Measuring the state of health of sacred forests, Kenya

Using the black-and-white Colobus monkey (*Colobus angolensis palliatus*) as an indicator species.

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September 2010

Colobus angolensis palliatus
(Anderson, 2004)

A thesis submitted in partial fulfilment of the requirements for the degree of Master of Science and the Diploma of Imperial College London

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List of acronyms

CBD	Convention on biological diversity
DF	Degrees of freedom
GLM	Generalised linear model
GIS	Geographical information system
GPS	Global positioning system
NMK	National Museums of Kenya
RD	Residual deviance
WWF	World wildlife fund for nature

Abstract

Tropical forests are under direct threats by anthropogenic activities which affect biodiversity, ecosystem services, atmospheric conditions and livelihoods. These threats are prominent in the highly endemic, biodiversity-rich coastal forests in Kenya which are severely fragmented and are declining in health.

This research project investigates the state of health of coastal sacred forests using *Colobus angolensis palliatus* as an indicator species. This data is supplemented by investigating disturbance, forest area and forest perimeter length. The trend in Colobus abundance and forest area can be directly compared to Anderson (2004) to establish changes since 2001. Finally, locally based monitoring was implemented using semi-structured questionnaires to determine whether local communities could successfully identify trends over time in order to establish if locally based monitoring could be used as a technique in the future.

Management status had no effect on Colobus density, disturbance rates, forest area and forest perimeter indicating a need to re-evaluate current management practices. There was no change in Colobus density over the 9 year time period. A higher forest area to forest perimeter ratio, however, resulted in higher densities of Colobus monkeys. Forest areas on average were found to be increasing in size. Finally, local communities did not predict the rate of change in Colobus abundance and forest area change, suggesting the respondents are out of synch with trends in environmental conditions.

This research demonstrates the use and importance of different approaches to monitoring forest health and highlights a need to incorporate local communities into forest conservation. There must be a combination of science, culture, economics, and locally engaged communities to achieve conservation goals.

Keywords: forest health, Colobus angolensis palliatus, forest fragmentation, locally based monitoring.

Word Count: 12,592

Acknowledgements

Firstly I would like to thank my supervisors Marcus Rowcliffe and Neil Burgess for their helpful advice and support throughout the project time. I would also like to like to express my thanks to Julie Anderson for her advice and knowledge of Colobus monkeys and Kenya as well as for supplying me with previous data materials. Also, thanks to Rodolphe Bernard for his GIS expertise

I would also like to express my sincere thanks to the people with me in Kenya: Elias Kimaru who worked relentlessly for WWF and supporting me throughout my stay in Kenya; Hamissi Pakir (bottom right in the picture below) my field guide, whose extensive knowledge of the coastal forests and Colobus identification and behaviour contributed enormously to the development of my field based skills. I would also like to thank Philly Dyson for her support in Kenya and for nursing me back to health after I contracted malaria.

Thanks to my parents for supporting me through my postgraduate study, and finally, a heartfelt thank you to Greg Tomlinson for his emotional support anywhere in the world.

1. Introduction

Globally, forests are the largest and most productive ecosystem, playing a critical role in ecosystem services, biodiversity, culture and human welfare (Costanza *et al.*, 1997, Wright, 2010). However, tropical forests are disappearing faster than any other biome (Myers, 1991); contemporary land use change in tropical forests is about 64, 000 km^2 per year (Wright, 2010). This high rate of forest loss is exacerbating the fragmentation of forests (Fischer and Lindenmayer, 2007). Forest fragmentation can be defined as the conversion of once large continuous blocks of habitat to a less continuous, spatial separation of habitat units (Fischer and Lindenmayer, 2007). This occurs primarily by anthropogenic disturbances, including agricultural land conversions, urbanisation and deforestation (Franklin *et al.*, 2002). The destruction of forests has a range of negative ecological consequences including detrimental effects on species richness (Begon *et al.*, 1990); this loss of biodiversity will, in turn, jeopardise the ability of ecosystems to function adequately (Chapin Iii *et al.*, 2000). Biodiversity loss decreases the resilience of ecosystems to environmental change (Chapin Iii *et al.*, 2000). Furthermore habitat fragmentation can be detrimental to species health. For example many studies provide evidence that primates exhibit more evidence of physiological stress (Dunn *et al.*, 2009, Martínez-Mota *et al.*, 2007) and a loss in genetic diversity (Craul *et al.*, 2009, James *et al.*, 1997) in fragmented landscapes. Landscape modification and habitat fragmentation are consequently major research themes in conservation biology (Fischer and Lindenmayer, 2007, Haila, 2002, Franklin *et al.*, 2002).

The coastal forests of East Africa are small and fragmented and are a prime example of landscape modification. This is especially seen in the Kwale district (3°30′, 4°45′ S; 38°31′ and 39°31′ E) in the Coastal Province of Kenya (Figure 1). These coastal forests are of exceptional importance globally due to their remarkably high level of endemism (Lovett, 1993). Their importance is emphasised by their classification as biodiversity hotspot regions under the criterion set by Myers (1990). However, this forest habitat is declining and being replaced by areas of agricultural land and ever increasing areas of urbanization and associated tourism facilities that follow (WWF-UK, 2005). If these trends continue, there will be further loss of forest cover, biodiversity and related environmental services (water, soil erosion, and loss of land productivity) (Matiku, 2004). This will have a negative impact on livelihoods of neighbouring communities, biodiversity conservation, national and global benefits, and goods and services (Matiku, 2004). Responses of organisms to these changes can provide important information on the viability of global life support systems in the area (Burgess *et al.*, 1998, Noss, 1990).

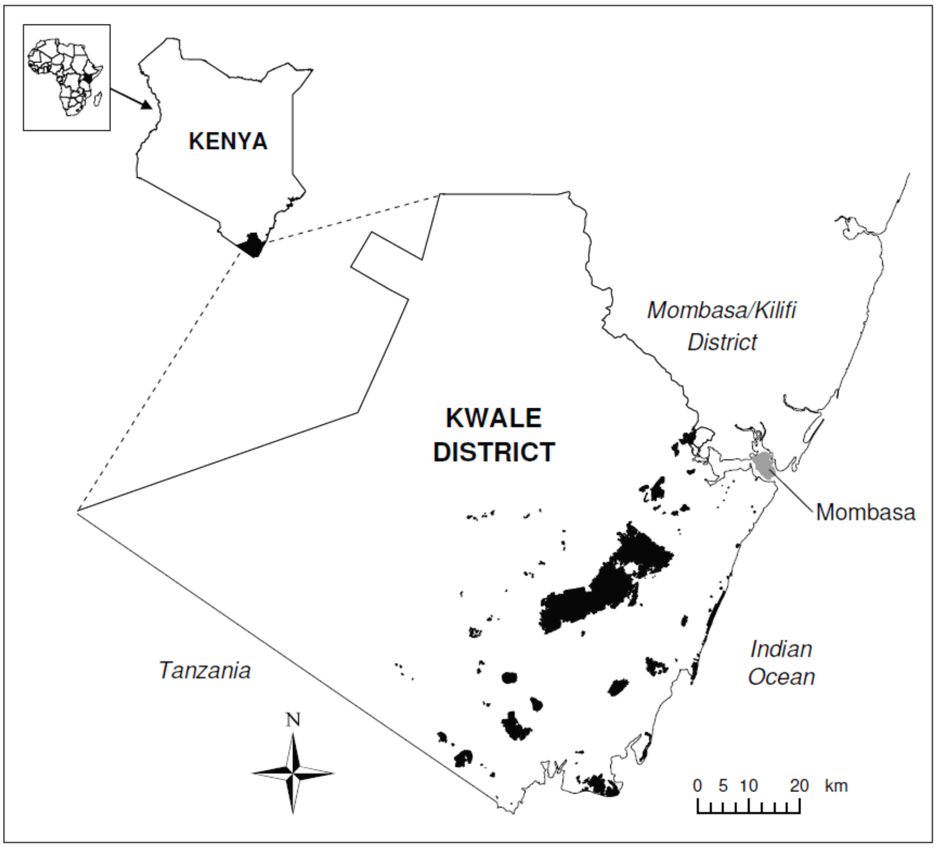


Figure 1: Distribution of coastal forest fragments in Kwale district, Kenya (Anderson et al., 2007a).

The need to conserve the remaining forests is heightened by the unique nature of a particular type of forest occurring in the coastal forest of Kenya: Kaya forests. These distinctive forests are of special socio-cultural significance and owe their existence to local communities as they provide a variety of complex functions: groves for worship, ceremonies, burial grounds and meeting places for special occasions (Tengeza, 2003, Matiku, 2004, Tinga, 2004). This has had significant biodiversity benefits, with social taboos discouraging deforestation and teaching respect for natural resources (Bhagwat and Rutte, 2006). Consequently, Kaya forests are biodiversity rich and globally unique areas (Robertson, 1993) harbouring at least one endemic species per forest (Burgess, 2000). The loss of these forests will have a significant impact on the fabric of Kenya, not only in the detrimental effect on biodiversity, but also on the consequent loss of culture and tradition in Kenya. Currently 42 Kaya forests are under a form of legal protection: they are gazetted as national monuments by the National Museum of Kenya (Anderson, 2004). Kaya conservation complies with article 10(c) of the Convention on Biological Diversity; "Protect and encourage customary use of biological resources in accordance with traditional cultural practices that are compatible with conservation or sustainable use requirements" therefore helping Kenya fulfil national targets (CBD, 2010).

The Angola black-and-white Colobus monkey (*Colobus angolensis palliatus*) is a useful indicator species of forest health for the coastal forests of Kenya as they are highly sensitive to disturbance (Anderson, 2004). Indicator species have been utilised to make fast assessments of ecosystem health and habitat composition (Noss, 1990, Landres *et al.*, 1988).

Colobus angolensis palliatus is a charismatic flagship species and is under direct threat by habitat fragmentation and habitat modification (Anderson *et al.*, 2007b, Anderson *et al.*, 2007a). Anderson *et al.* (2004) showed that the occurrence and abundance of Angola black-and-white Colobus monkeys were significantly influenced by the features of the coastal forests habitats, such as spatial (forest size), resource (tree diversity), structural (canopy cover) and disturbance (forest loss) characteristics (Anderson, 2004). This species is listed on Appendix II of CITES and on class B of the African Convention on the Conservation of Nature and Natural Resources (Anderson, 2004).

Anderson (2004) carried out the first evaluation of Colobus distribution, status and threats in the southern coastal district of Kwale in 2002; Baseline data were collected on Colobus counts and forest size. Forest fragment area was available from 1989 (from maps printed in 1991), as well as in 2001, (during the survey by Anderson). Most of the forests surveyed at this time have since declined in area between 1989 and 2001. A total of 38 out of 124 of the patches suffered between 3-96% decline in forest coverage, while a small number of forest patches increased in size (Anderson, 2004). Furthermore, forests which offered higher protection from anthropogenic exploitation were found to be Kaya forests, significantly more than forest reserves or unprotected forests. This was most likely due to their management status, which highlights the critical need for community involvement in successful conservation management in the coastal forests of Kenya. Recently, locally based monitoring has been developed, which has been described as an accurate, cost effective and adequately powerful alternative to professional monitoring to assess natural resource status (Rist *et al.*, 2010). It is also an effective monitoring technique because it engages the local community and this involvement can help change negative attitudes on sustainable use and the environment (Danielsen *et al.*, 2005).

Since 2002, threats to the coastal forests have increased due to population growth, urbanisation and tourist development (WWF-UK, 2005). This has increased demand for forest resources leading to currently unknown effects on the Kwale landscape.

1.1 Aims and Objectives of this study

With Kenya's coastal forests under severe threat, this study aims to determine the health of the biodiversity-rich and culturally important Kaya forests. More specifically this study has the following objectives:

1. **To quantify the abundance of Colobus monkeys in Kaya forests and to determine if populations have changed since the initial study by Anderson (2007) in 2001**.

2. **To quantify the intensity of anthropogenic disturbances in Kaya forests.**

3. **To assess if Kaya forests have changed in size since previous surveys conducted in 1989 and 2001.**

4. **To determine local perceptions of change in Colobus populations and forest condition in comparison to findings from censes and mapping monitoring methods. This is in order to try and incorporate community involvement in future conservation initiatives.**

5. **To recommend future conservation action to WWF and the National Museums of Kenya.**

It is anticipated that this study will contribute to the future conservation of coastal forests in Kenya, especially through partnership with the WWF and the National Museums of Kenya.

1.2 Hypotheses

This research will test the following hypotheses:

H1: There will be a reduction in forest health in Kaya forests, indicated by reduced frequency of Colobus monkeys, and increases in forest disturbance, indicated by felled or cut trees

H2: A reduction in health will be dependent on the formal protection of the Kaya; gazetted or un-gazetted.

H3: There will be a significant difference in forest area from 1989 and 2001 data compared to 2010 data.

H4: Rate of forest loss will be dependent on the formal protection of the Kaya; gazetted or un-gazetted.

H5: There will be a significant difference in forest perimeter to area ratio in 2001 compared to 2010, which has implications for 'edge effects'.

H6: Local community perceptions are in line with empirical findings.

1.3 Overview and thesis structure

Section 2 summarises the global causes of tropical forest degradation and forest loss and highlights how these effects can be minimised. As well as this, different types of monitoring are introduced before the study species, *Colobus angolensis palliatus,* and the study area, the coastal Kwale district in Kenya, are discussed.

Section 3 describes the processes of forest census, mapping and questionnaires used to collect data on forest health. Details are also given on the statistical analyses used to investigate

Section 4 presents the results of all areas of the research from the forest census, mapping and questionnaires.

Section 5 discusses the results in the context of the wider primate and habitat context, focusing in particular on how to improve monitoring in the future.

2. Background

2.1 Drivers of tropical deforestation and degradation

Humans are impacting the environment through the exploitation of natural resources (Chapin Iii *et al.*, 2000, Wright, 2005, Bradshaw *et al.*, 2009). Conservation strategies must therefore anticipate the associated threats as well as address the contemporary pressures in order to prove successful (Spector and Forsyth, 1998). The current anthropogenic impact on tropical forests is a focal point for conservation research due to the global importance of these forests in biodiversity, ecosystem functioning and carbon sequestration (Shearman *et al.*, 2009). The main existing anthropogenic drivers of tropical forest loss can be separated into two scales: local and global. Local drivers refer to land use change, wood extraction and hunting whilst global drivers include atmospheric change and climate change drivers (Wright, 2010, Wright, 2005). For the purposes of this literature review, I will only document the local drivers. Global drivers are, of course, important, but their effects are less likely to be detected by the research methodology employed in this study.

2.1.1 Local drivers

Land use change and wood extraction are the two key drivers of deforestation, posing different threats to tropical ecosystems. Approximately 30% of the global land surface has been deforested (Schmitt *et al.*, 2009). Furthermore, deforestation caused by land use activities has transformed approximately half of closed canopy tropical forest to other uses (Wright, 2005, Foley *et al.*, 2005) whilst rates of anthropogenic habitat conversion are currently at their historical maximum (Seabloom *et al.*, 2002). Land use activities include subsistence agriculture, intensifying farmland and production or expanding urban centres (Lambin *et al.*, 2003, Foley *et al.*, 2005). Deforestation in Latin America offers a useful case study for the impact of local drivers; the Amazonian rain forest has witnessed acceleration in deforestation occurring in areas suiTable for modern agriculture. In Brazil, Bolivia, Paraguay and Argentina, seasonally dry, high rainfall and flat surfaces are being exploited for soybean production (Fearnside, 2001). Consequently, croplands and pastures have become one of the largest terrestrial biomes globally (Foley *et al.*, 2005); more than 25% of the total land surface is managed through grazing. This is a larger geographic extent than any other form of land use (Asner *et al.*, 2004). There is a great concern that vast monocultures of fast growing, non-native tree species will take over landscapes with monumental negative effects on species biodiversity and ecosystem functioning. A persuasive example of this process can be found in the case of palm plantations. Oil palm (*Elaeis guineensis*) is currently the world's most rapidly expanding crop, replacing vast areas of forest in Southern Asia and elsewhere, currently estimated at over 13 million Ha, which has either directly, or indirectly, replaced tropical rainforest (Danielsen *et al.*, 2009). Oil palm plantations support fewer species than forest ecosystems and contribute to habitat fragmentation, pollution and greenhouse gas emissions (Fitzherbert *et al.*, 2008). The monotony of single species forests are exaggerated by uniform age structure and are structurally less complex than natural forests (Begon, 1990). This is demonstrated by a study by Danielsen *et al.* (2008) who found that in Indonesia, oil palm plantations were species poor and contained few forest species (Danielsen *et al.*, 2009). Furthermore Fitzherbert *et al.* (2008) found that palm oil had significantly fewer vertebrate species than primary forests and much lower species richness than disturbed (logged or secondary) forests.

One of the most fundamental ecological relationships is the interaction between species richness and area; the larger the forest is, the larger the number of species encountered (Begon, 1990). Forest fragments are analogous to islands in many respects; they reflect the reduced range of resources offered by smaller areas (Haila, 2002). Habitat destruction is considered the key cause of species extinction (Pimm and Raven, 2000). Thus, in forests with high rates of deforestation and encroachment, the decrease in forest fragment area will result in a decrease in number of species found. A 22 year investigation of ecosystem decay in Amazonian forest fragments found a relationship between species richness and forest fragment size; intact forests contained a higher number of species per unit area than in fragmented forests (Laurance *et al.*, 2002). The ecological consequences of biodiversity loss are controversial and widely documented in the scientific literature. It has been suggested that a large proportion of species richness is required to maintain ecosystem stability and sustain function (Schwartz *et al.*, 2000). Hooper (2005) states that ecological experiments, observations and modelling have shown that ecosystem properties depend on the characteristics of biodiversity, the size of the forest and the time in the ecosystem.

Finally, there is a concern that local extinctions of species can occur after a time lag following habitat loss or degradation (Kuussaari *et al.*, 2009, Pimm and Raven, 2000, Vellend *et al.*, 2006). For example, Struhsaker (1976) documented a 10 year lag period after the 90% loss of major food resources and a significant decline in Ververts (*Cercopithecus aethiops*) in Kenya.

Another major implication of deforestation is that it breaks up and fragments forests (Laurance, 2004). As mentioned in the introduction, habitat fragmentation is the conversion of once large continuous blocks of habitat to a less continuous, spatial separation of habitat units (Fischer and Lindenmayer, 2007). Habitat fragmentation has 3 key impacts on forests; a decrease in the size of the forest, an increase in fragment isolation and an increase in total forest edge (Fahrig, 2003).

Habitat fragmentation and the consequent effect on primates have been well documented in conservation science (Chapman *et al.*, 2007, Onderdonk and Chapman, 2000, Mbora and Meikle, 2004, Arroyo-Rodriguez and Mandujano, 2009, Estrada and CoatesEstrada, 1996, Wong and Sicotte, 2006, Wahungu *et al.*, 2005, Anzures-Dadda and Manson, 2007, Marsh, 2003). Habitat fragmentation is thought to be the principle threat to primates; studies mainly conclude that there is a negative effect of fragmentation on primate biology or ecology (Arroyo-Rodriguez and Mandujano, 2009).

One of the most critical consequences of habitat fragmentation is 'edge effects'. Edge effects are the result of the interaction between two adjacent ecosystems separated by an abrupt transition (Murcia, 1995). At edges there is an exchange or flow of energy and organisms across the boundary (Harper *et al.*, 2005). Edge effects effect physical variables such as radiation, moisture, temperature and humidity (Fischer and Lindenmayer, 2007), as well as ecological processes including nutrient cycling, decomposition and evapotranspiration. These processes in turn influence the changes in forest structure, including factors such as canopy cover and tree density. Edges also influence biodiversity, affecting dispersal, establishment, survival and growth (Harper *et al.*, 2005).

This has severe implications in light of tropical forest loss. Each year, 20 000km of new forest edge in the Brazilian Amazon alone is generated as a direct result of deforestation (Laurance, 2004).

Fragmentation isolates forest fragments in the theory of metapopulation the equilibrium of colonisation is dependent on isolation and extinction. Larger islands, or islands closer to the mainland (or in this case forest fragments), will contain more species than smaller isolated habitats (Arroyo-Rodriguez and Mandujano, 2009, MacArthur, 1967)

It seems unlikely that the rate of habitat conversion will slow in the near future given the current human population growth which is projected to double in the next 50 years (Tilman *et al.*, 1994, Grau and Aide, 2008). Thus the challenge to meet the increasing food needs of this growing population without destroying the remaining forest ecosystems arises.

Not only is the area of forests under threat, but the quality of the remaining forest habitat is at risk from wood extraction (Dangwal, 2005). Extracting more wood than the regenerative capacity of forests leads to slow degradation and consequently reduced forest health in the long run (Dangwal, 2005). The need for wood extraction comes from an array of pressures, including local, timber and firewood extraction in response to the demand from commercial industries for raw materials (Dangwal, 2005). Logging, and the related disturbance, alters ecosystem composition, biodiversity and opens remote areas to poaching (Laporte *et al.*, 2007). For example selective logging of Mahogany, *Swietenia macrophylla*, in the Brazilian Amazon has assisted regional deforestation; logging companies have opened 3000km of logging routes in southern Para (Veríssimo *et al.*, 1995). After logging, forests are often converted for cattle pasture and thus the forest resources are exhausted.

Commercial logging is responsible for the transition of primary forest to poorer quality, secondary forest. Furthermore, logging also reduces biomass, damages soils and other vegetation present, increases vulnerability to fire and conversions to grassland, scrub or agricultural land which may then persist for decades (Shearman *et al.*, 2009). Wood extraction is a growing concern; for example in the Amazonian rainforest alone, 12,075-19,825km^2 of area was logged per year between 1999 and 2002. This equates to 0.1 billion metric tons of carbon released into the atmosphere, (Asner *et al,.* 2005) which has severe implications for global drivers of tropical forest degradation: atmospheric and climate change.

Hunting is another local driver of forest degradation. When the extraction rate in hunting is above the species ability to reproduce it can lead to species extinction. For example, the result of habitat destruction coupled with hunting in 1999 led to the extinction of the Miss Waldron's red Colobus monkey (*Procolobus badius waldroni*) (Oates *et al.*, 2000). This has further ramifications if the surrounding plant species are dependent on the extinct species for dispersal or pollination. This threat is exacerbated by the development of modern hunting equipment such as guns, wire snares and battery powered lamps.

2.2 How can forest loss be prevented?

The protection of land from deforestation and degradation has helped conservation success over the last 20 to 50 years (Seabloom *et al.*, 2002). Globally, 18% of all tropical and subtropical moist forests and 9% of all dry tropical forests are protected (Brooks *et al.*, 2004). Legally Protected areas are seen to be the key defence against forest loss and species extinction (Joppa *et al.*, 2008); they have significantly lower rates of land clearing compared to non-protected areas (Nagendra, 2008). If the tree cover threshold is set at 10%, it shows that global forest cover is in the region of 39 million km^2 where only 7.7 % fall within protected areas under certain IUCN criteria (Schmitt *et al.*, 2009). When the global average forest cover is broken down into WWF ecoregions (taking into account differences between forest ecosystems), 65% of the 670 ecoregions have less than 10% of their forest cover protected (Schmitt *et al.*, 2009).

However, within the scientific literature, the effectiveness of protected areas has been hotly debated (Joppa *et al.*, 2008, Ewers and Rodrigues, 2008, Nagendra, 2008, Brooks *et al.*, 2004, Curran *et al.*, 2004, Bonham *et al.*, 2008). A protected area system is only as effective as the governments that protect them; corruption, political instability and economic crises can result in poor protected area networks (Curran *et al.*, 2004)

One important protective factor against forest degradation is that of sacred groves. Sacred groves are conservations first form of habitat protection (Dudley *et al.*, 2009). These are fragments of forest or stands of trees that local communities conserve primarily because of their associated religious importance (Mgumia and Oba, 2003). They include burial grounds and sites of deity worship (Bhagwat and Rutte, 2006, Chouin, 2002). They consequently offer a special form of protection whereby social taboos rather than laws influence human behaviour (Colding and Folke, 2001). Sacred groves have been proven to offer a higher form of forest protection than forest reserves (Campbell, 2005; Anderson, 2004) which can be demonstrated by the high levels of endemism found within these sacred groves. Due to centuries of community protection, sacred groves have become reservoirs or sanctuaries for biodiversity (Mgumia and Oba, 2003). For example, Burgess (1998) found that that the proportion of endemic species in sacred 'Kaya' forests in Kenya was consistently high for all species groups; 80% of millipedes found in Kaya forests are endemic. It has been suggested that government bodies should declare sacred groves as preservation sites and try and incorporate them into existing protected area networks to compliment the legal protection (Mgumia and Oba, 2003; Bhagwat and Rutte, 2006). These networks will be more effective with the support of local communities (Bhagwat and Rutte, 2006). The exclusion of local people is believed to be one of the reasons why protected areas can be ineffective, despite the large sums of money and management power in them (Bhagwat and Rutte, 2006). However, sacred groves are now under threat by the breakdown of traditional customs by the increasing influence of Islam and Christianity and immigration of people who do not owe allegiance to traditional authorities (Rodgers and Burgess, 2000).

2.3 Monitoring tropical forests

Monitoring is crucial for conservationists to gauge the effect of their interventions (Danielsen *et al.*, 2005). However, it is impossible for managers to monitor everything of potential interest within an ecosystem and the subsequent decision of what to measure is critical (Carignan and Villard, 2002). Biodiversity surveys are fundamental in protected area design; however the demands characteristic of full biodiversity surveys, especially in the tropics, greatly exceeds the capacity of scientific institutions. It is naive and unrealistic to assume monitoring the fate of all taxa is accomplishable (Spector and Forsyth, 1998).

Conservation focuses disproportionately on particular, charismatic endangered species to monitor and conserve in tropical forests (Simberloff, 1999). Conservation can be deemed more effective if it relates to both ecosystem complexity and incorporates biodiversity and its associated ecological processes (Lande, 1998). There should be a mixture of single species and ecosystem conservation in order to guarantee species of ecological importance, or those that facilitate the understanding of ecosystem health, are suitably managed.

2.3.1 Indicator species

To tackle this issue, conservationists have developed the concept of indicator species (Noss, 1990). Indicator species serve as surrogates for the entire ecosystem and are therefore a high priority for conservation research; wildlife habitat quality can be assessed using the assumption that the population density of an indicator is an index of habitat quality (Landres *et al.*, 1988). Plant and animal species have been used for decades as indicators of air and water quality and agricultural and range conditions (Noss, 1990). Ideally indicator species should be sensitive to changes in environmental conditions or stress in order to provide an early warning of negative trends over broad geographical areas. As well as provide a continuous assessment over a wide range of stress and should be easy and cost effective. Increasingly, vertebrates are being used to assess population trends and habitat quality for other species (Landres *et al.*, 1988).

The ideal indicator taxon is hard to identify (Spector, 1998). It is essential to choose a species which is a specialist or sensitive to change; highly sensitive taxa under the threat of extinction should, however, be used with caution. If indicators are to act as surrogates for the entire biota, then the geographic patterns of species richness and endemism should closely reflect those of other taxa (Spector, 1998).

Box 1: Case study of the northern spotted owl as an indicator species (Simberloff, 1999)

The northern spotted owl (*Strix occidentalis caurina*) was chosen as an indicator species because it is a charismatic, flagship species. In this case, the owl was used to reflect the state of health of the entire Pacific Northwest region. This species was chosen due to its vulnerable status and social interest; there was no empirical evidence, however, to support the idea that the species had the capacity to predict the health of the entire ecosystem and the species dwelling in them. Nevertheless, due to the spotted owls threatened, diminishing habitat requirements, (old-growth rainforests), the protection of this type of habitat had beneficial qualities. It was home to an array of species which were also consequently protected; saving enough of this habitat for the owl would therefore save the other species in it. However this does not yield any results on habitat quality or health and in this light could be costly and inefficient.

Simberloff (1998) reports several shortcomings of relying on them as indicator species for monitoring. Intensive management of an indicator species could increase the prevalence of the species without increasing the health of the ecosystem yielding a false result. Furthermore, no two species occupy the same niche and no single species should be expected to act as an indicator for an entire ecosystem there will be serious negative consequences if the indicator species concept is incorrectly applied or an inappropriate species is chosen (Lindenmayer, 1999).

However, there are merits in using this method as a guide. Using well chosen indicator species can be a useful tool to conservation science. At the very least, it can offer a cost-effective methodology for habitat monitoring, and represent a pragmatic response to limited resources. Finally, Spector and Forsyth (1998) call for increased efforts to define those indicator taxa which can yield the maximum amount of ecological and systematic information about the vanishing tropics.

2.3.2 Locally based monitoring

The current threats to tropical forests, as outlined above, calls for an increased need for effective monitoring that incorporates both scientific rigor and practical feasibility (Rist *et al.*, 2010). Monitoring, such as indicator species monitoring described above, is often expensive and therefore unsustainable, both logistically and technically.

Alternatives to professional based monitoring have recently been discussed in the scientific literature; useful information can be obtained using local knowledge and involving local communities as a basis (Rist *et al.*, 2010, Hockley *et al.*, 2005).

'Locally based monitoring' is a broad term used to refer to techniques such as participatory monitoring, community monitoring, hunter self monitoring and ranger based monitoring. These can be conducted by self monitoring of resource extraction by local users, censuses by rangers or inventories by unqualified naturalists. These monitoring techniques are fundamentally linked to resource management, ranging from individual species to ecosystems. The key distinction of locally based monitoring is that it is conducted at a local scale by low educated individuals.

When appropriately designed, locally based monitoring schemes can yield relevant results which are as accurate as professional monitoring (Danielsen *et al.*, 2005). This is illustrated by Rist *et al.* (2010) where bushmeat hunter catch per unit effort was measured in Equatorial Guinea. Local interviews were found to yield accurate, powerful and more cost effective methods, over the professional technique, to monitor the condition of natural resources, collecting 240% more catch and effort data than the professional technique.

Advantages of locally based monitoring include the reinforcement of existing community based resource management systems and can result to changes in local attitudes to the environment and sustainable resource management (Danielsen *et al.*, 2005). Furthermore it builds capacity between local communities and government authorities and enhances education and awareness on resource use.

Table 1 Comparison of suitability of locally-based and professional monitoring in relation to monitoring need, threats and availability of resources for monitoring. (Adapted from Danielsen et al. 2005)

Type of monitoring and resource	**Locally- based monitoring**	**Professional monitoring**
Species or population trends	Yes, but certain cryptic species not possible	Yes, however often practically difficult
Trends in the extent of habitats and thier conditions	Yes, especially habitat condition	Yes, especially large scale monitoring, the use of remote sensing can be used for large scale benefits
Trends in ecosystem services	Yes, at a local scale	Difficult at a local scale, but modelling and remote sensing can be used at large scale
Trends in threats	Yes, at a local scale (for example local harvesting)	Yes, at a larger scale
Trends in the impat of management interventions	Yes, at a local scale	Yes, at a larger scale
Financial resources for intitation of monitoring	High	High
Recurrent financial resources	Low	High

What kind of data can emerge from locally based monitoring? From Table 1 it is evident that locally based monitoring can generate cheaper and locally meaningful data on habitat condition, habitat size and population sizes of certain species. It can provide evidence for local changes in ecosystem services such as reliable provisions of clean water. A drawback of locally based monitoring is that it cannot make any meaningful estimates of global scale ecosystem services, such as carbon sequestration.

In conclusion, conservation planners should consider the use of local communities when developing monitoring initiatives locally based monitoring can address several of the shortfalls of professional monitoring. Finally, locally based monitoring has the capacity to be low cost, rapid, locally relevant and able to build capacity among local constituents (Danielsen *et al.*, 2005).

2.4 Study site

The study site is located in the Kwale district in the Coastal Province of Kenya. It is midway between Mombasa and the North eastern Tanzania; Kwale stretches approximately 8322km^2 in area. The population in the Kwale district stands at approximately 536,381, where 49% are below the age of 15 (WWF, 2009) and 45% of people live in absolute poverty (WWF-UK, 2005). The main type of habitat is agriculture; including grasslands, woodlands, swamps, shrub-lands, forestry plantations and annual and perennial cropland (Burgess *et al.*, 1998). The environmental conditions in the area have an average temperate of approximately 24.6-27.5°c (WWF-UK, 2005). The annual rainfall patterns are bimodal, where the main seasonal rains start in March and finish in August. Secondary rainfall begins in October and finishes in January. The remaining forests fragments in the Kwale district are remnants of formerly forested lowland rain forest, swamp forest and scrub forest (Burgess *et al.*, 1998). Interestingly, the unique forests grow on coastal sedimentary rocks (Hawthorne 1993).

These forests are home to the Angola black-and-white Colobus (*Colobus angolensis palliates*). Figure 2 and Table 2 show the 16 forest fragments surveyed in this study.

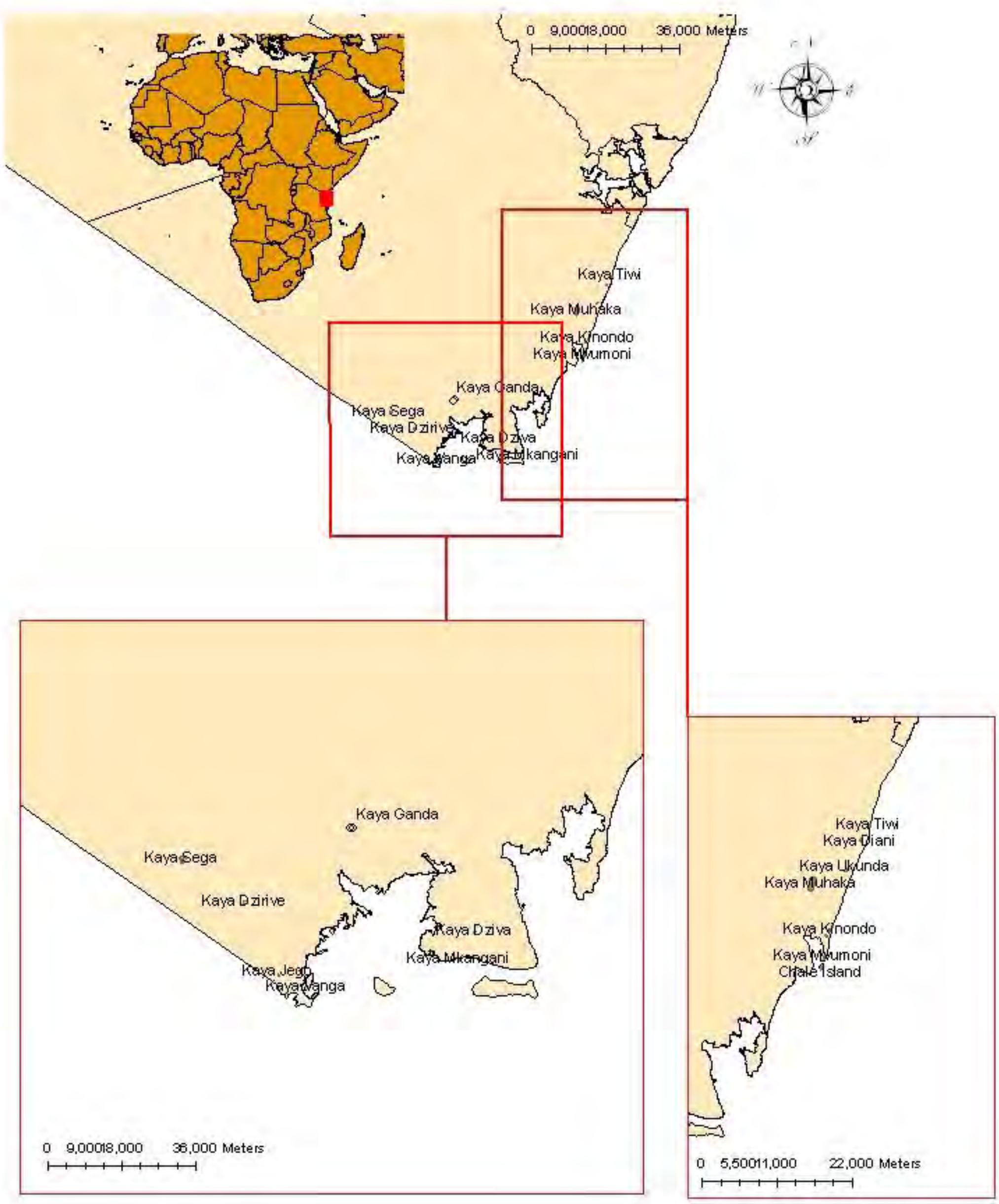


Figure 2. The study site: The Kwale district in Kenya. The 16 forest patches researched in this study are labelled.

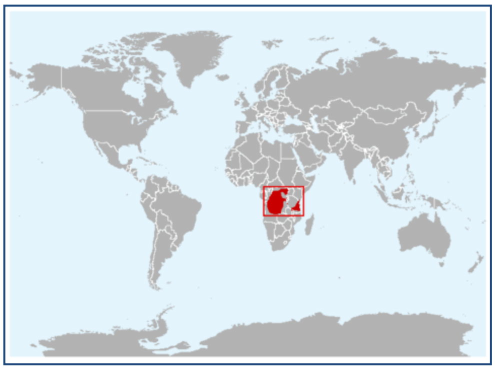
Table 2: The 16 forests surveyed displaying the toporegion and management status.

Kaya Name	Toporegion	Management status
Tiwi	Ukunda	Gazetted
Diani	Ukunda	Gazetted
Ukunda	Ukunda	Gazetted
Muhaka	Ukunda	Gazetted
Ganzoni	Ukunda	Gazetted
Kinondo	Ukunda	Gazetted
Timbwa	Ukunda	Un-gazetted
Chale Island	Ukunda	Gazetted
Muvmoni	Ukunda	Un-gazetted
Mkangani	Shimoni	Un-gazetted
Dzipha	Shimoni	Un-gazetted
Ganda	Shimoni	Un-gazetted
Jego	Vanga	Gazetted
Dzirive	Vanga	Un-gazetted
Sega	Vanga	Gazetted
Vanga	Vanga	Un-gazetted

2.5 Study species: Ecology of *Colobus angolensis palliatus*

The last remaining 31 colobine species are the remaining species after a long series of adaptive radiations; originally, the colobines diverged from the cercopithecis monkeys 15.97 ± 0.05 Ma to 11.608 ± 0.005 Ma (million years ago) in Africa (Oates, 1994a). More specifically, the study species of black-and-white Colobus monkeys are broadly regarded as a diverse group of 5 species; *C. santanas, C. polykomos, C. vellerosus, C.guereza and C. angolensis (Oates, 1994b).* Furthermore, the study species *Colobus angolensis* is sub-divided into 6 species; *C. a. angolensis, C. a. cordieri, C. a. cottoni, C. a. pallitus, C. a. prigoginei* and *C. a Ruwenzorii* (IUCN, 2008). Of specific interest to this research is C. a. *palliatus* which is discontinuously distributed across the southern highlands and coastal forests in southern and eastern Tanzania and south-eastern Kenya

Kenyan distribution of *C. a. palliatus* is solely restricted to the southern coastal forests of the Kwale District (Anderson *et al.*, 2007a). Figure 4 shows the distribution of the five sub-species of black-and-white Colobus monkeys.



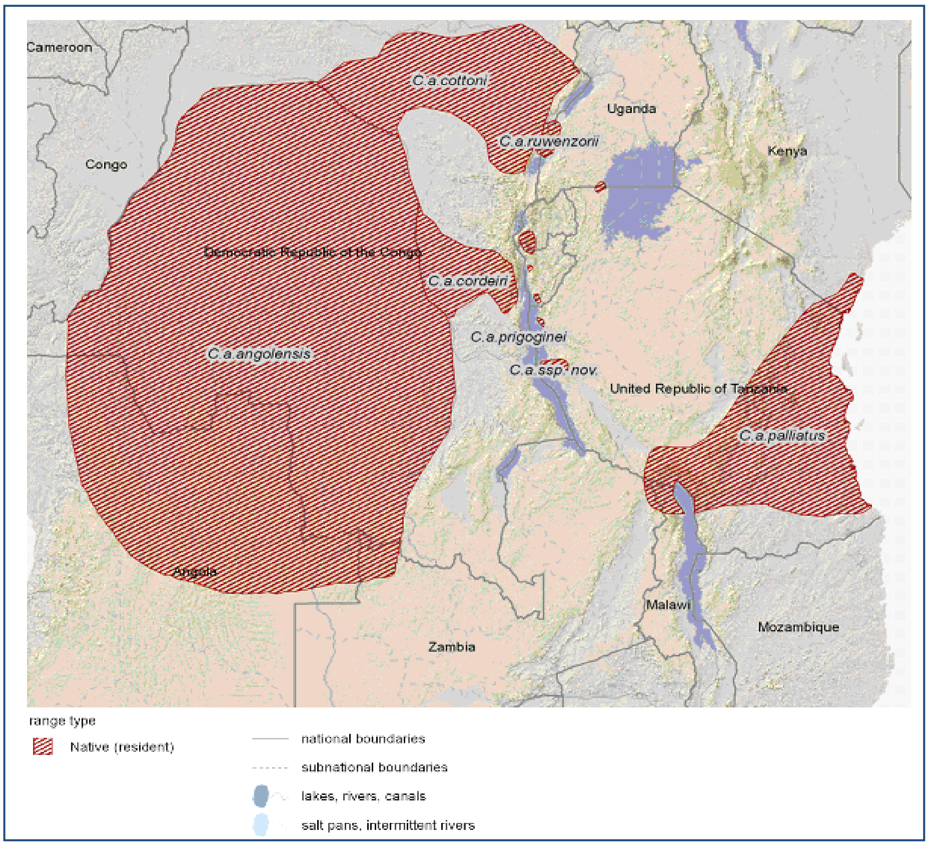


***Figure 3: Distributions of the five sub-species of black and white colobus monkeys* (Colobus angolensis) (Kingdon, 2008)**

C. a palliatus inhabits lowland, coastal, gallery and montane forests (Oates, 1996). Group composition structure typically comprises approximately 2-20 individuals, including 1 or more adult males and more than 1 adult female; this is dependent on the number of offspring within the group (Oates and Davies, 1994a) Colobine prevalence is limited largely by food resources within their home ranges, with limited apparent impacts of disease, predation and competition (Oates and Davies, 1994a). Ecologically, *C. a. palliatus* is a folivorous primate. It spends less time feeding and moving and more time resting in comparison to primates of an insectivore or frugivore nature (Although this means that colobines are successful canopy dwellers, it does make them vulnerable to changes in habitat as well as hunting pressures (Oates, 1994a). Furthermore, this vulnerability is exacerbated by the current threats to tropical forests. The most significant threat to Colobus population survival is habitat loss (Anderson, 2004; Oates, 1996). Although *C. a. palliatus*is not currently listed as threatened, the subspecies has been acknowledged as a species which has the potential for vulnerability (Kingdon, 2008). This is because the species is currently being confined to islands of fragmented forests in eastern Africa (Kingdon, 2008, Anderson 2007(a)). Tourist development schemes and rapid population growths (2.6% per year, WWF-UK, 2005) in this area have resulted in the increased need for forest resources such as timber (Marshall and Jenkins 1994; Robertson and Luke 1993). This coupled with forest clearances for intensive livestock rearing and agriculture, paints a grim picture for the remaining *C. a. palliatus* in eastern Africa.

3. Methods

This section describes the methodology applied in this research regarding data collection and subsequent statistical analyses including the rationale for the choice of statistics. Fieldwork for the data collection was conducted in eight weeks between May and June 2010 in the Kwale district in Kenya; all the Kaya forest fragments surveyed were in the toporegions of Ukunda, Shimoni and Vanga. A total of 16 forest fragments was surveyed (n=16). All statistics were calculated using the statistical computer program R 2.9.0.

Before any research could be conducted in the Kaya forests, permission from the National Museum of Kenya (NMK) and Kaya elders was sought.

3.1 Sampling methods - Sweep surveys

Data collected in this study were intended to be directly compared to baseline data collected by Anderson in 2001 (Anderson, 2004) and will therefore follow the same sampling methods. The sweep survey sampling method is deemed appropriate because it allows quick and effective surveys of small forest fragments (Whitesides *et al.*, 1988) and has frequently been used successfully in other primate censuses (Karere *et al.*, 2004).

Before each census started, we carried out a planning phase. This involved utilising existing maps, information from local guides and familiarity with given forest fragments by members of the research team to determine forest sizes, shapes, potential transect routes and starting positions. Further planning included performing a pilot study to determine feasibility, time frames and Colobus identification. The methods used were considered appropriate and were therefore executed in further forest surveys.

During the period of the 8th May – 15th of June 2010, all forest fragments were systematically surveyed using one day sweep sampling methods. Two teams were employed, each comprising of one trained team leader and one local guide. Surveying commenced between 06.30am and 07.00am. The teams walked parallel transects approximately 100m apart (consistent with censes from 2001), starting at the same time and moving at the same speed (approximately 2 miles per hour) through the Kaya forest. Sweep sample accuracy was facilitated by the maintenance of compass bearings throughout transects, the use of a Global Positioning System (GPS). Teams re-grouped after each forest transect in order to resynchronise movements. In smaller forest patches only one team was used, with two trained team leaders and a forest guide.

3.1.1 Colobus frequency

For each Colobus group encountered, the following measurements were recorded: group size, GPS location, time, direction of travel and demographics including the sex and age of the individual. The age class was categorised into 'adult', 'sub adult', 'juvenile' and 'infant'. To eliminate double counts, group composition and location were used to identify different groups.

3.1.2 Colobus data statistical modelling

During analysis of the demographic data collected, the mean, standard error, and the range were calculated for Colobus groups, as well as the total group and the age categories of Colobus monkeys.

A generalised linear model using quasi-poisson errors was used to analyse this data, due to the type of data (count data) of the Colobus variable. Area was accounted for using the 'offset' function and log(area) allowing me to model Colobus density in relation to the year the Colobus were surveyed, the management status of the forests and the amount of forest edge exposed to non-forest environments. Quasi-poisson GLM's accounts for over dispersion in the data, the residual deviance (r.d.) of the full model was higher than the residual degrees of freedom (d.f.) (r.d.= 199.19 on 29 d.f.). The variables included in the model are summarised in Table 3. This model was used to determine any relationships between the Colobus density, management status and the year the data was collected. Model simplification using model updates was used in accordance with Occam's Razor: "the correct explanation is the simplest explanation" (Crawley, 2005).

Table 3: Variables used in the statistical modelling of colobus monkeys, disturbance and forest area change.

Variable	Data	Type	Description
Colobus	Count	Response	Colobus frequency
Disturbance	Count	Response	Number of poles cut along the transect
Change	Continuous	Response	Rate of change (Log(Size of the forest 2001/size of the forest 2010))
Log(area)	Continuous	Explanatory	Log of the Size of the forest (km^2)
Status	Categorical	Explanatory	Management status of the forest: Gazetted or un-gazetted
Year	Categorical	Explanatory	Year of data collection
Interval	Categorical	Explanatory	Time periods 1989-2001 and 2001-2010
Forest edge	Continuous	Explanatory	Length of the forest perimeter

3.1.3 Disturbance

The number of tree stumps arising from pole harvesting by local people was recorded. For most tree species, tree stumps will remain for several years, thus enabling this measurement to be used as an index of recent change, approximately 10 years. (Although this method was not used during the census by Anderson (2007), Chapman *et al.* (2007) used this to account for anthropogenic disturbances). The tool used and size of stump was also recorded. Other anthropogenic disturbances were recorded during the sweep surveys. These included pitsaws, presence of loggers, snares, traps or presence of hunters and charcoal burning. There were, however, very few of each and therefore this data was not included in further analysis. When a disturbance was encountered, the GPS location was detailed and recorded.

3.1.4 Disturbance statistical modelling

Another quasi-poisson generalised linear model, using the offset function to control for area, was used to determine if there was any relationship between the disturbance found in the forest and the management status the data was collected. Again, a quasi-poisson was used to account for the over dispersion of the data (r.d. = 163.58, d.f. = 14). These variables are summarised in Table 1.

3.2. Sampling methods: Mapping

Baseline data of forests sizes from 1989 and 2001 was collected by Anderson including both forest perimeter sizes and the area. Forest size from 1989 was in the form of maps published in 2001. These maps were digitalised in order to determine size and perimeter of forest fragments.

In this study forest fragment boundaries were mapped using a Global Positioning System (GPS) unit, (Garmin eTrex H). The GPS was set to record positions every 5 seconds (recording positions in degrees/minutes/seconds) using the 'tracks function', while an observer walked the perimeter of each fragment. The boundaries of the fragment were defined by the local guide. GPS coordinates were loaded into a Geographic Information System (GIS) Map Source, and converted to (.KLM) files using an online GPS visualizer (www.gpsvisualizer.com/gpsbabel/). EZ Geowizard was then used to convert (.KML) files to shape files and these shape files were viewed in a GIS database (ArcMap 9.3.1). The tracks were then converted from lines into using EZ Geowizard. The polygons were projected onto the WGS1984 UTMZONE 375 coordinate system in ArcMap to calculate the forest fragment area and perimeter.

Kaya Muvmoni was excluded from this statistical analysis as the baseline data collected by Anderson (2007) encompassed this Kaya in amongst the larger forest which it resides in, 'Chale Point'. There was no exact size for Kaya Muvmoni from 2001.

3.2.1 Forest area statistical analysis

An analysis of variance (ANOVA) was carried out on the forest areas using variables from Table 2 (Change, Status and Interval). This was appropriate because all the explanatory variables were categorical data and the response variable was continuous data. This was used to test the interactions between the status, year and the rate of area change.

3.2.2 Forest perimeter statistical analysis

A t-test was used to distinguish whether there was a significant difference in the ratio of forest perimeter length to area (see Table 3 for variables) from 2001 to 2010. This will indicate whether the amount of forest edge exposed to non-forest edge is increasing.

3.3 Sampling method: Questionnaires

The questionnaire survey was conducted face-to-face with Kaya elders and forest guards, in a semi-structured interview framework (Milner-Gulland, 2007) ; this was designed to determine local opinions on the future of the Kaya management, and to investigate local knowledge on changes in trends over the past 10 year period in forest health (Appendix 1). The use of semi-structured interviews with a fairly open framework allowed for focused, conversational two-way communication, and opportunity to clarify any points of misunderstanding on either side.

Studies on sustainability involve measuring changes over time. This can be somewhat problematic as in most cases there will inevitably be a reliance on people's recall of the past; one disadvantage of this is that the past is filtered through people's perception and may not give an accurate representation of the series of events. However, in this study, these changes can be directly measured through the census studies and by comparing them to the baseline data from 2001. Therefore the Kaya elders' answers can be analysed to see whether they are in concordance with the actual findings. This will facilitate the development of future management strategies and determine whether the study methods used throughout this investigation should be interlinked with management in the future. It also seeks to understand the importance of local knowledge on the forests in order to develop the most accurate forest protection and monitoring in the future.

The original aim was to complete ten questionnaires per Kaya. However, in some cases this was not possible, but as many participants as possible were interviewed (n=152). They were carried out in Swahili and translated by a field guide with appropriate fluency in English.

A pilot study was carried out in Diani to develop the questionnaire and check if the phrasing of the questions and the response categories were appropriate. Three different respondents were chosen to develop the questionnaires, they were all Kaya elders.

Each Kaya elder was approached prior to research to ask permission to conduct semi-structured interviews with them; this gave the opportunity to introduce the project. Later, during the project, the elder was interviewed at their home. Elders were interviewed individually to minimise any conferring amongst elders, which could influence the results. Details of the entire study were explained, outlining the basic principles and aims of the project. The importance of their knowledge was highlighted to ensure they gave us honest opinions while their anonymity to the surveys was made clear.

For the full list of questions see Appendix 1.

3.3.1 Questionnaire statistical analysis

For the questions with which a uniform answer was given from all participants, no analysis was conducted.

If the census and mapping showed only a slight change in both Colobus abundance or forest size then the trend was assumed to have stayed the same. This 'slight change' was indicated by a 5% change margin either way for Colobus abundance and Forest size. This was assumed negligible, and the forest was assumed to have stayed the same. The following questions can be exactly related to the censes and mapping elements to this project.

1. ***Have you seen a change in the abundance of Colobus monkeys over the past 10 years?***
2. ***Has there been a change in the size of the Kaya over the past 10 years?***

First, a analytical test to identify whether elders were able to identify trends in area and Colobus abundance was conducted using a proportion test to investigate whether the overall proportion answering correctly matched that expected by chance. The probability a Kaya elder would pick an answer by chance was 0.33 (three response variables; increased, decreased stayed the same). This was also analysed using a proportion test on the data split by Kaya.

Next, it was beneficial to know whether this was consistent across Kaya's and to identify which, if any Kaya's are better at identifying trends than others.

Thirdly, an attempt to understand why there could be differences in the ability to predict these changes in forest size and Colobus abundance was made using qualitative comparisons.

Finally, an ANOVA was used to identify if one particular trend was easier to identify than another (Table 4). In the face of unsustainable forest resource use, if Kaya elders cannot identify a decrease in forest size then this will ultimately lead to unsustainable harvest, and is therefore necessary to investigate. This will be key in developing management strategies.

Table 4: Variables used in the ANOVA statistical analysis of the questionnaires

Variable	Data	Type	Description
Predicted response	Continuous	Response	Arcsign transformed data of the percentage of respondents who accurately predicted the trend in the data derived from the census
Actual response	Categorical	Explanatory	The trend from the data collected in 2010, compared to data collected in 2001 (Increased, Decreased, Stayed the same)

4. Results

4.1 Colobus frequency

Out of 16 forest patches, only 63% were found to hold resident Colobus monkeys (Table 5). The total area of occupancy was 2.5km^2. Kaya's Tiwi, Muvmoni, Mkangani, Ganda, Dziriphe and Vanga did not retain any Colobus groups. The total number of *C. a. palliatus* encountered during the census was 115. The average group size across all of the forests was 5 Colobus individuals (Table 6). Table 6 shows the mean, standard error, and range of Colobus occupancy the forest managements surveyed.

***Table 5: Descriptive statistics of forest cover and Colobus status in Kaya forests, Kenya*.**

Forest protection status	Forest cover		Patch size (Km^2)		Colobus status		
	Total area (km^2)	No. Patches	Median	Range	Occupancy	No. groups	No. individuals
Gazetted	2.42	9	0.15	1.47-0.018	6 (66.67%)	16	68
Non-Gazetted	0.35	7	0.05	0.27-0.007	4 (57.14%)	7	36
Total	2.78	16	N/A	N/A	10(62.5%)	23	104

Table 6: *C. angolensis palliatus* group demographics. Adult males (>6 years); adult females (>4 years), sub-adult males (2-5 years), sub-adult females (2-3 years) juveniles (1-2 years); infant (<1 year).

Group structure	Mean	Std. error	Range
Total group size	4.9	0.44	0-9
Adult males	1.22	0.125	0-3
Adult females	2.17	0.18	0-4
Sub-adult males	0.43	0.1	0-1
Sub-adult females	0.35	0.12	0-2
Juvenile males	0.09	0.06	0-1
Juvenile females	0.04	0.04	0-1
Infants	0.3	0.12	0-2

A total of 16 forests were mapped and censured during this study (Figure 2), covering an estimated 2.8km^2 of the coastal forest cover for the Kwale district. Forest patches ranged in size from 0.3km^2 to 1.5km^2 (Table 5). Table 5 sets out forest patch and Colobus occupancy for the forests surveyed

A quasi-poisson generalised linear model (GLM) revealed that there were no convincing effects of management status (gazetted or un-gazetted) or year (2001 and 2010) on the abundance of the Colobus found in the forests (Table 7). Area was controlled for within the model, the results of which can be summarised in Figure 7. The model showed, however, there was a relationship between the ratio of forest perimeter and area to the number of Colobus living in the forest fragments ($t= 3.18$, $d.f.=28$, $p<0.01$**). A higher ratio of habitat had a higher density of Colobus monkeys. Furthermore, an interaction was found between the Year the forest was surveyed and the management status of the forest. Figure 4 shows Colobus density was initially higher in gazetted forests, but has now declined a little while increasing in un-gazetted forests ($t=2.178$, $d.f.=25$, $p<0.05$*).

Table 7: Results of analysis of Colobus density in relation to forest protection status (gazetted/ non-gazetted) and year (2001/2010) and the ratio between forest perimeter and area. The model is a quasi-poisson generalised linear model including the log of area as an offset.

Explanatory variable	F	d.f.	P
Status	0.2287	1	0.6365
Year	0.1891	1	0.6673
Ratio of forest perimeter to area	0.2784	1	0.6021
Status*Year interaction	0.1544	1	0.6978

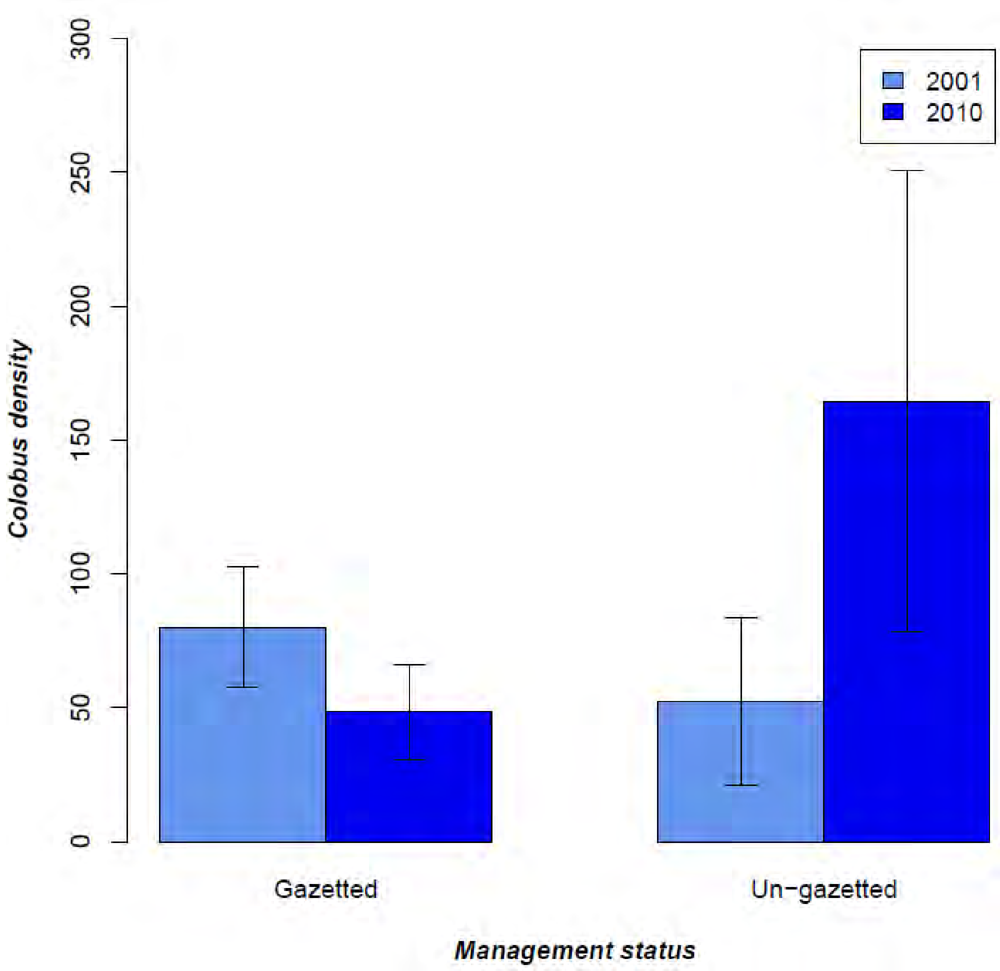


Figure 4: relationship between the Colobus density, the year in which they were sampled (2001/2010) and management status (gazetted/un-gazetted) (n=16). Error bars represent standard error: Gazetted 2001 = ± 22.7, Gazetted 2010= ± 17.9, Un-gazetted 2001 = ± 31.3, Un-gazetted 2010 = ± 86.3

4.2 Disturbance

As described in the methods section, disturbance is characterised by the number of trees or poles removed from the forest. A quasi-poisson GLM including the log of area as an offset, showed that there was no significant relationship between the number of disturbances and the forest management status (Table 8, Figure 5). This indicates that the null hypothesis cannot be rejected. The explanatory variable tested in these analyses therefore did not contribute to explaining the variation in the disturbance index in forest fragments.

Table 8: Results of analysis of disturbance in relation to forest protection status (gazetted/ non-gazetted)). The model is a quasi-poisson generalised linear model including the log of area as an offset.

Explanatory variable	F	d.f.	P
Status	0.68	1	0.42

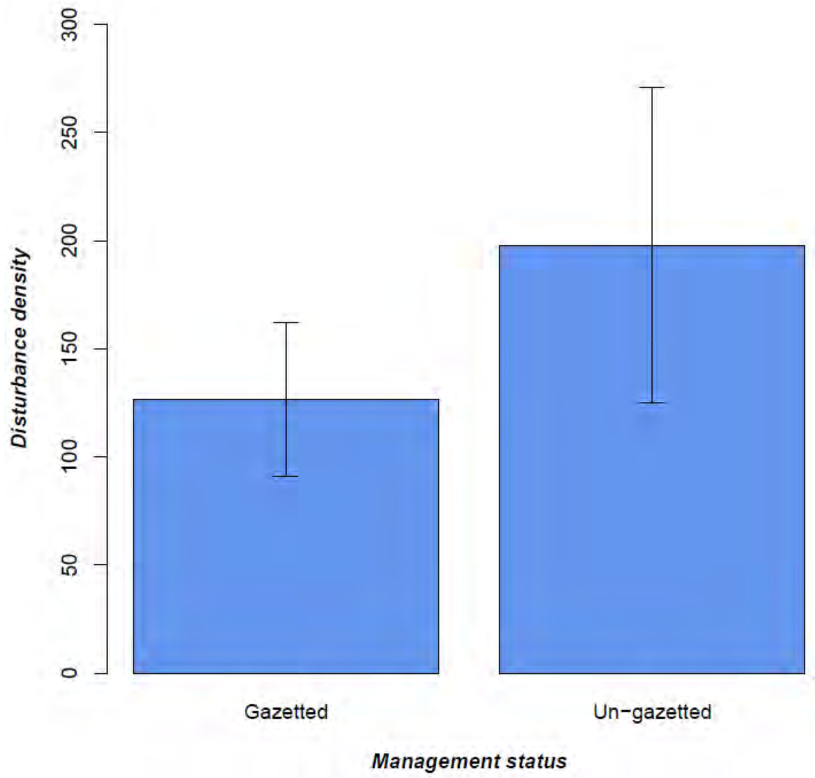


Figure 5: relationship between the density of disturbance in Gazetted and un-gazetted Kaya forests in 2010. Error bars represent standard error: Gazetted S.E. = ±35.5, Un-gazetted S.E. = ±72.9

4.3 Forest cover

An ANOVA testing the relationship between the rate of change and the year of the survey or the management status demonstrated that there is a statistically significant difference in forest area between the two time periods surveyed ($F= 6.4906$, d.f.=1, $p<0.05$*). It shows a decline between 1989 to 2001, but this was not apparent between 2001 to 2010, where the average size of the forest increased slightly (Figure 5). However, the results showed that management status of the forest fragments had no significant effect on the change in forest size. Finally, there was also no interaction between forest status and forest size. This has implications for the management of these areas because it suggests that gazetted Kaya's are in no better condition than un-gazetted Kaya's.

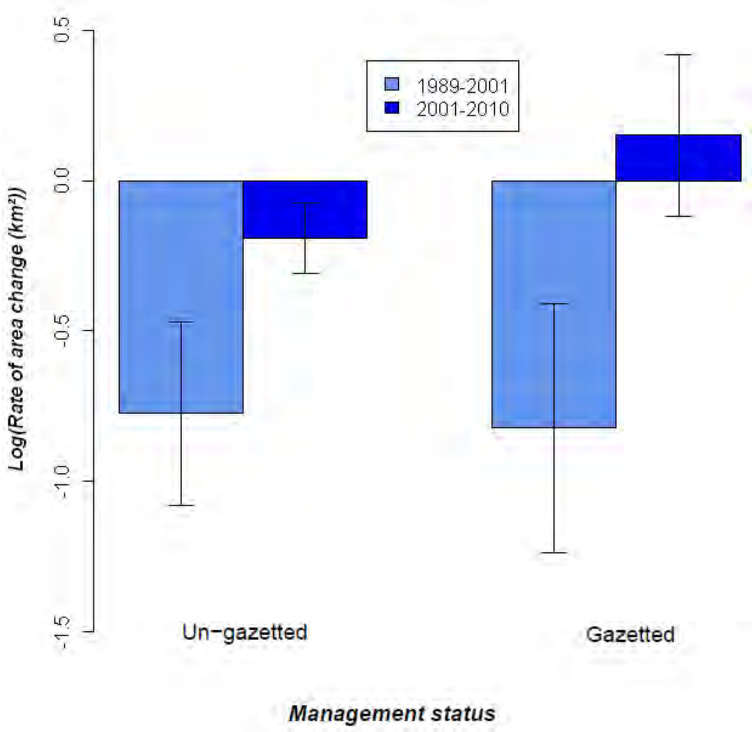


Figure 5: The relationship between the log rate of change of area between the time periods 1989 to 2001 and 2001 to 2010. (n=16). Gazetted 1989-2001 S.E.=± 0.41 mean=-0.8, Gazetted 2001-2010 S.E.=±0.27 mean=0.15, Un-gazetted 1989-2001 S.E.=±0.3 mean=-0.78, 2001-2010 S.E.=±0.12 mean=-0.2

4.4 Changes in forest edge over time

A t-test confirmed that the amount of forest edge differed significantly between 1989 to 2001 and 2001 to 2010 ($t= -7.9199$, d.f.= 29, $p<0.001$**). As indicated by Figure 6, the forest edge, on average, is decreasing in length. This suggests that the problems associated with edge effects, as discussed in the background, are becoming less pronounced over time.

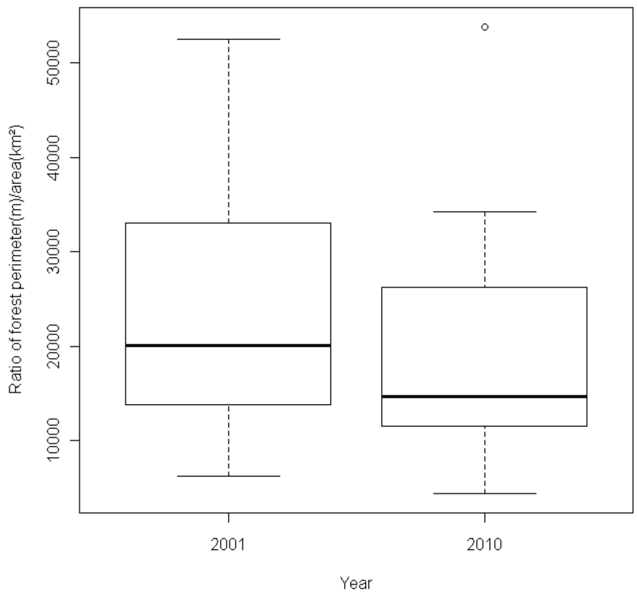


Figure 6: Ratio of the Kaya forest perimeter (m)/ area (km^2) change between the periods 2001 and 2010. 2001 S.E = ± 3682.5, mean= 24424.95,2010 S.E.= ±3421.35, mean= 19398.3

4.5 Questionnaire

Semi-structured questionnaires with Kaya elders revealed that some questions yielded considerable variation in responses, whilst others revealed an overall consensus. Table 9 summarises the main questions answered in the questionnaire with the condensed answers from all the forests recorded. Interestingly, 100% (n=152) indicated that they had seen Colobus recently, with the longest period being last month (Figure 7). This suggests that Colobus groups are usually found in all of the forest fragments. Another overall agreement of Kaya elders (100%, n=152) stated that they needed more support from external sources. It had been acknowledged that if Kaya elders could not identify a decrease in trend, this would have severe implications for sustainability within the forest fragments. Although no overall trend in elder responses has been identified, an ANOVA was conducted to test whether in fact a decrease in trend could still be identified. An ANOVA confirmed that no one trend was easier to identify than another for both Colobus monkeys and forest size; in other words, Kaya elders did not significantly notice a particular trend, such as decreased forest size, over an increase in forest size.

Table 9: Summary of answers from the semi structured questionnaires on Kaya elders (n=152).

Question	Answer option	Observation	
		Count	% of total
Q1. Have you ever seen a Colobus monkey in the Kaya forest?	Yes	152	100
	No	0	0
Q2. Have you seen a change in the abundance of Colobus monkeys over the past 10 years?	1. Increase	74	49
	2. Decrease	46	30
	3.Stayed the same	32	21
Q3. Has there been a change in the amount of disturbance (damage) in the Kaya over the past 10 years?	1. Increase	67	44
	2. Decrease	53	35
	3.Stayed the same	32	21
Q4. Has there been a change in the size of the Kaya over the past 10 years?	1. Increase	8	5
	2. Decrease	80	53
	3.Stayed the same	64	42
Q5. Can support from external sources be improved?	Yes	152	100
	No	0	0
Q6. Do you think without any future conservation action, the Kaya will disappear? If yes, when?	1. <6 months	66	43
	2. 6-12 months	18	12
	3. 1-2 years	47	31
	4. 2-5 years	10	7
	5. 5 years +	6	4
	6. Never	5	3

The summary of responses from Kaya elders to the questions asked during the survey are summarised in Figures 8, 9, 10 and 11 to give a representation of the variation of responses both between and within Kaya forests. Although some Kaya's do show good consensus, they are in varying directions. The next step is to identify whether these Kaya's predicting the correct trends.

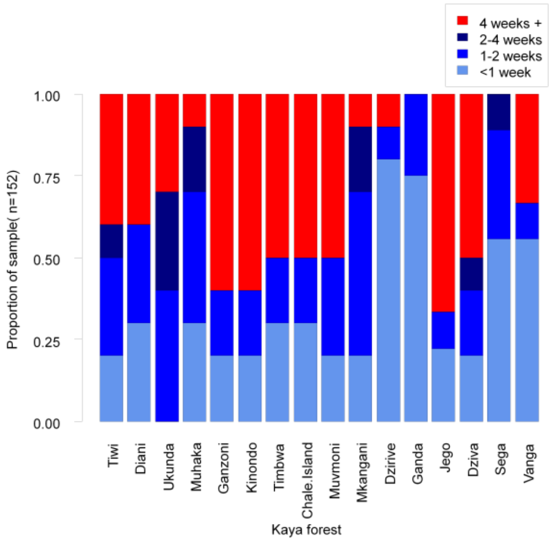


Figure 7: Summary of the answers from the respondents who answered the question: "when was the last time you saw a colobus monkey in the kaya forest fragment?" per Kaya forest(n=152)

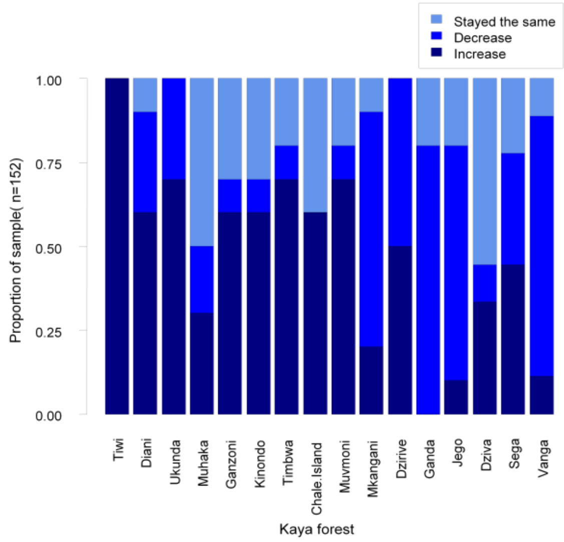


Figure 8: Summary of the answers from the respondents who answered question: 2- "Have you seen a change in the abundance of Colobus monkeys over the past 10 years?" per forest (n=152)

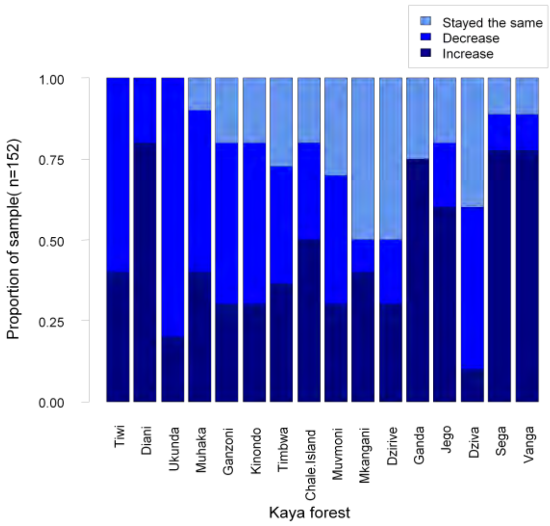


Figure 9: Summary of the answers from the respondents who answered question: 3- "Has there been a change in the amount of disturbance (damage) in the Kaya over the past 10 years?" per forest (n=152)

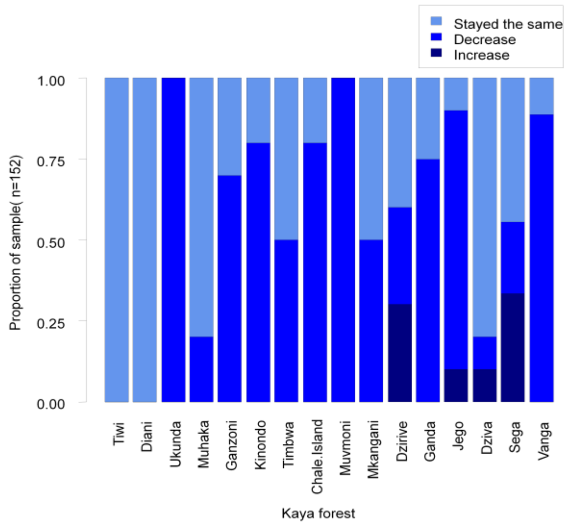


Figure 10: : Summary of the answers from the respondents who answered the question: 4 – "Has there been a change in the size of the Kaya over the past 10 years?" per forest (n=152)

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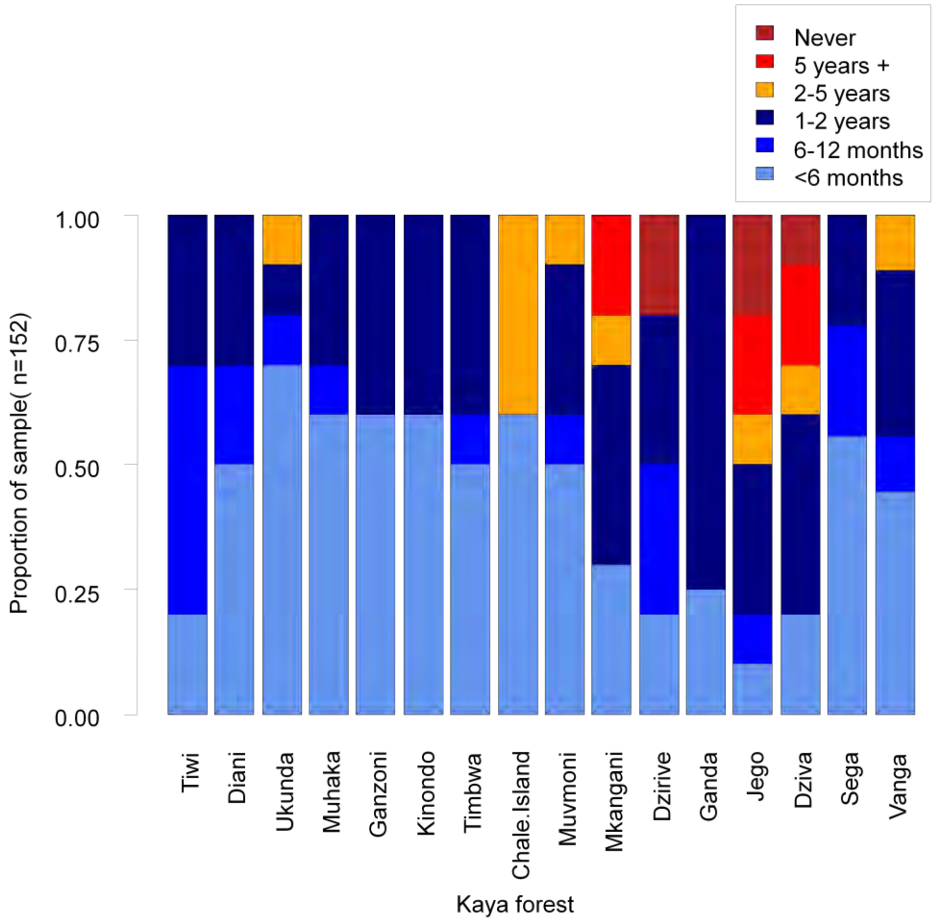

Figure 11: Summary of the answers from the respondents who answered question: 5 –" Do you think without any future conservation action, the Kaya will disappear?

The overall chance of Kaya elders correctly predicting the trend in Colobus abundance is not significantly different from what would be expect by chance. However, the probability of correctly predicting the trend in Colobus abundance varies between Kaya's (X^2=39.7384, p=<0.001***). Table 10 illustrates that the Kaya's which are more in tune with correctly predicting Colobus abundance includes Kaya's Ganzoni, Kinondo, Timbwa, Mkangani, Dzipha, Jego and Vanga.

Table 10: Shows the trend in Colobus monkeys found from the census compared to the number of Kaya elders which correctly identified the trend (Question 2)

Forest	Census response	Percent of elders which predicted this response	Is this answer chosen more or less than expected by chance (Less≤0.33%, more>0.33%)?
Tiwi	Stayed the same	0	Less
Diani	Decreased	30	Less
Ukunda	Decreased	30	Less
Muhaka	Decreased	20	Less
Ganzoni	Increased	60	More
Kinondo	Increased	60	More
Timbwa	Increased	60	More
Chale Island	Decreased	0	Less
Mkangani	Stayed the same	10	More
Dzipha	Increased	50	More
Ganda	Increased	0	Less
Jego	Decreased	70	More
Dzirive	Increased	30	Less
Sega	Decreased	33	Less
Vanga	Decreased	77	More

Similarly, the overall chance of Kaya elders correctly predicting the rate of change of forest fragments is not significantly different from what would be expected by chance. However, the probability of predicting the rate of change varies between Kaya's (X^2=71.1077, p=<0.001***). Table 11 illustrates that Kaya's Ukunda, Muhaka, Chale Island, Mkangani, Dzipha and Dzirive were more able to predict the change in forest size than by chance.

Kaya elders from both Kaya Mkangani and Kaya Dzipha were more able to successfully recognise the changes in trends in Colobus monkeys and forest size than by chance.

Table 11: Shows the trend in forest fragment size found from the census compared to the number of Kaya elders which correctly identified the trend (Question 4)

Forest	Census response	Percent of elders which predicted this response	Is this answer chosen more or less than expected by chance (Less≤0.33%, more>0.33%)?
Tiwi	Decreased	0	Less
Diani	Decreased	0	Less
Ukunda	Decreased	100	More
Muhaka	Stayed the same	20	More
Ganzoni	Stayed the same	30	Less
Kinondo	Stayed the same	20	Less
Timbwa	Increased	0	Less
Chale Island	Increased	0	More
Mkangani	Stayed the same	50	More
Dzipha	Stayed the same	40	More
Ganda	Stayed the same	75	Less
Jego	Stayed the same	80	Less
Dzirive	Decreased	10	More
Sega	Increased	33	Less
Vanga	Stayed the same	11	Less

Finally, from Figure 11 it is clear that most of the Kaya elders have indicated that the forest fragments will disappear in less than 10 years.

5. Discussion

5.1 Overview

The objective of this study is to determine the health of the coastal Kaya forests in Kenya. This study also aimed to address the application and importance of different methods to determine this. These methods included forest censuses, mapping and the use of questionnaires. Furthermore this thesis ultimately aimed to develop novel ways to unite inherent biological processes and the interactions between these processes and the local community that exist in conjunction with them.

5.2 Hypothesis summary

Table 12 summarises the level of support for each hypothesis

Table 12: Summary of the level of support for each hypothesis and the evidence for this support

Hypotheses	Support	Evidence
H1: There will be a significant change in forest area from 1991 and 2001 data compared to 2010 data.	**Supported**	Figure 1 – forest size decreased from 1991-2001 but increased slightly from 2001-2010
H2: Rate of forest loss will be dependent on the formal protection of the Kaya; gazetted or un-gazetted.	**None**	No significant effect of management status on forest area
H3: There will be a significant change in the length of forest edge exposed to matrix in 2001 compared to 2010.	**Supported**	Forest edge differed significantly between 1991 - 2001 and 2001-2010, forest edges are decreasing in length (Figure 2)
H4: There will be a reduction in forest health in Kaya forests, indicated by reduced frequency of Colobus monkeys, and increases in forest disturbance, indicated by felled or cut trees.	**None**	There was no change in Colobus density from 1991-2001 and 2001-2010 (Figure 3)
H5: A reduction in health will be dependent on the formal protection of the Kaya; gazetted or un-gazetted.	**None**	No significant effect of management status Colobus (Figure 4) or on disturbance (Figure 6)
H6: Local community opinions are in line with empirical l findings.	**None**	Kaya elders opinions differed from findings of the census and mapping data (Table 3, 4 and 5)

5.3.Censes

5.3.1 Colobus

Four forests did not have resident Colobus monkeys in this study. This contrasts with the findings from the semi-structured questionnaires which confirmed that Colobus had been sighted in the forest fragments in the immediate past i.e. less than two weeks previously. This either suggests that this study could have underestimated the Colobus abundance, or that the respondents exaggerated the time scale. The former seems more likely. Unless the colobus monkeys sighted by the respondents were actually seen in areas surrounding the forest fragments. From the literature, it has been shown that Colobus are able to utilise the matrix of habitat surrounding the forest fragments. Colobus sighted by respondents could have been using the fragments as corridors to other forests. Indeed, this has the same implications for the Colobus recorded during the census. This issue highlights the limitations of employing only a single methodology in attempting to estimate Colobus density.

Anderson, in her 2001 census, found lower Colobus densities in larger forests. This was also found in the present study, there was little difference in abundance between patches resulting in very high densities of Colobus in small patches. This would suggest that perhaps in larger forest fragments, it became harder to see the Colobus monkeys, or there was more space for the Colobus to hide in the canopy and be passed unnoticed. Another possibility is that the Colobus use the surrounding habitat matrix and other forest fragments extensively suggesting that the forest fragments are too small.

It is important to take into consideration that the majority of the forest fragments surveyed in this study were small (>1 km^2), however there were a couple of forests with a much larger forest size which could influence the results substantially.

Conversly Wong & Sicotte (2006) found no relationship between density of Colobus (*Colobus vellerosus*) and fragment size. It was postulated that food availability was the main factor in influencing Colobus density. However, in the present study the underline cause for the variation could not be established, the resource available did not allow for the investigation of the effect of food tree abundance and is therefore an important factor to consider in the future. In some primate studies, such as that by Zunino *et al.* (2007) population density of the black and gold howler monkey (*Alouatta caraya)* remained constant over a 20 year period, even in the face of deforestation. As described in the background, the influence of extinction debt may be at work in cases like this, or perhaps, the species is resilient enough not to be effected at all by habitat loss. The former, seems like a more reasonable explanation.

5.3.2 Ratio of area: perimeter

A higher density of Colobus was present in forests with a high ratio of area and forest perimeter. This can be explained because *C. a palliatus* exploits succession food resources such as young leaves, lianes and vines which result when edges are created in forest fragments. Folivorous primates sometimes increase following low levels of disturbance, such as near forest edges, in response to growth of high quality food (Zunino *et al.*, 2007). Chapman *et al.* (2007) found that black-and-white Colobus monkeys (*Colobus guereza*) exhibit considerable flexibility and prosper in degraded landscapes. This is supported by Wong & Sicotte (2006) who reported that disturbance had no effect on *Colobus vellerosus* frequency and were observed in higher densities in lightly logged areas compared to unlogged areas, confirming the ability of Colobus to flourish in disturbed areas (Wong and Sicotte, 2006). Furthermore, Flashing (2002) provides evidence that in *Colobus guerezas* are resilient to moderate degradation. However, it is important to emphasise that although black-and-white Colobus are not so affected by disturbance, this is not a universal finding; and should not be misused to argue that habitat degradation is not a threat to other primates. For example if disturbance levels are sufficiently high, it can actually cause *Colobus guerezas* to completely disappear (Chapman, 2003).

It may be the case that forest edges are beneficial for *C.a. pillatus* however, the underlying cause, fragmentation, has been shown to be highly detrimental to monkey health. Martínez-Mota (2007) found that black howler monkeys *(Alouatta pigra)* in fragmented forest patches had higher faecal cortisol metabolite levels which indicate long-term detrimental effects on fertility and ultimately survival.

5.3.1 Interaction between year and status

An interaction was found between the year of the census and the forest management status and Colobus density. Colobus was initially higher in gazetted forests but are now declining and Colobus in un-gazetted forests are increasing. Colobus could be increasing in un-gazetted Kaya's because *C.a. pillatus* find disturbed habitats preferable. If un-gazetted forests continue or increase in the disturbance rates, this will be potentially threatening to Colobus monkeys, the extent to which Colobus can withstand disturbance is not infinite. The fact that colobus are declining in gazetted forests is devastating and suggests that actively deterring habitat threats does not increase the abundance of *C.a. palliatus* .It would suggest that the environmental conditions are decreasing in these forests.

5.4 Mapping:

5.4.1 Forest cover

There was a significant change in forest area from the time period 1989-2001 and 2001-2010. Forests, on average, are getting bigger. This is a positive finding. However, although not significant, Gazetted forests are increasing in size and un-gazetted forests are decreasing (Figure 5). Certain forests, including Kaya Muhaka and Kaya Kinondo which increased in size over the time period have established conservation initiatives implemented in them. Currently only Kaya Kinondo has a form of income generating activities in the form of an eco-tour of the Kaya which was established by WWF in 2002. Kaya Muhaka is in partnership with 'Camp Kenya' and activities such as replanting and conservation education occurs with both local children international volunteers. This is interesting because both of these forests did indeed increase in size, perhaps one of the reasons for their increase was increasing the involvement of conservation initiatives in the surrounding areas and with the local community.

On average the ratio of perimeter length and area decreased between the periods of 1989 to 2001 and 2001 to 2010. Although as discussed above, Colobus thrive in a high ration of forest area to perimeter ratio, and this finding will most likely be a negative responses, this finding will be beneficial for many species including birds, for which edge effects frequently have negative consequences due to increased predation at forest edges, and negative avoidance of open habitat by forest interior.

5.5 Questionnaires

Surprisingly, from the questionnaires it appears that Kaya elders were not in tune with changes within the forest fragments. To ensure the elders who participated in the survey were able to make valid assessments of these changes, the question 'How often do you enter the forest?' was asked to determine if elders were familiar with the forests in question. All responses were relevant as most visited weekly, whilst others visited at least within one month. A certain amount of imprecision from locally based reporting's to be expected, even if the participants are not purposely misreporting. However more problematic is the deliberate misreporting, this may arise when there is a conflict of interest or if information is concealed from other members of the community. This could render data useless and therefore local motivations should be assessed prior to monitoring. This could be a potential problem with the data collected in this study using the semi-structured questionnaires.

Finally, 100% of elders said they need more support from external sources. As discussed above, Kaya Kinondo is the only Kaya with the capacity to generate an income to provide support for its conservation. This unanimous agreement suggests that the NMK need to do more conservation work in the Kaya's, and with the involvement of the Kaya elders

5.6 Management and policy implications

The management status of the forests has no effect on area change, disturbance and Colobus counts. Therefore the protection status of the forest is deemed ineffective because gazetted Kaya's exhibit a decrease in forest size as well as the presence of disturbances which should be prohibited under this management regime. Perhaps the deterrents for breaking the rules in the Kaya forest are not strong enough to stop deforestation or perhaps the poverty in some areas is high enough to warrant exploiting the forests even in the face of either a fine, administrative court of even jail. Chapman *et al.* 2007 suggested that unprotected forest fragments in areas with high human population density and economic growth, such as the coastal forests in Kenya are likely to be most threatened; perhaps these high densities of human populations and need for resources has resulted in a disregard for management status, this is emphasised by the fact that 45% of the population in the Kwale district is living in absolute poverty (WWF-UK, 2005). These high levels of poverty mean that forest-adjacent communities are highly dependent on forest resources for their daily needs (WWF-UK, 2005).

Finally, in order to make management status of forests more effective, perhaps higher forms of legal action should be applied.

5.7 Conservation implications

From the results obtained in this study, focal, priority areas for conservation research have been identified as Kaya Muhaka and Kaya Kinondo to build upon already existing foundations for conservation. Chale Island exhibits the highest potential for conservation initiatives. Regarding Chale Island, the development of an upper class international retreat makes it potentially a good base for ecotourism. It is a remote island, and the only inhabitants will be tourists which offers an ideal opportunity to raise awareness internationally and to generate external funds for monitoring effort, and for the conservation of *C. a. palliatus* and Kaya forests. This would be the only way Colobus conservation can benefit in the face of increasing tourist development in the remaining habitat fragments in Kenya.

Figure 12: Chale Island. From this image the tourist resort is visible. The rest of the island is a gazetted Kaya forest (Safari Expedition, 2003).

Finally, Kaya Mkangani and Dzipha elders were the most able to successfully recognise the changes in trends with both Colobus monkeys and forests sizes. This suggests these Kaya's exhibit a high potential for community based conservation and perhaps the development of locally based monitoring.

Improving public awareness is essential; the social surveys clearly showed a need for an awareness campaign to highlight the real trends occurring in forests and ways in which to minimise forest loss, increase Colobus abundance and minimise disturbances. If the local community is unaware how the forests surrounding them are fairing then this could result in unsustainable resource use. Although regular meetings are held with the Kaya elders, and between elders from different Kaya's, the financial support to make changes to Kaya management is absent. There does not seem to be any way to rectify this with the NMK funds limited and with the high demands and responsibility put upon WWF already, large scale conservation effort on the Kaya's currently seems unrealistic. Education is also essential in the protection of the forest fragment. As mentioned earlier, Kaya Muhaka has a local education component tied to 'Camp Kenya'. It would be beneficial for more conservation initiatives such as these to develop. Both awareness raising and education could be addressed if an appropriate locally based monitoring programme could be initiated. Biological monitoring and resource status alone is not sufficient to fully achieve conservation goals.

5.8 Limitations

Colobus angolensis palliatus has been suggested to be a key indicator species of forest health. However, in my research I have discovered that this may not necessarily be the case. *Colobus agolensis palliatus* may not be representative of the health of the entire ecosystem, however this does not invalidate the research carried out in this study. The decline of this charismatic, flagship species is an emotive issue which motivated members of the public to show an interest in biodiversity and conservation. This is reflected in the existence of the 'Colobus trust' which is dedicated to the conservation of Colobus, Sykes and Ververts in the coastal forests of Kenya. Furthermore there is no doubt that Colobus can act as an umbrella species as Colobus monkeys dwell in exceedingly threatened habitats and conservation of these areas will, as in the case of the spotted owl described in the background section, have the capacity to conserve other organisms in the same area. As discussed in the background section of this thesis, monitoring a particular species, such as *C. a. palliatus*, will be useful to building up a picture of how organisms respond to current threats and are an important part of developing management practices. Long-term species and habitat monitoring are essential. This is especially important in the case of the effects of primates in fragmented forests- due to the high variability in primate responses to changes in their environments. Crucially it directs future research away from the potentially misleading indicator species concept and towards meaningful long term monitoring of a scientifically interesting threatened species. Therefore, although the species may not give a complete representation of the health of the Kaya forests, the monitoring and understanding of its ecology is useful to determine threats in the coastal forests.

The present study was limited in its remit and power by resources. Had sufficient resources and time been available a larger sample size of Kaya forests would have yielded a higher statistical power throughout the data analyses section of the project. Also Kaya Ganda was surveyed during the study period, but due to technical difficulties in the field the mapping data for this Kaya was unfortunately lost and the work was consequently repeated by WWF researches .This could be potentially less accurate as the precision from a GPS.

Bias could have arisen during the semi-structured questionnaires due to the nature of the questions answered. People may have overestimated the threats to the forests and Colobus abundance in order to get more resources for management.

5.9 Future research

To continue the monitoring of *C. a palliatus*, another census should be carried out in approximately 10 years time. To increase the understanding of the distribution of *C. a palliatus*, the remaining forests in the Kwale region should be investigated. This includes the toporegions Mombassa, Msambweni and Ndavaya. This will give a more robust analysis of the Kaya forests and increase the sample size of the study. Furthermore the study should extend to other types of forest management; it would be beneficial for forest reserves as well as unprotected areas to be surveyed to broaden the study and to make it more applicable to global studies.

Finally, as previously mentioned, the development of a research methodology that incorporates the distribution of food tree species, as well as tree canopy cover and vegetation type, with survey methods employed in the present research would contribute significantly to the robustness of the study, making the findings more meaningful. This is emphasised by Umpathy & Kumar 2003 (Umpathy, 2003) who found that the occurrence of the lion-tailed macaque and the Nilgiri langur in forest fragments in India were related to area, canopy height and tree diversity; these parameters were not accounted for in this study and could be important variables to explore in further study. Furthermore Rodriguez-Luna *et al.* 2003 (Rodríguez-Luna, 2003) is in accordance with this finding, and suggest that the dynamics of howler monkey populations (*alouatta palliate Mexicana*) can be linked to the distribution and abundance of food resources. Furthermore Davies (1994) points out that the environmental factor most frequently shown to limit herbivore populations is food supply. This is an opportunity for future research.

Interestingly, during the semi-structured questionnaires kaya elders stated that in their opinion, one of the principle threats to Colobus was actual dehydration during the dry season and this would appear to be a limiting factor which should be explored in any further research. In other studies, lack of food resources has been deemed the principle reason for primate decline (Rodríguez-Luna, 2003, Umpathy, 2003).

References

ANDERSON, J. (2004) Habitat fragmentation and metapopulation dynamics of the Angola black-and-white colobus (*Colobus angolensis palliatus)* in coastal Kenya. PhD thesis, University College London.

ANDERSON, J., COWLISHAW, G. & ROWCLIFFE, J. M. (2007a) Effects of forest fragmentation on the abundance of *Colobus angolensis palliatus* in Kenya's coastal forests. *International Journal of Primatology,* **28,** 637-655.

ANDERSON, J., ROWCLIFFE, J. & COWLISHAW, G. (2007b) The Angola black-and-white colobus (*Colobus angolensis palliatus*) in Kenya: historical range contraction and current conservation status. *American Journal of Primatology,* **69,** 664-680.

ANZURES-DADDA, A. & MANSON, R. H. (2007) Patch- and landscape-scale effects on howler monkey distribution and abundance in rainforest fragments. *Animal Conservation,* **10,** 69-76.

ARROYO-RODRIGUEZ, V. & MANDUJANO, S. (2009) Conceptualization and measurement of habitat fragmentation from the primates' perspective. *International Journal of Primatology,* **30,** 497-514.

ASNER, G. P., ELMORE, A. J., OLANDER, L. P., MARTIN, R. E. & HARRIS, A. T. (2004) Grazing ststems, ecosystem responses, and global change. *Annual Review of Environment and Resources,* **29,** 261-299.

BEGON, M., HARPER, J.L., TOWNSEND. (1990) *Ecology: Individuals, populations and communities*, Blackwell Scientific Publications, London.

BHAGWAT, S. A. & RUTTE, C. (2006) Sacred groves: potential for biodiversity management. *Frontiers in Ecology and the Environment,* **4,** 519-524.

BONHAM, C., SACAYON, E. & TZI, E. (2008) Protecting imperiled “paper parks”: potential lessons from the Sierra Chinajá, Guatemala. *Biodiversity and Conservation,* **17,** 1581-1593.

BRADSHAW, C. J., SODHI, N. S. & BROOK, B. W. (2009) Tropical turmoil: a biodiversity tragedy in progress. *Frontiers in Ecology and the Environment,* **7,** 79-87.

BROOKS, T. M., BAKARR, M. I., BOUCHER, T. I. M., DA FONSECA, G. A. B., HILTON-TAYLOR, C., HOEKSTRA, J. M., MORITZ, T. O. M., OLIVIERI, S., PARRISH, J., PRESSEY, R. L., RODRIGUES, A. S. L., SECHREST, W. E. S., STATTERSFIELD, A. L. I., STRAHM, W. & STUART, S. N. (2004) Coverage provided by the global protected-area system: is it enough? *BioScience,* **54,** 1081-1091.

BURGESS, N. D., CLARKE, G. P. & RODGERS, W. A. (1998) Coastal forests of eastern Africa: status, endemism patterns and their potential causes. *Biological Journal of the Linnean Society,* **64,** 337-367.

BURGESS, N. D., CLARKE, G.P. (ed.) (2000) *Coastal Forests of Eastern Africa*. IUCN, Cambridge, UK.

CAMPBELL, M. O. (2005) SACRED GROVES FOR FOREST CONSERVATION IN GHANA'S COASTAL SAVANNAS: ASSESSING ECOLOGICAL AND SOCIAL DIMENSIONS. *Singapore Journal of Tropical Geography,* **26,** 151-169.

CARIGNAN, V. & VILLARD, M.-A. (2002) Selecting indicator species to monitor ecological integrity: a review. *Environmental Monitoring and Assessment,* **78,** 45-61.

CBD. (2010) *Article 10. Sustainable Use of Components of Biological Diversity* [Online]. Available: http://www.cbd.int/convention/articles.shtml?a=cbd-10
[Accessed 14th August 2010].

CHAPIN III, F. S., ZAVALETA, E. S., EVINER, V. T., NAYLOR, R. L., VITOUSEK, P. M., REYNOLDS, H. L., HOOPER, D. U., LAVOREL, S., SALA, O. E., HOBBIE, S. E., MACK, M. C. & DIAZ, S. (2000) Consequences of changing biodiversity. *Nature,* **405,** 234-242.

CHAPMAN, C. A., LAWES, M.J., NAUGHTON-TREVES, L., GILLESPIE, T. (2003) Primate survival in community-owned forest fragments: Are metapopulation models useful amidst intensive use? *In:* MARSH, L. K. (ed.) *Primates in fragments: ecology and conservation.* Kluwer Academic/Plenum publishers., New York.

CHAPMAN, C. A., NAUGHTON-TREVES, L., LAWES, M. J., WASSERMAN, M. D. & GILLESPIE, T. R. (2007) Population declines of colobus in Western Uganda and conservation value of forest fragments. *International Journal of Primatology,* **28,** 513-528.

CHOUIN, G. (2002) Sacred Groves in History. *IDS Bulletin,* **33,** 39-46.

COLDING, J. & FOLKE, C. (2001) Social taboos: “invisible” systems of local resource management and biological conservation. *Ecological Applications,* **11,** 584-600.

COSTANZA, R., D'ARGE, R., DE GROOT, R., FARBER, S., GRASSO, M., HANNON, B., LIMBURG, K., NAEEM, S., O'NEILL, R. V., PARUELO, J., RASKIN, R. G., SUTTON, P. & VAN DEN BELT, M. (1997) The value of the world's ecosystem services and natural capital. *Nature,* **387,** 253-260.

CRAUL, M., CHIKHI, L., SOUSA, V., OLIVIERI, G. L., RABESANDRATANA, A., ZIMMERMANN, E. & RADESPIEL, U. (2009) Influence of forest fragmentation on an endangered large-bodied lemur in northwestern Madagascar. *Biological Conservation,* **142,** 2862-2871.

CRAWLEY, J. M. (2005) *Statistics: an introduction using R*, John Wiley & Sons, Ltd, Chichester.

CURRAN, L. M., TRIGG, S. N., MCDONALD, A. K., ASTIANI, D., HARDIONO, Y. M., SIREGAR, P., CANIAGO, I. & KASISCHKE, E. (2004) Lowland forest loss in protected areas of Indonesian Borneo. *Science,* **303,** 1000-1003.

DANGWAL, D. D. (2005) Commercialisation of forests, timber extraction and deforestation in Uttaranchal, 1845-1947. *Conservation & Society,* **3,** 110-133.

DANIELSEN, F., BEUKEMA, H., BURGESS, N. D., PARISH, F., BRÜHL, C. A., DONALD, P. F., MURDIYARSO, D., PHALAN, B., REIJNDERS, L., STRUEBIG, M. & FITZHERBERT, E. B. (2009) Biofuel plantations on forested lands: double jeopardy for biodiversity and climate. *Conservation Biology,* **23,** 348-358.

DANIELSEN, F., BURGESS, N. D. & BALMFORD, A. (2005) Monitoring matters: examining the potential of locally-based approaches. *Biodiversity and Conservation,* **14,** 2507-2542.

DUDLEY, N., HIGGINS-ZOGIB, L. & MANSOURIAN, S. (2009) The links between protected areas, faiths, and sacred natural sites. *Conservation Biology,* **23,** 568-577.

DUNN, J. C., CRISTÓBAL-AZKARATE, J. & VEÀ, J. J. (2009) Differences in diet and activity pattern between two groups of *Alouatta palliata* associated with the availability of big trees and fruit of top food taxa. *American Journal of Primatology,* **71,** 654-662.

ESTRADA, A. & COATESESTRADA, R. (1996) Tropical rain forest fragmentation and wild populations of primates at Los Tuxtlas, Mexico. *International Journal of Primatology,* **17,** 759-783.

EWERS, R. M. & RODRIGUES, A. S. L. (2008) Estimates of reserve effectiveness are confounded by leakage. *Trends in Ecology & Evolution,* **23,** 113-116.

FAHRIG, L. (2003) Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology Evolution and Systematics,* **34,** 487-515.

FEARNSIDE, P. M. (2001) Soybean cultivation as a threat to the environment in Brazil. *Environmental Conservation,* **28,** 23-38.

FISCHER, J. & LINDENMAYER, D. B. (2007) Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography,* **16,** 265-280.

FITZHERBERT, E. B., STRUEBIG, M. J., MOREL, A., DANIELSEN, F., BRÜHL, C. A., DONALD, P. F. & PHALAN, B. (2008) How will oil palm expansion affect biodiversity? *Trends in Ecology & Evolution,* **23,** 538-545.

FASHING, P. J. (2002) Population status of black and white colobus monkeys (*Colobus guereza*) in Kakamega Forest, Kenya: are they really on the decline? *African Zoology,* **37,** 119-126.

FOLEY, J. A., DEFRIES, R., ASNER, G. P., BARFORD, C., BONAN, G., CARPENTER, S. R., CHAPIN, F. S., COE, M. T., DAILY, G. C., GIBBS, H. K., HELKOWSKI, J. H., HOLLOWAY, T., HOWARD, E. A., KUCHARIK, C. J., MONFREDA, C., PATZ, J. A., PRENTICE, I. C., RAMANKUTTY, N. & SNYDER, P. K. (2005) Global Consequences of Land Use. *Science,* **309,** 570-574.

FRANKLIN, A. B., NOON, B. R. & GEORGE, T. L. (2002) What is habitat fragmentation? *Studies in Avian Biology,* **25**, 20-29.

GRAU, H. R. & AIDE, M. (2008) Globalization and land-use transitions in Latin America. *Ecology and Society,* **13**.

HAILA, Y. (2002) A conceptual genealogy of fragmentation research: from island biogeography to landscape ecology. *Ecological Applications,* **12,** 321-334.

HARPER, K. A., MACDONALD, S. E., BURTON, P. J., CHEN, J., BROSOFSKE, K. D., SAUNDERS, S. C., EUSKIRCHEN, E. S., ROBERTS, D., JAITEH, M. S. & ESSEEN, P. A. (2005) Edge Influence on Forest Structure and Composition in Fragmented Landscapes. *Conservation Biology,* **19,** 768-782.

HOCKLEY, N. J., JONES, J. P. G., ANDRIAHAJAINA, F. B., MANICA, A., RANAMBITSOA, E. H. & RANDRIAMBOAHARY, J. A. (2005) When should communities and conservationists monitor exploited resources? *Biodiversity and Conservation,* **14,** 2795-2806.

JAMES, R. A., LEBERG, P. L., QUATTRO, J. M. & VRIJENHOEK, R. C. (1997) Genetic diversity in black howler monkeys (*Alouatta pigra*) from Belize. *American Journal of Physical Anthropology,* **102,** 329-336.

JOPPA, L. N., LOARIE, S. R. & PIMM, S. L. (2008) On the protection of "protected areas". *Proceedings of the National Academy of Sciences,* **105,** 6673-6678.

KARERE, G. M., OGUGE, N. O., KIRATHE, J., MUORIA, P. K., MOINDE, N. N. & SULEMAN, M. A. (2004) Population sizes and distribution of primates in the lower Tana River forests, Kenya. *International Journal of Primatology,* **25,** 351-365.

KINGDON, J., STRUHSAKER, T., OATES, J., HART, J.F., BUTYNSKI, T.M., DE JONG, Y. & GROVES, C.P. (2008) *Colobus angolensis ssp. palliatus* [Online]. IUCN. Available: www.iucnredlist.org [Accessed 12 August 2010].

KUUSSAARI, M., BOMMARCO, R., HEIKKINEN, R. K., HELM, A., KRAUSS, J., LINDBORG, R., OCKINGER, E., PARTEL, M., PINO, J., RODA, F., STEFANESCU, C., TEDER, T., ZOBEL, M. & STEFFAN-DEWENTER, I. (2009) Extinction debt: a challenge for biodiversity conservation. *Trends in Ecology & Evolution,* **24,** 564-571.

LAMBIN, E. F., GEIST, H. J. & LEPERS, E. (2003) Dynamics of land-use and land-cover change in tropical regions. *Annual Review of Environment and Resources,* **28,** 205-241.

LANDE, R. (1998) Anthropogenic, ecological and genetic factors of extinction. *In:* MACE, G. M., BALMFORD, A., GINSBERG, J.R. (ed.) *Conservation in a changing world.* Cambridge University Press, Cambridge.

LANDRES, P. B., VERNER, J. & THOMAS, J. W. (1988b) Ecological uses of vertebrate indicator species: a critique. *Conservation Biology,* **2,** 316-328.

LAPORTE, N. T., STABACH, J. A., GROSCH, R., LIN, T. S. & GOETZ, S. J. (2007) Expansion of industrial logging in Central Africa. *Science,* **316,** 1451-1451.

LAURANCE, W. F. (2004) Forest-climate interactions in fragmented tropical landscapes. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences,* **359,** 345-352.

LAURANCE, W. F., LOVEJOY, T. E., VASCONCELOS, H. L., BRUNA, E. M., DIDHAM, R. K., STOUFFER, P. C., GASCON, C., BIERREGAARD, R. O., LAURANCE, S. G. & SAMPAIO, E. (2002) Ecosystem decay of Amazonian forest fragments: A 22-year investigation. *Conservation Biology,* **16,** 605-618.

LINDENMAYER, D. B. (1999) Future directions for biodiversity conservation in managed forests: indicator species, impact studies and monitoring programs. *Forest Ecology and Management,* **115,** 277-287.

LOVETT, J. C., WASSER, S.K. (ed.) (1993) *Biography of Ecology of the Rainforests of Eastern Africa.* Cambridge University Press, Cambridge.

MACARTHUR, R. H., WILSON, E.O. (ed.) (1967) *The theory of island biogeography.* Princeton University Press. Princenton, NJ.

MARSH, L. K. (ed.) (2003) *Primates in Fragments: ecology and conservation.* Kluwer Academic/ Plenum Publishers, New York.

MARTÍNEZ-MOTA, R., VALDESPINO, C., SÁNCHEZ-RAMOS, M. A. & SERIO-SILVA, J. C. (2007) Effects of forest fragmentation on the physiological stress response of black howler monkeys. *Animal Conservation,* **10,** 374-379.

MATIKU, P. (2004) *The Coastal Forests of Kenya: Forest data, threats, socio-economic issues, values, stakeholders, challanges, strategies, investment and enabling enviroment*. WWF-EARPO Eastern Africa Coastal Forests Ecoregion Programme, Nairobi.

MBORA, D. N. M. & MEIKLE, D. B. (2004) Forest fragmentation and the distribution, abundance and conservation of the Tana river red colobus (*Procolobus rufomitratus*). *Biological Conservation,* **118,** 67-77.

MGUMIA, F. H. & OBA, G. (2003) Potential role of sacred groves in biodiversity conservation in Tanzania. *Environmental Conservation,* **30,** 259-265.

MILNER-GULLAND, E. J., ROWCLIFFE J.M. (ed.) (2007) *Conservation and Sustainable Use: A Handbook of Technique.* University Press, Oxford.

MURCIA, C. (1995) Edge effects in fragmented forests – implications for conservation. *Trends in Ecology & Evolution,* **10,** 58-62.

MYERS, N. (1990) The biodiversity challange: expanding hot-spots analysis. *Environmentalist,* **10,** 243-256.

MYERS, N. (1991) Tropical forests – present status and future outlook. *Change,* **19,** 3-32.

NAGENDRA, H. (2008) Do parks work? Impact of protected areas on land cover clearing. *Ambio,* **37,** 330-337.

NOSS, R. F. (1990) Indicators for monitoring biodiversity: a heirarchical approach. *Conservation Biology,* **4,** 355-364.

OATES, J. F. (1996) Habitat alteration, hunting and the conservation of folivorous primates in African forests. *Australian Journal of Ecology,* **21,** 1-9.

OATES, J. F., ABEDI-LARTEY, M., MCGRAW, W. S., STRUHSAKER, T. T. & WHITESIDES, G. H. (2000) Extinction of a West African red colobus monkey. *Conservation Biology,* **14,** 1526-1532.

OATES, J. F., DAVIES, A.G. (1994a) What are the colobines? *In:* OATES, J. F., DAVIES, A.G. (ed.) *Colobine Monkeys.* Cambridge University Press, Cambridge.

OATES, J. F., DAVIES, A.G., DELSON, E. (1994b) The diversity of living colobines. *In:* OATES, J. F., DAVIES, A.G. (ed.) *Colombine Monkeys.* Cambridge University Press, Cambridge.

ONDERDONK, D. A. & CHAPMAN, C. A. (2000) Coping with Forest Fragmentation: The Primates of Kibale National Park, Uganda. *International Journal of Primatology,* **21,** 587-611.

PIMM, S. L. & RAVEN, P. (2000) Biodiversity - Extinction by numbers. *Nature,* **403,** 843-845.

RIST, J., MILNER-GULLAND, E. J., COWLISHAW, G. & ROWCLIFFE, M. (2010) Hunter reporting of Catch per unit effort as a monitoring tool in a bushmeat-harvesting system. *Conservation Biology,* **24,** 489-499.

ROBERTSON, S. A., LUKE, W.R.Q. (1993) *Kenya Coastal Forests: The report of the NMK/WWF Coast Forest Survey*. National Museums of Kenya, Nairobi.

RODGERS, W. A., BURGESS, N.D. (2000) Coastal forest conservation problems. *In:* BURGESS, N. D., CLARKE, G.P. (ed.) *Coastal forests of Eastern Africa.* IUCN Publications Services Unit, Cambridge.

RODRÍGUEZ-LUNA, E., DOMÍNGUEZ-DOMÍNGUEZ, L.E., MORALES-MÁVIL J.E., MARTÍNEZ-MORALES, M. (2003) Foraging strategy changes in an *Alouatta palliata mexicana* troop released on an island. *In:* MARSH, L. K. (ed.) *Primates in fragments: ecology and conservation.* Kluwer Academic/Plenum Publishers, New York.

SAFARI EXPEDITION (2003) *The Sands at Chale, Chale Island, South Coast, Kenya* [Online]. Available: http://www.extreme-safari.com/campsandlodges/southcoast/sands_at_chale.html [Accessed 20th August 2010].

SCHMITT, C. B., BURGESS, N. D., COAD, L., BELOKUROV, A., BESANÇON, C., BOISROBERT, L., CAMPBELL, A., FISH, L., GLIDDON, D., HUMPHRIES, K., KAPOS, V., LOUCKS, C., LYSENKO, I., MILES, L., MILLS, C., MINNEMEYER, S., PISTORIUS, T., RAVILIOUS, C., STEININGER, M. & WINKEL, G. (2009) Global analysis of the protection status of the world's forests. *Biological Conservation,* **142,** 2122-2130.

SCHWARTZ, M. W., BRIGHAM, C. A., HOEKSEMA, J. D., LYONS, K. G., MILLS, M. H. & VAN MANTGEM, P. J. (2000) Linking biodiversity to ecosystem function: implications for conservation ecology. *Oecologia,* **122,** 297-305.

SEABLOOM, E. W., DOBSON, A. P. & STOMS, D. M. (2002) Extinction rates under nonrandom patterns of habitat loss. *Proceedings of the National Academy of Sciences of the United States of America,* **99,** 11229-11234.

SHEARMAN, P. L., ASH, J., MACKEY, B., BRYAN, J. E. & LOKES, B. (2009) Forest conversion and degradation in Papua New Guinea 1972–2002. *Biotropica,* **41,** 379-390.

SIMBERLOFF, D. (1999) The role of science in the preservation of forest biodiversity. *Forest Ecology and Management,* **115,** 101-111.

SPECTOR, S., FORSYTH, A.B. (1998) Indicator taxa for biodiversity assessment in the vanishing tropics. *In:* MACE, G. M., BALMFORD, A., GINSBERG, J.R. (ed.) *Conservation in a changing world.* Cambridge University Press, Cambridge.

STRUHSAKER, T. T. (1976) A further decline in numbers of Amboseli vervet monkeys. *Biotropica,* **8,** 211-214.

TENGEZA, A. H. (2003) Traditional leadership and gender: handling controversies in sacred forests. *Compas Magazine***, 6,** 29-30.

TILMAN, D., MAY, R. M., LEHMAN, C. L. & NOWAK, M. A. (1994) Habitat destruction and the extinction debt. *Nature,* **371,** 65-66.

TINGA, K. K. (2004) The presentation and interpretation of ritual sites: The Mijikenda Kaya case (A pilot study by the National Museums of Kenya). *Museum International,* **56,** 8-14.

UMPATHY, G., KUMAR, A. (2003) Impacts of forest fragmentation on lion-tailed Macaque and *Nilgiri langur* in Western Ghats, South India. *In:* MARSH, L. K. (ed.) *Primates in fragments: ecology and conservation.* Kluwer Academic/Plenum Publishers, New York.

VELLEND, M., VERHEYEN, K., JACQUEMYN, H., KOLB, A., VAN CALSTER, H., PETERKEN, G. & HERMY, M. (2006) Extinction debt of forest plants persists for more than a century following habitat fragmentation. *Ecology,* **87,** 542-548.

VERÍSSIMO, A., BARRETO, P., TARIFA, R., UHL, C. (1995) Extraction of a high- value natural resource in Amazonia: the case of mahogany. *Forest Ecology and Management,* **72,** 39-60.

WAHUNGU, G. M., MUORIA, P. K., MOINDE, N. N., OGUGE, N. O. & KIRATHE, J. N. (2005) Changes in forest fragment sizes and primate population trends along the River Tana floodplain, Kenya. *African Journal of Ecology,* **43,** 81-90.

WHITESIDES, G. H., OATES, J. F., GREEN, S. M. & KLUBERDANZ, R. P. (1988) Estmating primate densities from transects in a West-African rain-forest – a comparison of techniques. *Journal of Animal Ecology,* **57,** 345-367.

WONG, S. N. P. & SICOTTE, P. (2006) Population size and density of *Colobus vellerosus* at the Boabeng-Fiema Monkey Sanctuary and surrounding forest fragments in Ghana. *American Journal of Primatology,* **68,** 465-476.

WRIGHT, S. J. (2005) Tropical forests in a changing environment. *Trends in Ecology & Evolution,* **20,** 553-560.

WRIGHT, S. J. (2010) The future of tropical forests. *Annals of the New York Academy of Sciences,* **1195,** 1-27.

WWF-UK, (2005) Eastern African Coastal Forest Ecoregion Programme Document.

WWF (2009) *WWF Eastern Africa Coastal Forests Conservation Ecoregion Program – an atlas of the coastal forests*. Eds*.,* SALEHE, J., ONG'ANDA, H., BURGESS, N. pp 89.

ZUNINO, G. E., KOWALEWSKI, M. M., OKLANDER, L. I. & GONZALEZ, V. (2007) Habitat fragmentation and population size of the black and gold howler monkey (*Alouatta caraya*) in a semideciduous forest in northern Argentina. *American Journal of Primatology,* **69,** 966-975.

Appendix 1.

Questionnaire

Age category <16/ 16-25/ 25-40/ 40-60/ >60 **Sex:** M /F

Birth place: **Profession:**

Education level: **Name of nearest Kaya:**

How often do you visit the Kaya? Every day/ Three times a week/ Once a week/ Once every two weeks/ once a month/ more than once a month

1. Forest health – Colobus monkeys

1.1 Have you ever seen Colobus monkeys in the Kaya? Yes/ No

1.1.1. - If yes, When was the last time you saw a Colobus monkey? Today/ Yesterday/ Last Three days/ Last week/ Last two weeks/ Last month/ Last three months/ Last six months/ Last year/ Last year +/ Other

1.2. Have you seen a change in Colobus number over the last ten years? Decrease/ stayed the same/ increase

1.3. Is there any conflict between Colobus and humans? Yes/ No

2. Forest health - Disturbance

2.1. What type of degradation (damage) do you see in the Kaya?

Deforestation/ Plantation/ Land grabbing/ Littering/ Hunting/ Snaring/ Quarry mining/ Dumping/ Slash and burn agriculture/ Charcoal burning/ Fire (external sources)/ Other

2.2. Have you noticed a change in amount of disturbance (damage) in the Kaya? Decrease/ Stayed the same/ Increase

2.3. Have you noticed a change in size of the Kaya? Decrease/ Stayed the same/ Increase

3. Management

3.1 Can support from external sources be improved (financial, resource, time ect.)? Yes/ No

4. Future conservation within the Kaya

4. Do you think without any further conservation action, the Kaya will disappear?

>6 months, 6-12 months, 1-2 years, 2-5 years, 5-10 years, 10-20, 20-50 years, 50 years +, Never

Appendix 2

Table x: Raw data used in the project

Kaya forest	Colobus abundance	Colobus density	Disturbance	Forest area	Forest perimeter	Percent of elders able to predict Colobus change	Percent of elders able to predict forest change
Tiwi	0	0.000	19	0.083	1151	0	0
Diani	14	124.608	9	0.112	1387	30	0
Ukunda	8	51.207	44	0.156	1694	30	100
Muhaka	20	13.600	74	1.471	6450	20	20
Ganzoni	6	66.827	0	0.090	413	60	30
Kinondo	20	133.319	9	0.150	1608	60	20
Timbwa	11	109.242	7	0.101	1484	60	0
Chale Island	8	50.157	2	0.160	1951	0	0
Muvmoni	Na	Na	23	0.104	1514	Na	Na
Mkangani	0	0.000	4	0.007	377	10	50
Dzipha	0	0.000	2	0.019	629	50	40
Ganda	13	670.103	2	0.040	1000	0	75
Jego	8	200.000	4	0.018	616	70	80
Dzirive	0	0.000	3	0.058	1069	30	10
Sega	7	120.275	37	0.183	2772	33	33
Vanga	0	0.000	8	0.025	688	77	11