and illegal resource extraction from the park, in prior chapters. But are these relationships still relevant when considered within the context of the socio-economic landscape surrounding Kibale National Park? The first section of this chapter continues the progressive contextualization (Theoretical framework) of the influence of benefits and losses on illegal resource extraction from Kibale National Park, employing structural equation modelling to assess the relational influence of benefits and losses relative to demographic, livelihood, and market factors. A discussion of future research that could be undertaken to further understand conservation benefits and losses follows the analysis. Finally, the conclusions of this dissertation are presented, highlighting the implications of the findings for conservation management.

## 8.1 Benefits & losses in context

The influences of benefits and losses on illegal extraction, found in prior chapters, include:

1. A lower number of illegal park entry trails were found near villages with high valued resource access agreements (RAAs), typically RAAs for beekeeping, and villages with beekeeping RRAs also exhibited lower illegal tree harvesting within KNP.
2. Agreements for legal tree harvesting inside the park were linked to higher levels of illegal tree harvesting.
3. The village chairperson's perceived benefit of the revenue sharing program was correlated with lower levels of human disturbance inside the park, specifically with lower levels of in-park grazing and poaching.
4. Illegal tree harvesting was higher near villages with higher village-valued income from park-based enterprises, such as tourism, research, and tree planting for carbon sequestration. Specifically, villages closer to the FACE carbon sequestration operations, had higher illegal harvesting of exotic tree species.
5. Higher village-valued NGO activities were also correlated with high illegal tree harvesting, although this may be due to the NGOs typically being in the same locations as park-based employment.
6. Losses to crop raiding dominated people's attitudes towards the park, but no linkage with illegal extraction was found. Although few animal poaching signs were found near villages, the number of poaching signs did correlate with high village-valued predation losses, suggesting people may take animals from the park in compensation for livestock losses.

However, socio-economic factors were also found to influence illegal resource extraction from the park. In Chapter 3 the productive practices of the local communities, were linked to illegal extraction and in Chapter 4, urban centre demand for resources, the number of households in close proximity to the park boundary, adult education levels, employment rates, and household wealth were found to be predictors of illegal resource extraction from KNP (Table 4-3). Therefore, the influence of benefits and losses, accrued as a result of the existence of KNP, need to be assessed in context, including benefits, losses, demographic, livelihood, and market variables. Structural Equation Modelling (SEM), as described in Section 2.2.2, is used to model this complex interaction.

Data was collected for 25 villages, so the number of variables included in the analysis had to be less than 25 to avoid over identification of the model. However, 28 potential predictor variables have been developed in this dissertation. One approach would have been to include only variables that significantly correlated with illegal extraction in prior chapters. However, there is a growing trend to include non-significantly correlated variables in regression analysis so that size effects can be interpreted (Cohen, 1994; Keith, 2006). Since the focus of the research question is the influence of benefits and losses, the four village-valued benefits (park-based employment, revenue sharing, RAAs, and NGO activities), the three village-valued losses (crop raiding, predation, and UWA fines), and the village chairperson's perceived benefit of

the revenue sharing program were included as modelling variables. All of the demographic and socio-economic variables used to model illegal resource extraction in Chapter 4 were included, except the village-mean number of income generating activities per household, as the findings from Chapter 3 indicate that certain income generating activities might influence illegal resource extraction more than others. Data had been collected on ten income generating activities, however, inclusion of all these activities as model variables would have lead to an over identified model. Therefore, income generating activities were chosen based on being a prevalent activity (growing cash crops), being linked to tree harvesting (percent of households selling firewood, making charcoal and owning woodlots), and potentially being linked to harvesting grasses from KNP for craft production (percent of households selling to tourism).

These 23 variables were used to create SEM models for the number of trees illegally harvested/km of boundary, the number of illegal entry trails into the park/km of boundary, and the number of livestock seen grazing inside the park. Predictor variables were down-selected, by backward exclusion within a linear regression model ($p<0.1$), and finally by choosing the best SEM model based on goodness of fit (Fig. 2-4). A viable model for the number of poaching signs found could not be created due to poaching signs only being found adjacent to six villages.

### 8.1.1 Illegal tree harvesting

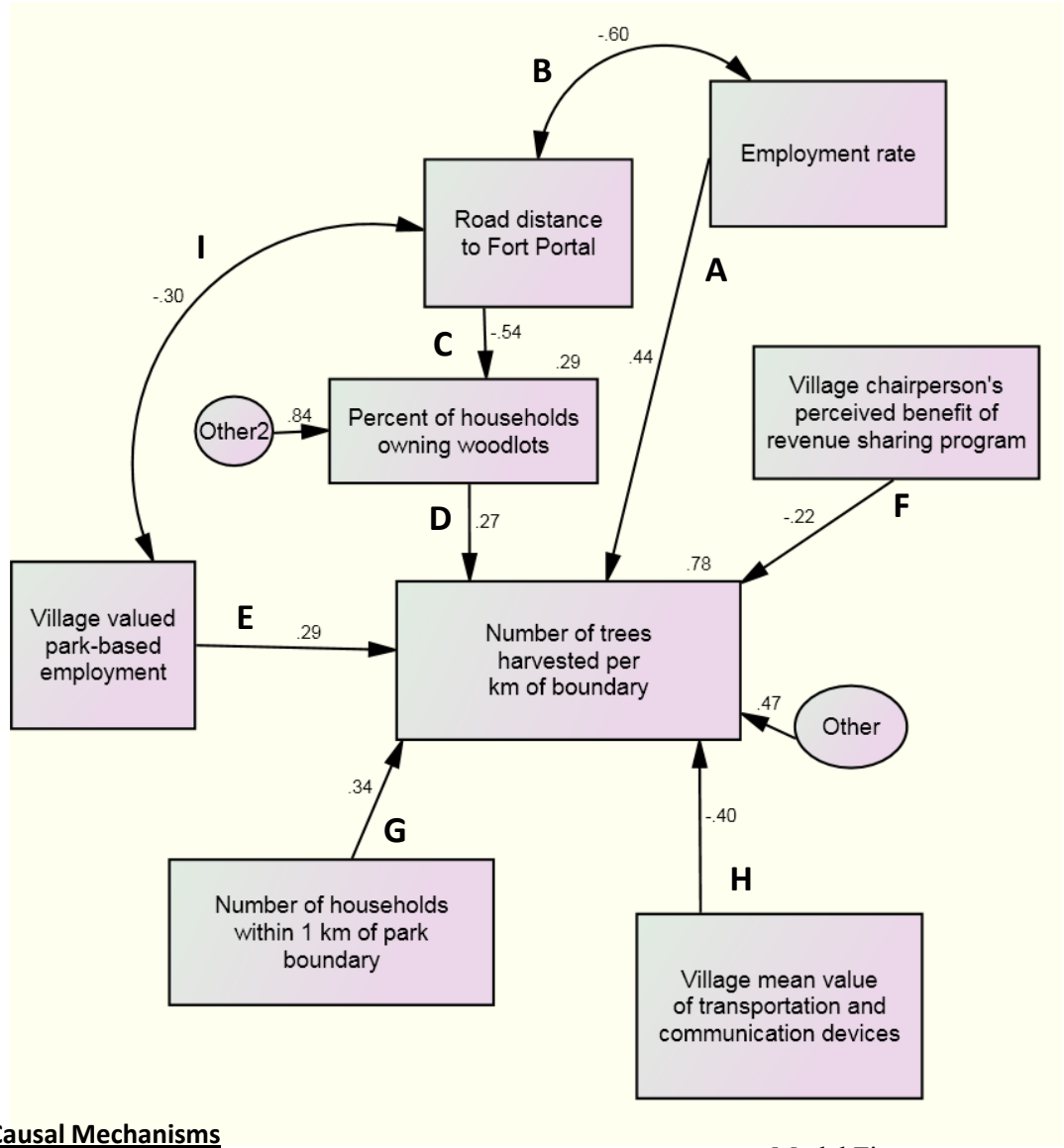
Nine variables (Table 8-1) were down selected by backward elimination linear regression for illegal tree harvesting (Adjusted $R^2$= 0.773, F=10.08, $p<0.001$). Since the percent of households owing woodlots and the village employment rate were found in Chapter 4 to be strongly linked to urban demand, the road distance to Fort Portal was added to the SEM model as an intervening variable (Keith, 2006), but not directly linked to illegal tree harvesting (Fig. 8-1) as it had been eliminated from the down-selected variable list.

The final tree harvesting SEM model ($R^2$=0.78), exhibited a very good goodness of fit, included six direct causal variables and one intervening variable, road distance to Fort Portal (Fig. 8-1). Four direct causal variables contributed to higher illegal tree harvesting: Employment rate, the number of households within 1 km of the park boundary, the village-value of park-based employment, and the percent of households owning woodlots. Household members being employed, either in a park-based enterprise or elsewhere, means that labour within the household has to be redistributed to cover the household tasks. This may result in less time to search for firewood and construction materials, making the park an attractive source for wood. For households close to the boundary, the park is the most convenient source for fuelwood and construction poles. Park-based employment as research assistants, tourist guides, or FACE tree planters provides access to the park, which may facilitate illegal tree harvesting. Higher woodlot ownership indicates people are planting trees in villages with higher local and urban demand for wood, but woodlots are not sufficiently mitigating extraction from the park.

**Table 8-1: Down-selected variables to start path analysis for illegal tree harvesting**

| *Variable* | *Standardized $\beta$* | *Significance* | *Potential Causal Mechanism* |
|---|---|---|---|
| Transportation & communication devices | -0.467 | <0.001 | Wealthy can afford to buy wood, or may have own trees |
| Employment rate | 0.542 | 0.001 | Less household labour time to search for wood |
| Percent of households selling firewood | -0.317 | 0.047 | Supplement supply, but may sell deadwood (collected not cut) |
| Percent of households owning woodlots | 0.463 | 0.005 | Supplement supply/sales for urban demand |
| Percent of households growing cash crops | -0.364 | 0.019 | Can afford to buy wood & may have trees on their land |
| Number of households within 1 km of park | 0.370 | 0.002 | Local demand & opportunity |
| Village chairperson's perceived benefit of revenue sharing | -0.327 | 0.008 | Reciprocity for receiving revenue sharing project |
| Village-value of park-based employment | 0.371 | 0.010 | Demand, less labour to search for wood, and park access |
| Village-value of resource access agreements | 0.218 | 0.077 | Contractual reciprocity, but also have authorized access |

Two variables mitigated illegal tree harvesting: Household wealth, as represented by ownership of bicycles, motorcycles, cell phones, and radios, and the village chairperson's perceived benefit of the revenue sharing program. Wealthy households may not need to extract trees from the park, as they can afford to buy wood, or may have trees on their own land. If the village chairperson perceives the village has benefitted from revenue sharing, they appear to be reciprocating by encouraging their village members to refrain from extracting resources from the park.




**Causal Mechanisms**

**A: Household has less labor to search for wood**
**B: More jobs near urban centre**
**C: Urban demand for wood**
**D: Supplement supply/sales**
**E: Demand, less labor to search for wood & park access**
**F: Reciprocity**
**G: Local demand & opportunity**
**H: Wealthy can afford to buy wood or have own trees**
**I: Park-based employment on major road to Fort Portal**

Model Fit
$X^2$=11.566, p=0.903
df=19
GFI=0.900
CFI=1.000
TLI=1.263
RMSEA<0.001
Model assessment:
**VERY GOOD**

**Figure 8-1: Path analysis for illegal tree harvesting in Kibale National Park, Uganda**

Therefore, the contextual influences of benefits and losses on illegal tree extraction are inconsistent. The revenue sharing program limits extraction, if the program delivers projects that the village chairperson perceives as being beneficial to the village. However, park-based employment seems to increase the extraction of wood from KNP.

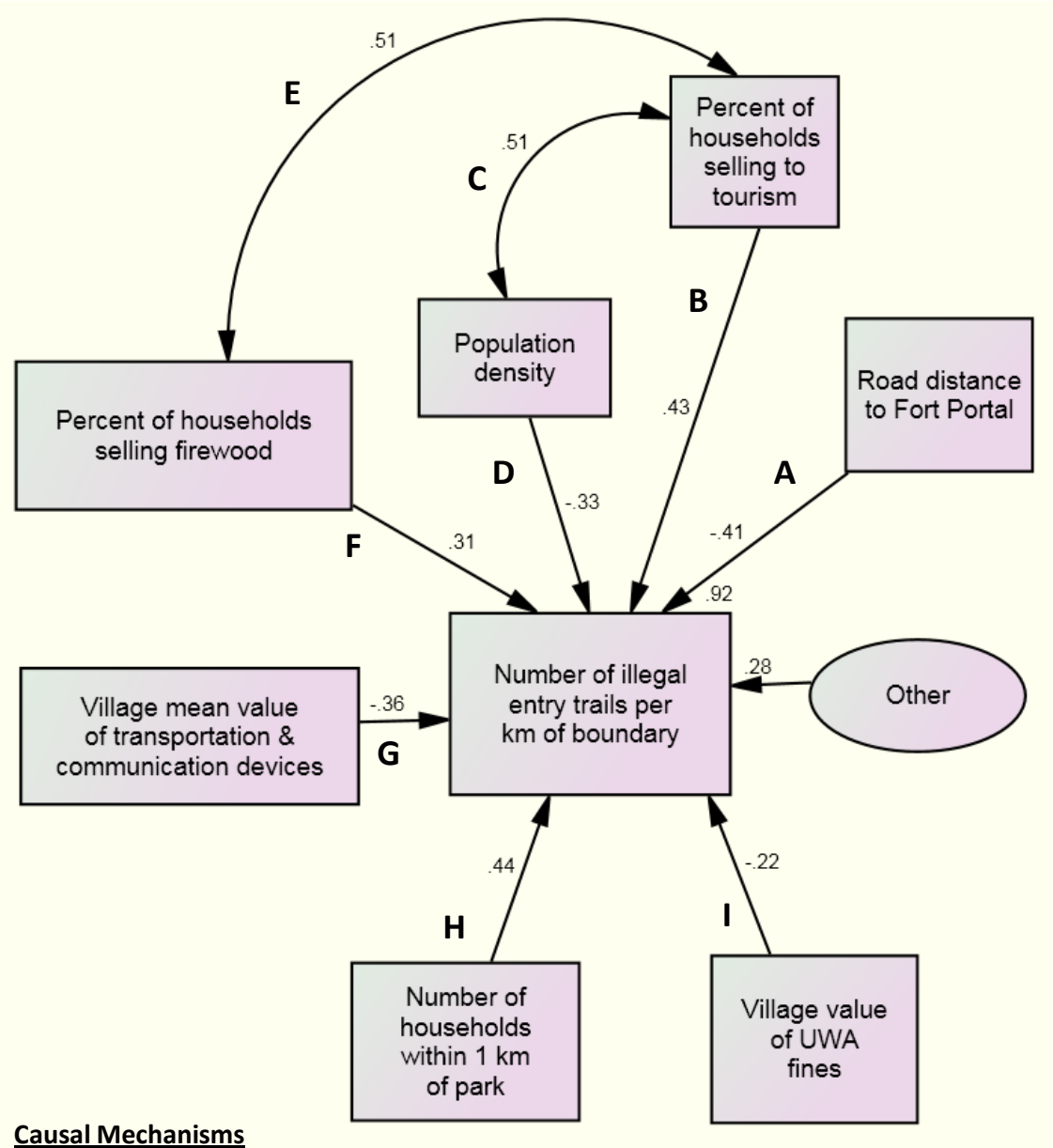
### 8.1.2 Illegal entry trails

Nine variables were down selected by backward elimination in the regression model for illegal entry trails (Adjusted $R^2$= 0.896, F=23.90, p<0.001). However, a prior regression model in the backward elimination iterations, that included these nine variables and five additional variables, had a better adjusted $R^2$ (0.941) and F-test result (F=28.47, p<0.001), so 14 variables were included in the first SEM iteration (Table 8-2).

The best SEM fit was found after removing seven of the 14 variables ($R^2$=0.92), leaving seven direct causal variables in the model (Fig. 8-2). Four variables contributed to a higher number of illegal entry trails into the park: shorter road distance to Fort Portal, the percent of households selling firewood and selling to tourism, and the number of households within 1 km of the park. More illegal entry trails in villages near Fort Portal may be indicative of the urban demand for park resources, while households close to the park may represent more localized demand, coupled with convenient, opportunistic access. Firewood sellers, often sell fallen deadwood that they have collected, so firewood sellers are more likely to be using the entry trails to look for deadwood in the park, while women who sell crafts to tourists may need to get their basket-making materials from inside KNP.

**Table 8-2: Down-select variables to start path analysis for illegal entry trails**

| *Variable* | *Standardized $\beta$* | *Significance* | *Potential Causal Mechanism* |
|---|---|---|---|
| Road distance to Fort Portal | -0.719 | <0.001 | Demand for resources from urban centre |
| Population density | -0.465 | <0.001 | Demand for resources |
| Village-mean years of education: Adults | 0.217 | 0.024 | More education leads to more wealth to buy resources |
| Transportation & communication devices | -0.482 | <0.001 | Wealthy can afford to buy resources |
| Employment rate | -0.243 | 0.027 | Less dependent on forest resources |
| Percent of households selling firewood | 0.274 | 0.022 | Enter park to search for deadwood |
| Percent of households growing cash crops | -0.256 | 0.022 | Wealthy can afford to buy resources |
| Percent of households selling to tourism | 0.523 | 0.001 | Enter park for craft & fuel materials |
| Number of households within 1 km of park | 0.400 | <0.001 | Local demand & opportunity |
| Village chairperson's perceived benefit of revenue sharing | 0.108 | 0.190 | Reciprocity for receiving revenue sharing project |
| Village-value of losses to crop raiding | -0.120 | 0.057 | Fear of park animals |
| Village-value of losses to predation | 0.141 | 0.070 | Taking their own compensation from the park |
| Village-value of UWA fines for being in park | -0.239 | 0.007 | Fines deter entry into park |




**Causal Mechanisms**

**A: Urban demand for park resources**
**B: Gather craft supplies from park**
**C: Population density higher near tourist facilities**
**D:Demand for resources**
**E: Sell both firewood & crafts**
**F: Collect dead wood along trails for sale**
**G: Wealthy can afford to buy resources**
**H: Local demand & opportunity**
**I: UWA fines deter park entry**

Model Fit
$X^2$=20.868, p=0.344
df=19
GFI=0.825
CFI=0.974
TLI=0.961
RMSEA=0.064
Model assessment:
**MARGINALLY GOOD**

**Figure 8-2: Path analysis for illegal entry trails into Kibale National Park, Uganda**

Three variables contributed to lower numbers of illegal entry trails into the park: household wealth as represented by the ownership of bicycles, motorcycles, cell phones and radios, the village-value of fines received

from UWA for illegal entry and resource extraction from the park, and village population density, when households engaged in tourism sales. Wealthy households can possibly afford to buy the resources they need, and UWA fines are intended to deter people from illegally entering the park. The reason for villages with higher population density to have fewer illegal entry trails is less clear, as more people should result in higher demand for resources from the park and hence more entry trails. However, the linkage between population density and tourism sales may indicate that tourism is helping to mitigate illegal resource extraction from the park, although some villagers do still enter the park for resources to make crafts to sell to tourists, or to collect deadwood for tourism facilities.

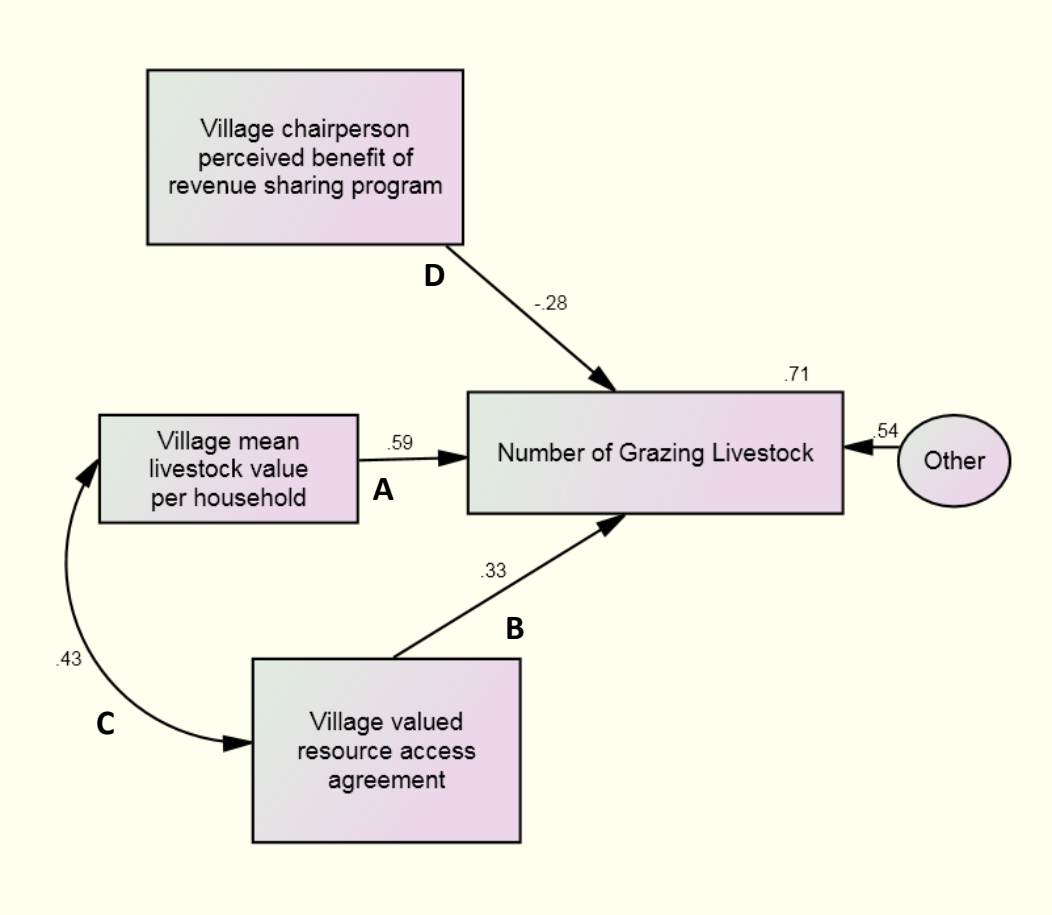
The driving factors influencing the number of illegal entry trails into the park appear to be demand and proximity. The influence of village-valued loss to UWA fines indicates that fines are an effective deterrent to illegal activity inside the park. However, no village benefits were specifically reducing illegal entry into the park. The existence of a tourism market may be increasing illegal entry, as people search for materials to convert into crafts to sell to tourists, but may also have a mitigating influence on illegal park entry in more populated villages where one might expect the demand for park resources to be higher.

### 8.1.3 Illegal in-park grazing

Eight variables (Table 8-3) were down selected by backward elimination in the regression model for illegal in-park grazing (Adjusted $R^2$= 0.783, F=11.82, p<0.001). The final in-park grazing SEM model ($R^2$=0.71), exhibited a very good goodness of fit, but included only three direct causal variables (Fig. 8-3). Two direct causal variables contributed to higher illegal in-park grazing: the value of livestock owned, and the village-value of RAAs. As people own more cattle, domestic animals are being grazed inside the park. The influence of RAAs to increase in-park grazing was surprising, but might be explained if profits from RAAs are being reinvested in livestock ownership, as suggested by the strong covariance between RAA value and livestock ownership in the SEM model (Fig. 8-3, path C). Once again, the influence of the village chairperson's perceived benefit of the revenue sharing program provided a mitigating influence for in-park grazing.

**Table 8-3: Down-select variables to start path analysis for illegal in-park grazing**

| *Variable* | *Standardized $\beta$* | *Significance* | *Potential Causal Mechanism* |
|---|---|---|---|
| Village-mean years of education: Adults | 0.425 | 0.018 | More education leads to more wealth to buy livestock |
| Village-mean land owned | -0.316 | 0.048 | Use their own land for grazing |
| Village-mean livestock wealth (CEU) | 0.477 | 0.001 | More livestock, more demand for grazing |
| Percent of households owning woodlots | 0.429 | 0.019 | Putting land into woodlots not grazing pasture |
| Percent of households growing cash crops | -0.586 | 0.001 | Cash crop growers tend to own less livestock |
| Village chairperson's perceived benefit of revenue sharing | -0.456 | 0.001 | Reciprocity for receiving revenue sharing project |
| Village-value of resource access agreements | 0.480 | 0.002 | Contractual reciprocity, but may put profits into buying livestock |
| Village-value of NGO benefits | -0.449 | 0.022 | Contact with NGO informs people about conservation |




**Causal Mechanisms**
**A: More livestock, more demand**
**B: Contractual reciprocity & access**
**C: Invest RAA profits in livestock**
**D: Reciprocity**

Model Fit
$X^2$=1.080, p=0.583
df=2
GFI=0.978
CFI=1.000
TLI=1.102
RMSEA<0.001
Model assessment:
**VERY GOOD**

**Figure 8-3: Path analysis for illegal in-park grazing in Kibale National Park, Uganda**

## 8.2 The next contextual layer

The contextual analysis suggests that the demand for wood to supply study villages and Fort Portal is, at least partially, being met by illegal extraction of trees from KNP. However, demand is only half of the economic equation required to understand the wood market around KNP and the need for illegal harvesting of trees from the park. The other factor is supply and is most likely the most influential missing variable from the tree harvesting SEM model. Lack of woody biomass outside the park could be a reason for local communities to harvest trees, either for home consumption or for sale. Therefore, the tree cover outside the park, in proximity to the study villages, needs to be characterized and quantified. This could be achieved by classifying satellite imagery, preferably collected in 2008, to quantify the area of land outside the park covered by forest fragments and woodlots.

This classification could also be combined with the classification of satellite images from other years (satellite images between 1973 and 2008 are available), to study the rate of tree cover decline outside the park (Hartter & Southworth, 2009). Understanding the relative area of tree cover extending from Fort Portal over time could determine the rate of deforestation and conversion of land to woodlots emanating from the urban centre, helping to predict the spatial progression of the threat of urban demand on the forest habitat of KNP in the future.

The in-park grazing path model indicates that 29% of the variance is not explained by the model. The availability of grazing pasture outside the park may be an important variable that could also be investigated using classification of satellite imagery.

Finally, classification of a 2008 satellite image could support the development of crop raiding defences. Inverse Distance Weighted (IDW) interpolation was used to rasterize household survey data to visualize patterns of crop raiding for all of the study villages (Fig. 8-4a). Raster mapping of the fraction of staple crop production lost to wild animal crop raiding, as estimated by the household survey respondents, indicated areas of the study villages more prone to crop raiding than others. A comparison of these patterns with a Digital Elevation Model (DEM; CGAIR-CSI, 2011) for each village (Fig. 8-4b) suggests that households near low lying areas that intersect the park boundary seem to be more susceptible to crop raiding loss. Coupling this spatial visualization of crop raiding with a land cover map and the DEM could identify specific landscapes that are more prone to crop raiding. This information could then be used to preferentially target crop raiding defences to areas that match the highest risk landscape attributes, helping to mitigate the losses accrued by local people around KNP.

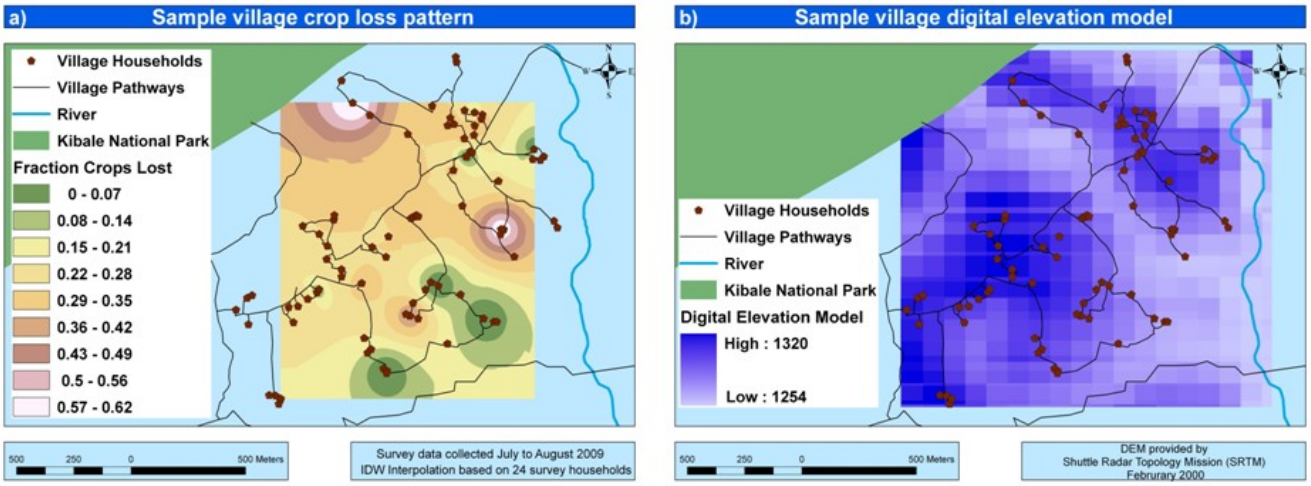



**Figure 8-4: Crop raiding patterns linked to elevation**

## 8.3 Another scale of analysis

This dissertation has analyzed the relationship between PA benefits and losses, and illegal extraction of PA resources at the village scale. Analysis at the village scale treats the neighbouring communities as an aggregate threat, masking individual motivations to enter the park, and statistically removes variance from household survey data resulting in a simplified explanation of the drivers of illegal extraction; demand and proximity, primarily mitigated by wealth and the revenue sharing program. In the future, I propose to reanalyze the data at the household scale using the sustainable livelihoods framework (DFID, 1999) to investigate the interaction between household well-being, park-based benefits and losses, and household admitted illegal resource extraction from KNP.

Centred on the capital assets of the rural household, the sustainable livelihoods framework examines interactions between the household, formal institutional processes and vulnerability shocks that result in specific livelihood strategies being implemented to create desired livelihood outcomes. Recently, this framework has been used to study whether conservation initiatives support the Millennium Development Goals (WWF, 2006), permitting a balanced assessment of development objectives within a conservation context (Igoe, 2006; Bennett, 2010). To date, use of the livelihoods framework has focused on community livelihood outcomes with little attention to conservation objectives. By including the benefits and losses accrued by the household as a result of the existence of the park, and focusing on the stated desire to access park resources, and the admitted illegal access to resources in the park, the sustainable livelihoods framework could be used to investigate the factors that influence rural households to engage in the livelihood strategy of illegal resource extraction from KNP.

## 8.4 Implications for conservation

### 8.4.1 Benefits

Four valued benefits were assessed at the village scale by this dissertation: revenue sharing, resource access agreements, park-based employment, and NGO activities. All four benefit mechanisms do contribute to development in villages adjacent to KNP. However, the effectiveness of these benefits to reduce illegal extraction from the park was found to be variable and sometimes even contrary to conservation objectives.

The revenue sharing program around KNP does provide benefit to the local people and improves conservation behaviours if the projects specifically deal with the villagers' primary problem of crop raiding by park-protected animals. However, generally revenue sharing money is considered by villagers and local government representatives to be too small to have a significant effect on illegal resource extraction. Since crop raiding is worse for households within 0.5 km of the park boundary, revenue sharing projects should be targeted to households within this distance from the park, rather than the current project distribution of up to seven kilometres from the park boundary. Extraction of park resources was lower for villages where revenue sharing money had been spent on elephant trenches; the only type of project preferentially desired by villagers, and showing significantly higher perceived benefit of the program by households and village chairpersons. The perceived benefit of revenue sharing by village chairpersons remained a significant predictor of lower resource extraction, even when all other influences were considered, specifically mitigating illegal tree extraction (Fig. 8-1) and in-park grazing (Fig. 8-3), presumably through the chairperson's ability to influence village members to reciprocate the benefit of the revenue sharing program by staying out of the park. The influence of the village chairperson on actions of individuals in the village needs more study, but my findings imply that a closer relationship between UWA and the village chairpersons might lead to improved local advocacy for conservation in villages near KNP. The revenue sharing program is supporting the conservation objectives of KNP, and is most effective when the money is targeted to mitigating crop losses.

RAAs significantly contributed to the village-valued benefit accrued from the park by a study village, especially in villages with lucrative beekeeping RAAs. However, the influence of RAAs on illegal extraction from the park was mixed. The volume of wood extracted from the park was lower near beekeeping RAA villages, and villages with higher valued RAAs tended to have fewer illegal entry trails into the park. However, villages with

agreements to legally extract trees from the park had higher illegal tree extraction. When all contextual factors influencing illegal off-take were included, the village-value of RAAs did not significantly mitigate illegal extraction of trees (Table 8-1), and legal trails made for RAA members may have provided access for others to extract resources from the park. It also appears that profits from RAAs may be invested in livestock ownership, further increasing in-park grazing (Fig. 8-3). Although RAAs significantly contribute to the household perceived benefit of living next to KNP (Table 7-1), the implementation of RAAs with local community associations is a marginally effective conservation strategy, dependent on the type of resource accessed, and requiring UWA to accept the potential risk of increased illegal resource extraction.

The existence of KNP was responsible for over 700 jobs for local people, a substantial number when the average village overall employment rate is only 22%. Village population density tended to be higher in villages near park-based enterprises. Park-based employment was the largest contributor to the village-value of park benefits, and significantly contributed to the household perceived benefit of living next to KNP (Table 7-1). However, villages with higher values of park-based employment income also had more illegal tree harvesting, an influence that remained significant even when all contextual variables were considered (Fig. 8-1). Therefore, UWA needs to consider the potential migratory pull of park-based enterprises, and plan to provide alternative sources of wood, such as woodlots to be directly used for fuelwood in the home, for communities adjacent to these operations, as high park-based employment appears to increase illegal tree harvesting which is counter to conservation objectives.

NGO activities tended to be located close to park-based employment or near main roads. NGO activities had a small, but significant influence on the perceived benefit of living next to KNP (Table 7-1), tended to mitigate in-park grazing (Table 8-3), and strengthened people's perception of how their activities in the park could be counter to conservation objectives (Table 7-4). Villages with higher values of NGO activities also had more illegal tree harvesting, however this is probably due to being located near park-based enterprises, since village-valued NGO benefit was not a predicting variable in the SEM tree harvesting model. It would appear that NGOs are supporting conservation objectives by having a positive effect on local attitudes towards the park, but a fairly weak influence on illegal activity in the park.

### 8.4.2 Losses

Three valued losses were assessed at the village scale by this dissertation: crop raiding, predation, and UWA fines for illegal park entry and extraction of park resources. Village-valued losses tended to be larger than benefits in most villages (17 of 25 villages), consistent with focus group feedback indicating that losses generally out-weighed the benefits accrued from living next to KNP.

The largest accrued loss due to the existence of KNP was crop raiding, and in comparison with prior studies, crop raiding appears to be getting worse as conservation initiatives lead to the recovery of animal populations and human population growth, including in-migration in search of land, increases agriculture along the park boundary. The average annual value of crop raiding losses to households represented 3% of the median household capital asset wealth. Crop raided households were more prone to food insecurity, self-reported more malaria and other human and livestock diseases, were more prone to injury from wild animals while guarding their crops, and children from villages that were very troubled by crop raiding had lower scholastic achievement, most likely due to being held back from school to guard crops.

Crop raiding by park protected animals dominated peoples' attitudes towards the park (Table 7-2), and tended to override any benefits that the village had received from the park. However, the village-valued loss to crop raiding was not a predicting variable in any of the illegal resource extraction models, suggesting people are not taking resources from the park as compensation for their losses. Most focus group respondents felt the benefits did not sufficiently compensate for crop raiding losses, requesting direct cash compensation for their losses. However, given UWA does not have the money for a compensation program, and that crop raiding is getting worse around KNP, a more sustainable solution would be to fund the development of better crop raiding defences. Mitigating crop losses, and providing development programs to deal with the social costs of crop raiding (*e.g*. aid for crop raiding induced food insecurity, tutoring for children who have to guard crops instead of going to school, and malaria prevention strategies for people who have to guard crops at night), would support conservation

objectives by improving the attitudes of local residents towards the park.

Predation losses were much lower than crop raiding losses, tending to be larger where people owned more livestock. Village-valued predation losses were correlated to higher numbers of poaching signs inside the boundaries of KNP. However, since poaching signs were only found near a few villages, the influence of predation on poaching in a contextual model could not be investigated. Although statistically weak, the correlation between predation losses and poaching signs may suggest people could be extracting their own compensation for these losses from the park; however, this needs further investigation. Revenue sharing projects in the higher predation areas have started to include the provision of goats and pigs to villages near the park, however, this does not ensure that the households that actually lost livestock are the ones who receive revenue sharing project livestock. The potential linkage between predation and poaching requires further study to determine an effective conservation strategy.

Very few households reported receiving UWA fines for illegal park access and resource extraction, but the prevalence of fines was higher within 8 km of a UWA outpost. Although UWA fines did not appear to influence attitudes towards KNP (Tables 7-2 & 7-4), UWA fines were effective at reducing the number of illegal entry trails into the park (Fig. 8-2). Fines are therefore seen as an effective conservation strategy to reduce illegal activity in KNP.

### 8.4.3 Spatial distribution of benefits & losses

Benefit distribution around the periphery of the park tended to be localized with the exception of revenue sharing projects which were located in most parishes around the park. Resource access agreements tended to be in the southern half of the park with a majority in Kamwenge district. Since most RAAs were for beekeeping, this concentration of agreements in Kamwenge is probably linked to the development of a honey processing and distribution plant in Kamwenge Town. Park-based employment was concentrated near the park-based enterprises offering the employment opportunities, while NGO activities tended to be along the north western and central eastern boundaries. Most NGO activities were started by long-term researchers, and they have had a tendency to set-up their NGOs near their research locations, which tend to be near Makerere University Biological Field Station, in the northwest, and Bigodi, next to the central eastern boundary of the park.

Losses were more evenly distributed around the park, although some areas were more prone to crop raiding and predation than others. Crop raiding losses tended to be higher in areas raided by elephants or where baboon raiding was particularly concentrated, and predation losses were higher along the northeast boundary. Many of the villages with the highest losses to crop raiding were villages that benefited from park-based employment and NGO activities. However, more northern and north western villages that experienced high crop raiding and predation losses were accruing minimal benefits from the park. UWA has been counselling local government to use the revenue sharing funds for building elephant trenches and providing livestock to frontline villages. Mitigation of crop raiding and predation losses needs to be the continued focus of UWA and local leadership, with emphasis placed on the locations most troubled by park-protected animals.

Crop raiding and predation losses were higher for households located closer to the park, with 90% of the crop raiding damage occurring within 530 m of the park. Although the majority of losses were accrued within 0.5 km of the park, benefits accrued up to 15 km from the park boundary: park-based employment 15 km, revenue sharing projects 7 km, NGO activity 5.3 km, while most RAA members lived within 5 km of the park. Therefore those living closest to the park are accruing the largest proportion of the losses, while benefits are more diluted by being accrued over a wider area. UWA, in partnership with local leadership, need to consider this spatial inequity, and target benefits to help households located closer to the park. UWA has been asking councils to target revenue sharing projects closer to the park, and this analysis clearly supports this objective.

### 8.4.4 The demand for wood

Indicators of human disturbance in the park were found to cluster and be spatially coherent, identifying hotspots for extraction of particular resources. The most wanted and extracted resource from KNP was wood for fuel and construction. Although the concentration of extraction on exotic and early successional tree species indicates current levels of extraction may be sustainable, increased access for local communities is not recommended

because illegal wood extraction was higher adjacent to villages with agreements to legally extract wood.

It is clear that the socio-economic landscape is shaping illegal harvesting of trees far more than the benefits and losses accrued by local communities (Figures 8-1, 8-2 & 8-3). All cooking and heating around KNP is fuelled by wood collected from areas of brush, small woodlots, village forest patches, and the park (Naughton-Treves *et al*., 2007). Local people also rely on wood for building materials (Hartter, 2010), to make charcoal, and to fuel gin and brick production (Naughton-Treves *et al*., 2007). Given the lack of a local substitute for wood and the potentially high cost of substitutes, the widespread deforestation outside KNP (Hartter & Southworth, 2009), and the demand for wood from local communities and the urban centre of Fort Portal, there is a need to develop strategies to increase wood supply in order to reduce illegal tree harvesting inside KNP. However, since woodlots essentially turn trees into a cash crop, there is a risk that wood produced near KNP could be sold to more profitable markets in the capital, Kampala. Therefore, this research supports the need to incentivize farmers to grow trees specifically for local use, to supply tree seedlings to communities near KNP, and to expand the existing energy-saving stove programs around the park.

## 8.5 Concluding remarks

I have found that providing benefits to local communities can have a positive influence on conservation objectives, but to be effective, the benefits have to be perceived as beneficial by local communities, and need to be targeted closer to the park boundary. In the case of KNP, this requires that the primary problem of crop raiding by park-protected animals be addressed. Losses were affecting attitudes towards the park but, with the possible exception of predation losses, people did not appear to be collecting park resources as retribution for losses accrued due to the existence of the park.

Contrary to conservation objectives, park-based employment was linked to increased illegal tree harvesting, possibly because people working for park-based enterprises had more access to the park, or due to more people and consumption near park-based enterprises, or simply because the park is a convenient place to find wood for busy households with labour engaged outside the home. Generally, the influence of benefits to mitigate illegal resource extraction was small when compared with livelihood needs and market forces. Therefore, strategies to augment supply of park-based resources, especially wood, outside KNP should be pursued in parallel with incentives for conservation, such as the revenue sharing program.

The spatial distribution of benefits did not equitably address the spatial distribution of losses. To address this, strategies to mitigate crop raiding and predation losses need to be the priority. UWA, in partnership with local governments, needs to ensure revenue sharing funds are directed to this end, and the conservation community needs to raise funds to help mitigate financial and social costs, by creating barriers to stop park-protected animals entering neighbouring communities.

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