MAKERERE UNIVERSITY

SELECTING SPECIES FOR ACTIVE AND PASSIVE RESTORATION OF DEGRADED FORESTS IN KIBALE NATIONAL PARK

BY

ERIC MUKAMA
13/U/8577/PS

A DISSERTATION SUBMITTED TO SCHOOL OF FORESTRY, ENVIRONMENTAL AND GEOGRAPHICAL SCIENCES IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE AWARD OF THE BACHELOR'S DEGREE IN CONSERVATION FORESTRY AND PRODUCTS TECHNOLOGY OF MAKERERE UNIVERSITY

JUNE 2018

DECLARATION

I Eric Mukama declare that this dissertation is my original work and has never been submitted to any other University or Institution for any degree award or its equivalent.

Signed: ..... Date: 06/06/2018.....

Eric Mukama (13/U/8577/PS)

This dissertation has been submitted with approval of my university supervisor;

Signed: ..... Date: 06th June 2018

Mr. Enock Ssekuubwa

Department of Forestry, Biodiversity and Tourism

School of Forestry, Environmental and Geographical Sciences

College of Agricultural and Environmental Sciences

Fuculty of Forestry and Nature Conservation

Makerere University

ssekuubwa@caes.mak.ac.ug

DEDICATION

This dissertation is dedicated to my family for standing by me in and out of season. They were very supportive and encouraged me even when the work became very tough to a point of despair. They held me up with prayers, sacrifice and patience and with powerful kind words throughout my education.

ACKNOWLEDGEMENT

I sincerely thank the Almighty God for granting me an opportunity to undertake my bachelor's degree studies at Makerere University, His provisions, and grace that has been sufficient upon my life. He has given me the insight, sound mind, energy, good health and the grace to come this far with this thesis. I sincerely appreciate whoever generously advised and supported me when I was writing this dissertation. My deepest appreciation and hearty thanks go to my supervisor, Mr. Enock Ssekuubwa, whose valuable guidance and constructive criticisms as well as tireless efforts in advising me throughout this report, have made the completion of this work possible. I am most grateful and would like to thank Mr. Enock Ssekuubwa from the Faculty of Forestry and Nature Conservation School of Forestry, Environmental and Geographical Sciences for his supervision, encouragement, guidance and support throughout my research. No matter how fixed he has been, he has always endeavored to be there for me whenever needed. I owe special gratitude to my family for their prayers, moral and financial support and all other people who in one way or another have contributed to the success of this study. My special thanks goes to my sister Naiwumbwe Susan for her financial support without which my studies couldn't have been a success. Special thanks also go to my fellow students for their encouragement and moral support during the whole period of my study.

TABLE OF CONTENTS

ABSTRACT

The use of native species in forest restoration has been increasingly recognized as an effective means of restoring ecosystem functions and biodiversity to degraded areas across the world. However, successful selection of species adapted to local conditions requires specific knowledge which is often lacking, especially in developing countries. In order to scale up forest restoration, experimental data on the responses of native species to propagation and restoration treatments across a range of local conditions are required. In this study, species that can be passively restored by natural regeneration were distinguished from those requiring active restoration. Tree species dominance was quantified (measured by an importance value index, IVIi) and used abundance–size correlations to select those species suitable for passive and/or active restoration of disturbed riparian vegetation in Kibale National Park. We sampled riparian vegetation in a 50×10–m plot in each of the sixty two sample plots in the digraded ecosystems. All the species which were regenerating and had a diameter of atleast 5cm were selected, and Pearson's rank correlation between abundance and diameter classes was calculated. For thirteen species, it was determined that passive restoration could be sufficient for their establishment. The remaining fifty twe species could be transplanted by means of active restoration. The high number of tree species found in the degraded ecosystem suggests that the species pool for ecological restoration is large. However, sampling in the degraded ecosysem helped to reduce the number of species that requires active restoration. Restoration objectives must guide the selection of which methods to implement; in different conditions, other criteria such as dispersal syndrome or social value could be considered in the species selection.

Key words: active and passive regeneration, ecological groups, recovery.

LIST OF ACRONYMS

%	Percent
°C	Degrees celcius
Cm	Centmetre
Dbh	Diameter at breast height
FAO	Food and Agriculture organization
FLR	Forest landscape restoration
IPBES	Intergovernmerntal Science-Policy Platform on Biodiversity and Ecosystem Services
IFER	Institute of Forest Eosystem Research
IVIi	Importance value index of a species
KNP	Kibale National Park
MWE	Ministry of water and environment
NPLD	Non pioneer light demanders
PION	Pioneer
rs	Correlation coefficincy
SAVA	Savanna
SHTO	Shade tolerant
SWAM	Swamp

CHAPTER ONE: INTRODUCTION

1.1 Background to the study

Understanding the complexity of forest regeneration has a considerable potential for future conservation of secondary forests. This involves selecting sites composed of secondary and primary forests to determine the capacity of a forest to regenerate from disturbance (Rocha *et al.,* 2018). This helps to reestablish the characteristic species assemblage in a degraded or destroyed ecosystem and appropriate community structure in a reference ecosystem. In the initial stage of designing any ecological restoration project, the first question should be whether the degraded ecosystem may recover spontaneously in a reasonable time period or active intervention is needed for particular species (Meli *et al.,* 2013). In this effort, assessing restoration outcomes through basic observational studies of long-term projects is key to evaluating the efficiency of alternative approaches (Meli *et al*., 2017).

In a recent global meta-analysis there was no difference between active and passive regeneration in terms of ecosystem recovery (Meli *et al*., 2017). Nonetheless, selection of the best restoration approach for particular species (natural regeneration versus active restoration) must consider a suite of biotic and abiotic factors known to affect the rate of recovery of different forest properties like species composition during restoration and regeneration, such as the amount and type of forest cover at the landscape scale, annual precipitation, and intensity of past disturbance or previous land use (Chazdon & Guariguata, 2016). This is achieved by comparison of different restoration approaches in separate study areas of varying ages to determine their success in restoration of different species (Meli *et al.,* 2017).

There are stories of successes associated with passive restoration of some species (Morrison & Lindell, 2011). Many tropical and humid temperate ecosystems can recover with little or no human intervention when the soil has not been severely degraded. This happens if the system crossed an ecological threshold resulting in a new stable, degraded state (Pardini *et al.,* 2010). In these cases, "cessation of activities that are causing degradation or preventing recovery" (passive restoration) (Andrade *et al.,* 2015) is enough to drive ecosystem recovery, and can be considered the first step in ecological restoration (Meli *et al.,* 2008). The forests recover quickly when the

impeding past land use (e.g., agriculture, logging) ceases and natural succession proceeds (Chazdon & Guariguata, 2016) (hereafter, passive restoration).

For passive restoration to be successful, site resilience and species performance have to interact with landscape and disturbance histories to dictate the succession trajectory of a forest plot (Chazdon, 2016, Pardini *et al.*, 2010). Both stochastic and deterministic processes drive the composition of secondary forests (Crouzeilles *et al.*, 2017). Stochastic processes such as chance dispersal can interact with deterministic processes including niche availability, local competition with early arriving species, and density dependent build-up of predators and pathogens, resulting in unpredictable changes in the species composition (Chase & Myers, 2011). This process can also be assisted through human interventions such as fencing to control livestock grazing, weed control, and fire protection (Crouzeilles *et al.*, 2017).

The sites with a high abundance of generalist species from the regional flora, high levels of seed dispersal, and the presence of near-by old-growth forest remnants facilitate passive regeneration. In these situations plant species richness quickly increases within the first few decades, especially for species with smaller sized stems (Bechara *et al.*, 2016) and later there can be a clear convergence with mature forest community composition, supporting an equilibrium model of succession (Ma & Letcher, 2017). However, tropical forests have a notoriously variable rate of recovery. Some stands recover structure rapidly and without human intervention within a couple of decades but in other cases may take even centuries. On the other hand, to recover pre-disturbance species composition, forests may take from one century up to thousands of years (Corbin & Holl, 2012).

However, in some cases, there are stories of complete failure associated with use of passive restoration to restore both composition and structure (Holl & Aide, 2011, Norden *et al.*, 2009, Suding, 2011) as some degraded tropical ecosystems may remain in a state of arrested forest succession (Ghazoul *et al.*, 2015) where passive restoration of some species fails in highly degraded landscapes (Chazdon, 2013). This implies that although passive restoration sometimes may be sufficient for some species, others need active restoration (Meli *et al*., 2013). Active restoration is often favored in such areas where natural regeneration is hindered, such as isolated

sites with extensive deforestation, low precipitation rates, and long history of intensive disturbances or land uses that led to severe soil degradation, weed infestation, or loss of the seed bank and root sprouts (Lennox *et al.*, 2011). Under high disturbance conditions, as mining areas, for example, where soil removal, compaction or degradation has occurred, a site may never return to a state similar to original conditions (Chazdon, 2008) hence people often intervene in various ways to accelerate recovery of unsuccessful species (hereafter, active restoration).

Active restoration requires planting of nursery-grown seedlings, direct seeding, and/or the manipulation of disturbance regimes (for example, thinning and burning) to speed up the recovery process, often at a high cost to establish structural features of the vegetation by establishing the species in question (hereafter, active restoration), reassemble local species composition, and/or catalyze ecological succession (Holl & Aide, 2011, Chazdon & Guariguata, 2016). The most popular active restoration approach in tropical forests is the establishment of plantations in order to establish a canopy covers and catalyze native forest succession (Lamb, 2011; Elliott *et al*, 2013). Thus, understanding the success of ecological restoration (compared to a reference condition) for a particular species requires distinguishing species that can be passively restored from those species requiring active restoration as this can greatly reduce the cost and effort of a restoration project (Meli & Dirzo, 2013).

1.2 Statement of the problem

Recent studies suggest that regeneration following large-scale disturbance in Kibale National Park is slow or possibly arrested (Colin A Chapman, Chapman, & Kaufman, 2009) due to extensive herb layer growth (Löf *et al.*, 2015). There have been attempts to use indigenous tree species but with unsatisfactory results. This is because some species are not suited to the environmental conditions at the planting or soon after (Frissel *et al.*, 2014). Different types of trees or stands require different combinations of these factors depending on their particular adaptations.

Selecting species for active and passive restoration can help to determine the species with low regeneration potential to accelerate species succession (Meli *et al.*, 2013). However, successful selection of species adapted to local conditions requires specific knowledge which is often

lacking, especially in developing countries. A few attemps have been made to select species for active and passive restoration depending on species requirement or species impact (Devictor *et al.*, 2010). This study will help to determine species wih a high regeneration potential so as they can be recommended for passive regeneration and those with a low regeneration potential so as they can be identified for active regeneration to accerelate their regeneration potential.

1.3 Objectives of the study

1.3.1 Overall Objective

The main objective of this study was to select species for both passive and active restoration of Kibale National Park

1.3.2 Specific Objectives

1. To assess the dominance of different species in the damaged ecosystems of Kibale National Park.
2. To determine the regeneration potential of different species in terms of abundance-size correlation.

1.4 Research questions

1. What is the dominance of each selected spicies in the damaged ecosystems of Kibale National Park?
2. What is the regeneration potential of each spicies in the damaged ecosystems of Kibale National Park?

1.5 Significance of the study

This study was conducted in order to provide information to the actors implementing restoration of tropical forests on the preliminary list of useful species for restoration and provide recommendations for possible restoration strategies for particular species. This study would help to develop a comprehensive species list based on their abundance and size as to distinguish species for active and passive restoration as to reduce the cost and efforts of a restoration project.

CHAPTER TWO: LITERATURE REVIEW

2.1 Tropical forests

Tropical rainforest biomes are found in locations throughout the world in a band around the equator known as the "tropics". The tropics wrap around the world in a band approximately 3,000 miles (4,800 kilometers) wide between the Tropic of Cancer and the Tropic of Capricorn.

Over half of the world's tropical rainforest are found in Latin America with about one third of the world's tropical rainforest, including the Amazon Rainforest, located in Brazil.The largest tropical rainforest on earth is the Amazon Rainforest. The tropical rainforest in Africa are found along the west coast and mostly in middle Africa in the Congo Basin; which is the basin of the Congo River. The size of the Congo Basin rainforest is second only to the Amazon Rainforest. The Congo Basin rainforest is mainly found in the Democratic Republic of Congo but also extends into Central African Republic, Gabon, Republic of Congo, Equatorial Guinea, and Cameroon. El Yunque rainforest in Puerto Rico that attracts millions of tourists found on the north east part of the island along the slopes of the Sierra de Luquillo Mountains.

2.2 Tropical forest degradation

2.2.1 Extent of forest loss

The degradation of forests continues to be a global concern because of the threat to both the functioning of ecosystems and the well-being of human communities (Nichol *et al.,* 2017). Rainforests are dissappearing at about 80 acres per minute, day and night (Chazdon & Guariguata, 2016). The tropical rainforest once covered more than 16% of the earth's total land surface, but now covers less than 6% of the earth's total land surface. According to the World Resources Institute, the world has lost about half of its forest cover. Despite a number of initiatives to stop forest decline, the world continues to lose some 15 million hectares of forests every year. Deforestation over the period 1980-1990 reached 8.2% of total forest area in Asia, 6.1% in Latin America and 4.8% in Africa. Most modern deforestation takes place in developing countries, particularly in tropical areas (Ghazoul *et al.,* 2015).

2.2.2 Drivers of forest degradation

The changes in forest cover are the result of various drivers. These drivers can be divided into two types namely proximate and underlying. Proximate causes are those which are visible drivers whereas underlying causes are hidden behind the proximate drivers of deforestation and forest degradation (FAO, 2016). Direct drivers can be grouped into different categories such as agriculture expansion, expansion of infrastructure and wood extraction (Bikram *et al.,* 2017).

2.2.3 Direct causes

2.2.3.1 Expansion of farming land

Agricultural expansion has been determined as the key driver of deforestation in the tropics (Gibson *et al.,* 2011). About 60 per cent of the clearing of tropical moist forests is for agricultural settlement (Specht *et al.,* 2015). Shifting agriculture also called slash and burn agriculture is the clearing of forested land forraising or growing the crops until the soil is exhausted of nutrients and/or the site is over taken by weeds and then moving on to clear more forest. It has been often reported as the main agent of deforestation. It is feared that agricultural expansion which is the main cause of deforestation in the tropics might replace forestry in the remaining natural forests (Bikram *et al.,* 2017).

2.2.3.2 Logging and fuel wood

Logging does not necessarily cause deforestation. However, logging can seriously degrade forests (Putz *et al.,* 2011). Logging in the tropics is more intensive and can be quite destructive. Fuelwood gathering is often concentrated in tropical dry forests and degraded forest areas (Chakravarty *et al.,* 2011). However, logging provides access roads to follow-on settlers and log scales can help finance the cost of clearing remaining trees and preparing land for planting of crops or pasture. Logging thus catalyzes deforestation (Specht *et al.,* 2015).

2.2.3.3 Overgrazing

Overgrazing is more common in drier areas of the tropics where pastures degraded by overgrazing are subject to soil erosion. Stripping trees to provide fodder for grazing animals can also be a problem in some dry areas of the tropics but is probably not a major cause of deforestation. Clear cutting and overgrazing have turned large areas of Qinghai province in China into a desert. Overgrazing is exacerbated by socio-ecological phenomena called "the tragedy of the common." People share land but raise animals for themselves and try to enrich them by rising as many as they can. Animals remove the vegetation and winds finished the job by blowing away the top soil, transforming grasslands into desert (Liebsch *et al.,* 2008).

2.2.3.4 Fires

Fires are a major tool used in clearing the forest for shifting and permanent agriculture and for developing pastures. Fire is a good servant but has a poor master. Fire used responsibly can be a valuable tool in agricultural and forest management but if abused it can be a significant cause of deforestation (Bikram *et al.,* 2017). Based on the data available from 118 countries representing 65 per cent of the global forest area, an average of19.8 million hectares or one per cent of all fores**ts** were reported to be significantly affectedeach year by forest fires (Romero-sanchez, 2017).

2.2.3.5 Mining

Mining is very intensive and very destructive (Specht *et al.,* 2015) forexample, according to an assessment by the World Wildlife Fund-Guianas, the deforestation rate due to mining activities in Guyana from 2000 to 2008 increased by 2.77 times (Purnamasari, 2010). Similarly, in the Philippines, mining, along with logging, hasbeen among the forces behind the country’s loss of forest cover: from 17 million hectares in1934 to just three million in 2003 or an 82 per cent decline (Carandang *et al.,* 2013). Nearly 2,000 hectares of tropical forest in the Municipality of Coahuayana in the State of Michoacán (south-western Mexico) will completely be destroyed by mining iron minerals planned bythe Italo-Argentine mining company TERNIUM (Chakravarty *et al.,* 2011). Similarly, Nyamagari hills in Orissa India currently threatened by Vedanta Aluminum Corporation's plan to startbauxite mining will destroy 750 hectares of reserved forest. Massive

and unchecked mining of coal, iron ore and bauxite in Jharkhand, India has caused large scale deforestation and created a huge water scarcity (Chakravarty *et al.*, 2011).

2.2.3.6 Tourism

National parks and sanctuaries beyond doubt protect the forests, but uncautioned and improper opening of these areas to the public for tourism is damaging. Unfortunately, the national governments of tropical and sub-tropical countries adopt tourism for easy way ofmaking money sacrificing the stringent management strategies. Further, many companies and resorts who advertise themselves as eco-tourist establishments are in fact exploiting the forests for profit. In Cape Tribulation, Australia, for example, the rain forest is being threatened by excessive tourism (Specht *et al.*, 2015, Prideaux *et al.*, 2012). For instance, the Chilapatta Reserve Forest is opened for eco-tourism for its ancient ruins deep in the forest and a tree species *Myristica longifolia* that exudes a blood like sap when injured. The site has become a popular eco-tourist destination because of the ruins and for this blood exuding tree. In the whole forest only eight individuals were found but two of the trees in the near vicinity of the ruins completely dried away due to repeated injuries caused to the plants by the curious tourists (Shukla, 2010). In fact, in the name of eco-tourism, infra-structure development is taking place mostly be the private players in these wilderness areas which are further detrimental in terms of attracting peoples other than tourists also, causing deforestation especially deep in the forest (Crotti & Misrahi, 2015).

2.2.4 Indirect causes

2.2.4.1 Overpopulation and poverty

The role of population in deforestation is a contentious issue (Sands, 2015). Poverty and overpopulation are believed to be the main causes of forest loss according to the international agencies such as FAO and intergovernmental bodies. It is generally believed by these organizations that they can solve the problem by encouraging development and trying to reduce population growth. There is good evidence that rapid population growth is a major indirect and over-arching cause of deforestation. More people require more food and space which requires more land for agriculture and habitation. The growing population in rich industrialized nations is

responsible for much of the exploitation of the earth and there is a clear link between the overconsumption in rich countries and deforestation in the tropics (Chakravarty *et al.*, 2011).

Poverty is well considered to be an important underlying cause of forest conversion by small-scale farmers and naturally forest-dense areas are frequently associated with high levels of poverty (Chakravarty *et al.*, 2011). The population also often lacks the finance necessary for investments to maintain the quality of soil or increase yields on the existing cleared land (Purnamasari, 2010).

2.2.4.2 Land rights and land tenure

Cultivators at the forest frontier often do not hold titles to land (absence of property rights) and are displaced by others who gain tenure over the land they occupy (Sands, 2015). This means they have to clear more forest to survive. Poorly defined tenure is generally bad for people and forests (Bikram *et al.*, 2017). In many countries government have nominal control of forests but are too weak to effectively regulate their use. This can lead to a tragedy of the commons where forest resources are degraded. In frontier areas deforestation is a common practice and legalized way of declaring claim to land and securing tenure (Kauffman *et al.*, 2012).

2.2.4.3 Undervaluing the forest

Forests gain value only when they are cleared for obtaining legal title through 'improvement' (Sands, 2015). The extraction of non-wood forest products has been suggested as a way to add value to the forest but it is not economical when compared to clearing options. If some means could be devised where those who benefit from theenvironmental values could pay the forest owners or agents of deforestation for them, then the option to not clear would become more competitive. Alternatively, if the national governments value the environmental benefits, it could apply a tax or disincentives to clear. However, even though maintenance of the environmental services is essential for sustained economic development, deforesting nations usually have more immediate goals and are unprepared to take this step (Purnamasari, 2010).

2.3 Effects of forest degradation

2.3.1 Impact of forest degradation on biodiversity

Forests especially those in the tropics serve as storehouses of biodiversity and consequently deforestation, fragmentation and degradation destroys the biodiversity as a whole and habitat for migratory species including the endangered ones, some of which have still to be catalogued. Tropical forests support about two thirds of all known species and contain 65 per cent of the world's 10, 000 endangered species (Aerts & Honnay, 2011). Retaining the biodiversity of the forested areas is like retaining a form of capital, until more research can establish the relative importance of various plants and animal species (Devictor *et al.,* 2010). Another negative effect of deforestation is increasing incidents of human-animal conflicts hitting hard the success of conservation in a way alienating the people's participation in conservation. Elephant habitat located at northern West Bengal in India is part of the Eastern Himalaya Biodiversity Hotspot which is characterized by a high degree of fragmentation. The heavy fragmentation of this habitat has resulted into an intense human-elephant conflict causing not only in loss of agricultural crops but also human and elephant lives. Mortality of about 50 persons and 20 elephants was reported due to these severe human-elephant conflicts from this hotspot area annually (Pardini *et al.,* 2010).

2.3.2 Impact of forest degradation on livelihood

2.3.2.1 Economic losses

The tropical forests destroyed each year amounts to a loss in forest capital valued at US $ 45 billion (Bau, 2016). By destroying the forests, all potential future revenues and employment that could be derived from their sustainable management for timber and nontimber products disappear.

2.3.2.2 Social consequences

Deforestation, in other words, is an expression of social injustice (Meli *et al.,* 2017). The social consequences of deforestation are many, often with devastating long-term impacts. For indigenous communities, the arrival of civilization usually means the destruction/change of their traditional life-style and the breakdown of their social institutions mostly with their displacement from their ancestral area. The intrusion of outsiders destroys traditional life styles, customs and religious beliefs which intensifies with infra-structure development like construction of roads which results into frontier expansion often with social and land conflicts (Schmink & Wood, 2009).

The most immediate social impact of deforestation occurs at the local level with the loss of ecological services provided by the forests. Forests afford humans valuable services such as erosion prevention, flood control, water treatment, fisheries protection and pollinationfunctions that are particularly important to the world's poorest people who rely on natural resources for their everyday survival. By destroying the forests we risk our own quality of life, gamble with the stability of climate and local weather, threaten the existence of other species and undermine the valuable services provided by biological diversity (Specht *et al.,* 2015).

2.4 Forest restoration

Restoration is defined as any intentional activity that initiates or accelerates the recovery of an ecosystem from a degraded state (Chapman *et al.,* 2009). Restoration is about improving landscapes throughout the world that are deforested, degraded, or underutilized (Winterbottom, 2014). In places where forests have been lost or degraded, restoration or reforestation projects may be undertaken in order to guarantee or accelerate the recovery of forests (Meli *et al.,* 2014). Aerts & Honnay (2011) who focuse on the restoration of vegetation, believe that ecological restorations can reconstruct a self-maintaining natural community and maintain its continuity, while *(Lennox et al.,* 2011) indicates that ecological restoration is the *p*rocess of reconstructing historical regional plant and animal communities and maintaining the sustainability of the ecological system as well as its traditional cultural functions. Ecological restoration assists the recovery and management of ecological integrity," including a "critical range of variability in

biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices" (Society of Ecological Restoration).

2.4.1 Restoration approaches

Ecological restoration strategies can be divided into two broad categories that is, passive and active restoration. Passive restoration is where environmental stressors such as cattle grazing and agriculture are removed and colonization by shrubs and trees and secondary succession takes place naturally. Active restoration is where the land is managed by planting vegetation, weeding, burning, and/or thinning to achieve a desired structure (Meli *et al.,* 2014). Natural forest regeneration enables the spontaneous recovery of native tree species to colonize and establish in abandoned fields or natural disturbances. This process can also be assisted through human interventions such as fencing to control livestock grazing, weed control, and fire protection (Meli *et al.,* 2013, Zahawai *et al.,* 2014).

In contrast, active restoration requires planting of nursery-grown seedlings, direct seeding, and/or the manipulation of disturbance regimes (for example, thinning and burning) to speed up the recovery process, often at a high cost to establish structural features of the vegetation (hereafter termed vegetation structure), reassemble local species composition, and/or catalyze ecological succession (Auestad *et al.,* 2015). In areas with extensive deforestation, a combination of limited seed dispersal, aggressive exotic vegetation, microclimatic extremes, and/or soil degradation can result in slow or no recovery (Larson *et al.,* 2011). Thus people often intervene in various ways to accelerate recovery, such as planting trees, amending soil, and recontouring topography (hereafter, active restoration).

2.4.2 Species selection for forest restoration

The selection of native species for ecological restoration is much more complex and challenging than the selection of plant materials for monoculture plantations (Meli *et al.,* 2014). Consideration of functional groups enables a more rapid categorization of species, especially in highly diverse forests (Chazdon *et al.,* 2014). Plant functional groups are assemblages of species showing similar responses to the environment and similar effects on ecosystem processes, and can be a useful tool in selecting optimal species for restoration (Ostertag *et al.,* 2015). In

particular, species successional status can be a means for assigning relevant functional groups to restoration activities (Roma *et al.,* 2012).

Several methods of forest restoration using native species exist at present. Knowles and Parrotta document the maximum diversity method, which uses a large number of later successional species to initiate the succession (Meli *et al.,* 2014). This approach is technically challenging and expensive. By contrast, the interplanting of selected pioneer species and later successional species can be used as a simpler and cheaper method of reestablishing the appropriate successional trajectory at degraded sites (Aerts & Honnay, 2011). The pioneer species can act as nurse species (Chase & Myers, 2011) which quickly establish a protective tree canopy, shade out competing weeds, and facilitate the establishment of species from later successional stages. One such approach is the framework species method, which involves the planting of early and later successional tree species at the same time, and which has been shown to be an efficient method of forest restoration in Australia and Thailand (Elliott *et al.,* 2013). However, this method has so far only been shown to be effective for small scale planting, because it requires consideration of a series of ecological properties of tree species (such as weed suppression, attraction of seed dispersal agents, and tolerance of fire), and knowledge about such traits is often limited (Elliott *et al.,* 2013). In the future, providing simple, evidence-based methods for native tree species selection will support and facilitate local restoration initiatives.

3.5 Ecological groups

Ecological species groups consist of co-occurring plant species sharing similar environmental comunities (Grabherr *et al.,* 2013). Species groups are based on the theory that evolutionary and community processes such as competition confine species to environmental complexes where they are best adapted (Kashian *et al.,* 2013). Species group research identifies environmental gradients correlated with species distributions, classifies species assemblages occupying similar environmental complexes, and relates species distributions to management-oriented variables such as tree growth (Host & Pregitzer, 2010). While often all species of a group occur together on a site, presence of one species of a group has been interpreted to suggest that the site meets requirements of all species of that group (Kashian *et al.,* 2013). Including several species in a group for indicating environmental conditions may compensate for absences of individual

species resulting from reasons unrelated to environmental site factors (Barnes *et al.,* 2014). This has been perceived as an advantage of using species groups, rather than individual species, for indicating environmental conditions (Host & Pregitzer, 2010).

Species groups have typically been constructed using combinations of field observations, inspection of tabular species site matrices, and multivariate analyses such as cluster analysis (Kashian *et al.,* 2013). The size of groups varies within and among studies, ranging from fewer than three to greater than eight species (Host & Pregitzer, 2010). As in many multivariate studies in plant ecology, species groups are hypotheses about species distributions and their relationships to environmental factors. These hypotheses have practical value for estimating site conditions, and are tractable for refinement through experimental research developing causal relationships about species distributions (Pardini *et al.,* 2010). Analyzing environmental relationships of plant communities as wholes, indicator and single-species distributions, and ecological species groups are complementary approaches to studying vegetation– environment relationships (Naghi *et al.,* 2014).

3.6 Species dominance

The development of forests over time leads to changing competitive environments for each tree, with some trees increasing dominance over others (Chapman *et al.,* 2011). Larger, dominant trees typically capture more light (and perhaps other resources such as soil water and nutrients) than smaller trees, which provides a positive feedback that further accentuates differences in tree sizes. This expectation would be true only if the efficiency of converting captured light into biomass for large trees was similar (or greater) than the light use efficiency of smaller trees.

Dominance is an important indicator of species composition in a forest (Lohbeck *et al.,* 2014). The dominance of a species refers to its relative importance in its habitat (Chase, and Leibold, 2003) which determines the degree of influence of the species on the ecosystem (Burak *et al.,* 2011). In a forest, species dominance has been studied from the forest attributes of vertical and horizontal characteristics (Magurran, 2013). The forest's vertical structure shows the differentiation of trees into height categories, which results into various canopies. Bohlman & Pacala (2012) noted that in a natural uneven aged forest, canopy differentiation often results in

emergents, the top canopy, the middle canopy and the lower canopy, which in most cases, are species specific. (Bohlman & Pacala, 2012) defined the analysis of a forest's horizontal structure as assigning individual trees into specific sizes. (Magurran, 2013) used the species Importance Values Index (IVI) to rank the dominance of tree species in a forest based on the horizontal forest structure. He evaluated species dominance in terms of basal area's contribution to the specific forest, the number of individuals of the species counted and the frequency of occurrence in the sample sites.

Therefore, (Magurran, 2013) defined a dominant species as one that has big sized trees, many individuals and is spread out over the study area. In a normal forest, there may be no single dominant species (Lin & Tian, 2010). For example, explained that species dominance is shared over more than 270 tree species in the Amazon with either of these species ranking slightly above the rest depending on the local ecological conditions (Schmink & Wood, 2009). In conditions of no disturbance, (Schroeder *et al.,* 2010) have shown that it is actually possible to predict the occurrence of a species in a habitat based on its ecological niche. (Larson *et al.,* 2011) defined "rough distribution areas" for species of trees, shrubs and lianas in Kenya although absence or over dominance of species in such areas can be associated with stress factors like fires and over extraction which characterise sections of the Mau forest complex (Kinyanjui *et al.,* 2014).

CHAPTER THREE: STUDY AREA AND METHODS

3.1 Study area

Kibale National Park (795 km^2) lies in the Albertine Rift in western Uganda (00°13′–00°41′ N, 30°19′–30°32′ E). The park receives an average 1,750 mm of precipitation annually, with average daily temperatures ranging from 15.1–23.1°C. The vegetation varies from ever-green and semi-deciduous forest in the north to grasslands and woodlands in the southwest, due to a decline in elevation from 1,590 m in the north to 900 m in the southwest (Jacob *et al.*, 2017). The vertebrate community mainly consists of mammals and birds (Plumptre *et al.*, 2017). This study was conducted south of the park (Fig.1), where old-growth forests are moist semi-deciduous with *Cynometra alexandri* C. H. Wright as a climax species on poor soils, and *Celtis* spp. and *Chrysophyllum* spp. on rich soils (Ssekuubwa *et al.*, 2017). In 1971, agricultural encroachers cleared about 10,000 ha of forests in the south (Hamilton, Karamura, & Kakudidi, 2016). The forests that survived were mostly fragments along waterways. In 1992, encroachers were resettled outside the park. The formerly encroached areas became dominated by elephant grass (*Cenchrus purpureus* (Schumach.) Morrone) because regular fires spreading from nearby gardens prevented natural regeneration (Oliveira *et al.*, 2017).

In 1994, the Government of Uganda, through Uganda Wildlife Authority (UWA), and FACE Foundation started restoring forests as carbon offsets on formerly encroached areas (Fisher, Cavanagh, Sikor, & Mwayafu, 2018). Active and passive restoration methods were implemented. Active restoration involved planting of native tree species every year from 1995 to 2010, except in 2001. The restoration plantings were nursery-grown seedlings and wildings collected from intact forests. The main tree species planted were *Albizia* spp., *Bridelia micrantha* (Hochst.) Baill, *Croton* spp., *Shirakiopsis elliptica* (Hochst.) Esser, *Celtis gomphophylla* Baker, and *Warburgia ugandensis* Sprague.

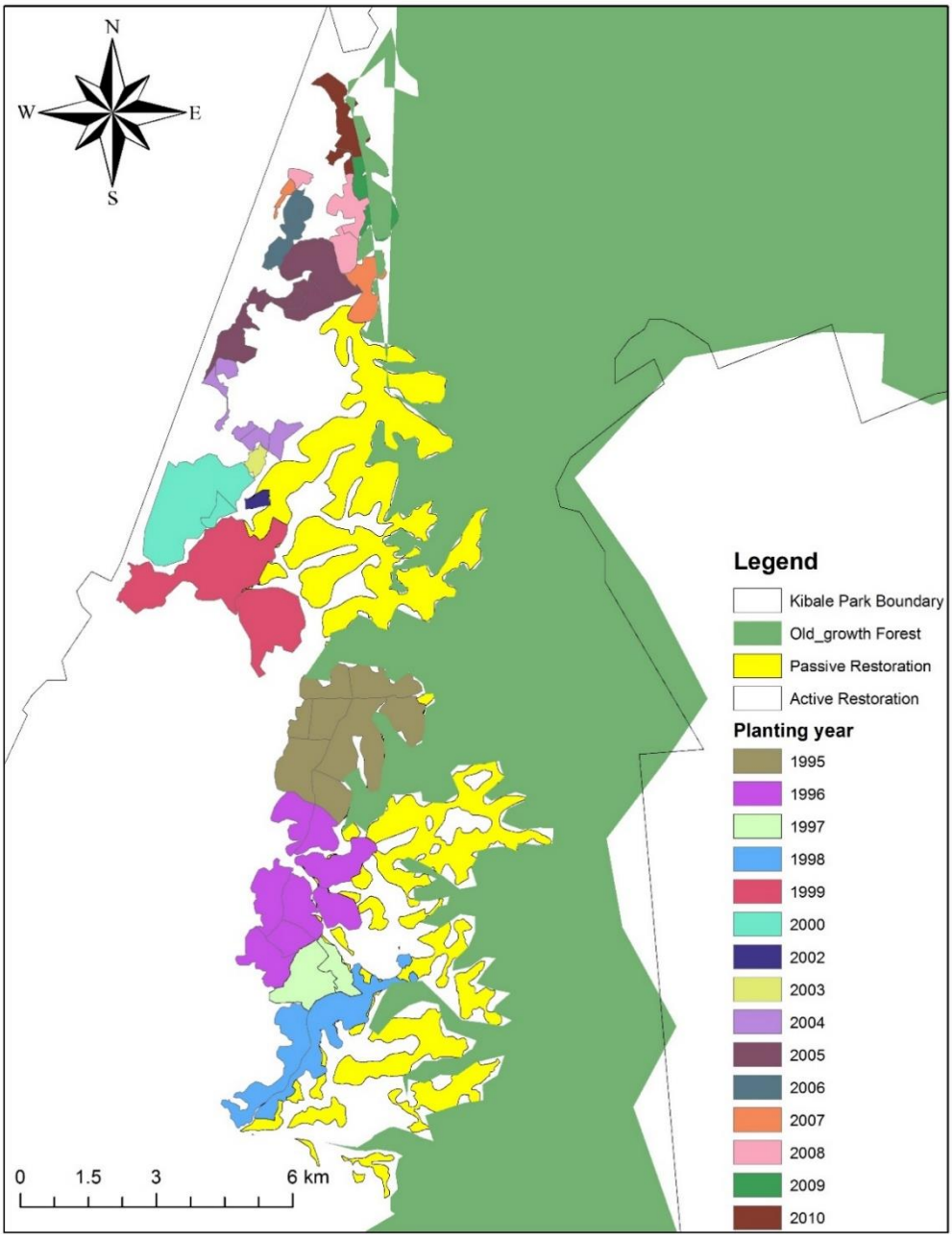

Figure 1. Location of actively and passively restored forests and old-growth forest in Kibale National Park, western Uganda. The actively restored forests are of different planting years (i.e., 1995 – 2010). Passively restored forests are of a single restoration year (i.e., 1995). The open space below the 1999 restoration plantings is a protected grassland within the park being colonized by trees and shrubs. The open spaces among the restored forests are remnant forests.

Vegetative propagation by cuttings accounted for 5% of trees planted (UWA-FACE, 2011). Planting sites were prepared by clearing elephant grass along 2 m wide trails spaced in a 5 x 5 m grid, and planting pits were dug every 5 m along trails at densities of 400 per ha. The planted area was divided into compartments of different sizes and weeding was carried out 2–3 times/year. In addition, fire breaks were cut between compartments, and fires were fought by UWA and local communities (Ssekuubwa *et al.,* 2017). Passive restoration involved protection against fires and livestock grazing in 1995, to facilitate natural regeneration of native woody species. The size of actively and passively restored forests since restoration started was 3,996 and 2,593 ha respectively (Crouzeilles *et al.,* 2017). The actively restored forests of different ages are geographically close to each other, with a 24 km maximum distance between the two most distant areas (Fig. 1). The passively restored forests are also close to each other with a distance of 19 km between the most distant areas. The forests restored actively and passively in 1995 are bordered by the old-growth forest. Small streams and highly diverse native forest fragments are scattered throughout the actively and passively restored forests. Current management of the restoration trajectory involves control of invasive woody species and grasses, livestock grazing and fires, and regulated use of restored forests by communities for extraction of non-timber forest products, such as fuelwood and medicinal plants (Corbin & Holl, 2012).

3.2 Experimental Design

The data used in this study was collected by Face Foundation to monitor carbon accumulation in actively and passively restored forests. A regular sampling grid consisting of clusters of four permanent sampling plots with a spacing of 500 x 500 m (Fig. 2) was applied to compartments of actively restored forests and two old-growth forest compartments using Field Map technology (IFER , 1994) . The same grid consisting of clusters of three permanent sampling plots was applied to compartments of passively restored forests. Each sampling plot (2,000 m^2) consisted of four 500 m^2 circles, i.e., one key circle at the bottom left of each plot, and three other circles. The key circle contained a small eccentric subplot (12.6 m^2) located 8 m north from the key circle centre and a concentric internal circle (201.1 m^2). Woody seedlings of diameter < 5 cm and height > 10 cm were measured in eccentric subplots. Trees of diameter at breast height (dbh, at 1.3 m above the ground) ≥ 5 cm were measured in the concentric internal subplots, while trees of dbh ≥ 30 cm were measured the remaining three circles of the plot. Individuals were identified to

species level and diameter, height and origin (i.e., whether planted or from natural regeneration) were recorded. Species identification followed guides by Katende *et al.* (1995) and Eggeling (1940). We acquired the data from Face Foundation through a data sharing agreement (Appendix 1). Table 1 shows the plots surveyed in study sites. Distance from old-growth forests or remnant forests to sampling plots in restored forests was estimated using local area maps and was confirmed by UWA staff.

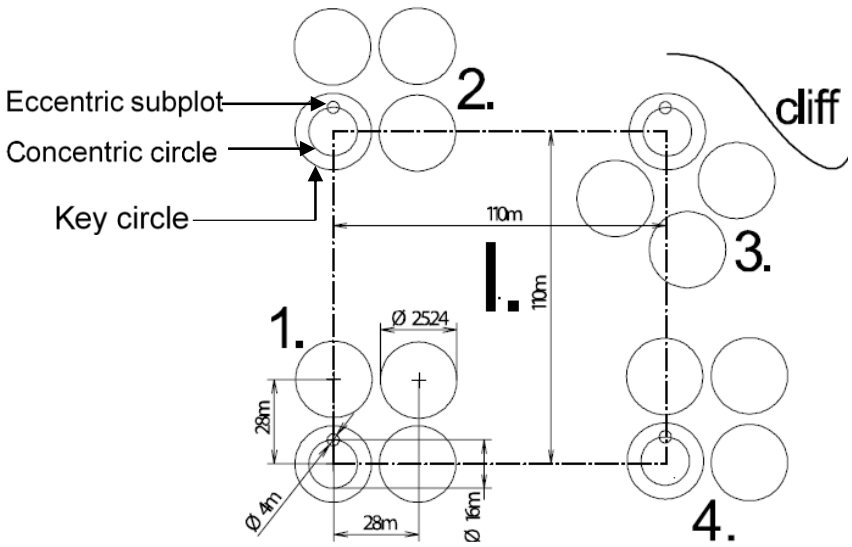

Figure 2. The monitoring cluster (I) consisted of four permanent sampling plots (2,000 m^2) in compartments of actively restored forests or three permanent sampling plots compartments of passively restored forests in Kibale National Park, western Uganda. Each plot was composed of four 500 m^2 circles. Plot 3 was rotated because of an obstacle (e.g., a road, cliff or river). Woody seedlings of diameter < 5 cm and height > 10 cm were measured in eccentric subplots. Trees of diameter at breast height (dbh, at 1.3 m above the ground) ≥ 5 cm were measured in the concentric internal subplots, while trees of dbh ≥ 30 cm were measured the remaining three circles of the plot.

Table 1. Study sites, restoration years, number of compartments, clusters and sample plots at each site in Kibale National Park, western Uganda. Old-growth forest are of unknown size (N/A).

Study sites	**Restoration year (area in ha)**	**Compartments**	**Clusters**	**Plots**
Actively restored forests	1995 (688.8)	8	15	60
	1996 (591.8)	7	11	44
	1997 (168.6)	1	3	12
	1998 (359.7)	2	4	16
	1999 (565.2)	2	12	48
	2000 (339.2)	2	6	24
	2002 (15.6)	1	1	4
	2003 (30.0)	1	1	4
	2004 (98.3)	2	3	12
	2005 (282.1)	5	6	24
	2006 (79.8)	2	3	12
	2007 (89.9)	1	3	12
	2008 (115.5)	2	3	12
Passively restored forests	1995 (2593)	4	21	63
Old-growth forest forests	N/A	2	3	5
Total		42	95	352

3.3 Data analysis

To analyze the acquired data gathered from plot sampling, tree diameter was considered. The dbh data were converted to basal area values using $\pi\times$ (dbh×0.5). For each transect and species, an importance value index (IVIi) was calculated as the sum of the species' relative density, relative frequency, and relative basal area divided by 3 (Meli *et al.*, 2013). These parameters were obtained as follow:

RF = (frequency of species i / sum frequencies of all species) × 100,

RDe = (number of individuals of species i / total number of individuals) × 100.

RDo = (total basal area for species i / total basal area of all species) × 100 (Pereki *et al.*, 2013). This IVIi was used to determine the species dominance which increased with IVIi. The higher the IVIi, the more dominant the species was and the lower the IVIi, the less dominant the species. For each plot, each species' abundance (*N*i, number of stems of species i per transect) in each of 14dbh classes (from 5 to >100 cm, with 10-cm intervals) was calculated from which each species abundance was derived. For each plot and species, the correlation (Pearson rank correlation, *r*s) between abundance [log (*N*i +1)] and the midpoint of the dbh classes (hereafter called abundance–size correlation) was calculated (Meli *et al.*, 2013). A high regeneration potential was represented by a diminishing number of individuals as diameter size increased. This trend resulted in a high negative correlation (high availability of small-sized trees), and therefore an acceptable potential for passive establishment of the species. A positive (lack of small-sized trees) meant that the species does not establish naturally and therefore need for active restoration.

CHAPTER FOUR: RESULTS

4.1 Regeneration potential, dominance and ecological groups of the tree species in Kibale National Park.

A total of 65 species were found in the study area of which Sapium ellipticum had the maximum IVIi (12%) and only fifteen species had an IVIi >2% (Table 2). All the species were characterized for the restoration assessment in order to give a clear view to the restoration actors on the best restoration approach for each species in Kibale National Park in order to reduce on the restoration costs and efforts. Thirteen species showed a higher negative abundance–size correlation (rs <−0.6) in this area, suggesting that passive restoration could be sufficient for their successful establishment (Table 3). At the other extreme, 52 species were absent in most plots and their abundance–size correlation was not significant, suggesting that these species could be introduced by active restoration. The five most dominant species were Sapium ellipticum, Bridelia micrantha; Albizia zygia, Mimusops kummel and Bridelia scleroneura in their respective ascending order and the least dominant were nineteen with an IVIi of 0.3531. Five distinct ecological groups were found in Kibale National Park which included pioneers, non pioneer light demanders, shade tolerants savanna species and swamp species. The pioneers consisted of twenty seven samples of which six samples were for passive restoration and twenty one species were for for active restoration. The non pioneer light demanders consisted of twenty two samples of which three samples were for passive restoration and nineteen for active restoration. Shade torelants consisted of twelve samples of which two were for passive restoration and ten for active restoration. Savanna group consisted of three samples of which two were for passive restoration and one for active restoration and the swamp group consisted of one sample for active restoration.

Table 2. Species importance value index (IVIi) of native tree species found in Kibale National Park, western Uganda.

Specics name	Number of trees	Frequency	Relative frequency	Density	Relative density	Bassal area	Relative basal area	Importance value index
Sapium ellipticum	168	0.48	11.33	5.79	3.10	263.89	22.42	12.28
Bridelia micrantha	60	0.38	8.98	2.61	1.40	94.25	8.01	6.13
Albizia zygia	71	0.20	4.69	5.92	3.17	111.53	9.47	5.78
Mimusops kummel	31	0.03	0.78	15.50	8.29	48.69	4.14	4.40
Erythrina abyssinica	30	0.28	6.64	1.71	0.91	45.55	3.87	3.81
Bridelia scleroneura	29	0.18	4.30	2.64	1.41	45.55	3.87	3.19
Celtis Africana	26	0.18	4.30	2.36	1.26	40.84	3.47	3.01
Acacia hockii	27	0.21	5.08	2.08	1.11	27.41	2.33	2.84
Funtumia elastic	22	0.05	1.17	7.33	3.92	34.56	2.94	2.68
Crassocephalum mannii	15	0.03	0.78	7.50	4.01	23.56	2.00	2.27
Uvariopsis congensis	18	0.05	1.17	6.00	3.21	28.27	2.40	2.26
Funtumia Africana	18	0.07	1.56	4.50	2.41	28.27	2.40	2.12
Alstonia boonei	17	0.11	2.74	2.43	1.30	26.70	2.27	2.10
Flueggea virosa	18	0.08	1.95	3.60	1.93	28.27	2.40	2.09
Chrysophyllum albidum	12	0.03	0.78	6.00	3.21	18.85	1.60	1.86
Ficus sur	12	0.07	1.56	3.00	1.61	18.85	1.60	1.59
Albizia coriaria	10	0.08	1.95	2.00	1.07	15.71	1.33	1.45
Entada abyssinica	9	0.10	2.34	1.50	0.80	14.14	1.20	1.45
Spathodea campanulata	9	0.08	1.95	1.80	0.96	14.14	1.20	1.37
Mimusops bagshawei	5	0.02	0.39	5.00	2.68	7.85	0.67	1.24
Grewia mollis	5	0.02	0.39	5.00	2.68	7.85	0.67	1.24
Acacia sieberiana	7	0.08	1.95	1.40	0.75	11.00	0.93	1.21
Diospyros mespiliformis	6	0.08	1.95	1.20	0.64	9.42	0.80	1.13
Combretum molle	7	0.05	1.17	2.33	1.25	11.00	0.93	1.12
Kigelia Africana	6	0.07	1.56	1.50	0.80	9.42	0.80	1.06

Table 2 Continued. Species importance value index (IVIi) of native tree species found in Kibale National Park, western Uganda.

Specics name	Number of trees	Frequency	Relative frequency	Density	Relative density	Bassal area	Relative basal area	Importance value index
Celtis durandii	4	0.02	0.39	4.00	2.14	6.28	0.53	1.02
Mangifera indica	4	0.02	0.39	4.00	2.14	6.28	0.53	1.02
Tabernaemontana holstii	4	0.02	0.39	4.00	2.14	6.28	0.53	1.02
Morus alba	5	0.03	0.78	2.50	1.34	9.42	0.80	0.97
Dissotis perkinsiae	5	0.03	0.78	2.50	1.34	7.85	0.67	0.93
Strombosia scheffleri	5	0.03	0.78	2.50	1.34	7.85	0.67	0.93
Caloncoba schwernsferthii	4	0.05	1.17	1.33	0.71	6.28	0.53	0.81
Markhamia platycalyx	4	0.05	1.17	1.33	0.71	6.28	0.53	0.81
Cassia spectabilis	3	0.02	0.39	3.00	1.61	4.71	0.40	0.80
Grewia bicolor	3	0.02	0.39	3.00	1.61	4.71	0.40	0.80
Monodora angolensis	3	0.02	0.39	3.00	1.61	4.71	0.40	0.80
Psidium guajava	3	0.02	0.39	3.00	1.61	4.71	0.40	0.80
Pterygota mildbraedii	4	0.03	0.78	2.00	1.07	6.28	0.53	0.80
Trilepisium madagascariensis	4	0.03	0.78	2.00	1.07	6.28	0.53	0.80
Allophylus africanus	3	0.05	1.17	1.00	0.54	4.71	0.40	0.70
Warburgia ugandensis	3	0.05	1.17	1.00	0.54	4.71	0.40	0.70
Dichrostachys cinerea	3	0.03	0.78	1.50	0.80	4.71	0.40	0.66
Euadenia eminens	2	0.02	0.39	2.00	1.07	3.14	0.27	0.58
Aningeria altissima	2	0.02	0.39	2.00	1.07	3.14	0.27	0.58
Crassocephalum africanus	2	0.02	0.39	2.00	1.07	3.14	0.27	0.58
Ficus mucuso	2	0.02	0.39	2.00	1.07	3.14	0.27	0.58
Maytenus undata	2	0.02	0.39	2.00	1.07	3.14	0.27	0.58
Neoboutonia macrocalyx	2	0.02	0.39	2.00	1.07	3.14	0.27	0.58
Treculia Africana	2	0.02	0.39	2.00	1.07	3.14	0.27	0.58
Clausena anisate	2	0.03	0.78	1.00	0.54	3.14	0.27	0.53

Table 2 Continued. Species importance value index (IVIi) of native tree species found in Kibale National Park, western Uganda.

Specics name	Number of trees	Frequency	Relative frequency	Density	Relative density	Bassal area	Relative basal area	Importance value index
Croton megalocarpus	2	0.03	0.78	1.00	0.54	3.14	0.27	0.53
Ficus natalensis	2	0.03	0.78	1.00	0.54	3.14	0.27	0.53
Pseudospondias microcarpa	2	0.03	0.78	1.00	0.54	3.14	0.27	0.53
Crassocephalum afromontanum	3	0.03	0.01	1.50	0.80	4.71	0.40	0.40
Cola bracteata	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Cola gigantean	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Cordia crenata	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Entandrophragma angolense	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Entandrophragma utile	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Erythrina excels	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Ficus exasperate	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Lychnodiscus cerospermus	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Mimulopsis arborescens	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Phoenix reclinata	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Polyscias fulva	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35
Teclea nobilis	1	0.02	0.39	1.00	0.54	1.57	0.13	0.35

Table 3. Species, ecological groups and Pearson rank correlation coefficient (*r*s) for native tree species found in Kibale National Park and recommendations for passive or active restoration.

Species name	Group	Correlation	Passive	Active
Acacia hockii	SAVA	-0.449763688		✓
Acacia sieberiana	SAVA	-0.641227452	✓	
Albizia coriaria	SAVA	-0.69380804	✓	
Albizia zygia	NPLD	-0.738996738	✓	
Allophylus africanus	NPLD	-0.449763688		✓
Alstonia boonei	PION	-0.673643037	✓	
Aningeria altissima	NPLD	0.303746412		✓
Bridelia micrantha	PION	-0.539273439		✓
Bridelia scleroneura	PION	-0.540777475		✓
Caloncoba schwernsferthii	PION	-0.449763688		✓
Cassia spectabilis	PION	-0.364400821		✓
Celtis Africana	PION	-0.704796512	✓	
Celtis durandii	PION	-0.390088157		✓
Chrysophyllum albidum	SHTO	-0.776377414	✓	
Clausena anisate	PION	-0.449763688		✓
Cola bracteata	NPLD	-0.308921041		✓
Cola gigantean	NPLD	-0.449763688		✓
Combretum molle	NPLD	-0.573254353		✓
Cordia crenata	NPLD	-0.102809904		✓
Crassocephalum africanus	SHTO	0.000361287		✓
Crassocephalum afromontanum	SHTO	-0.459550897		✓
Crassocephalum mannii	SHTO	-0.538550191		✓
Croton megalocarpus	NPLD	-0.449763688		✓
Dichrostachys cinerea	NPLD	-0.449763688		✓
Diospyros mespiliformis	PION	-0.449763688		✓

Table 3 Continued. Species, ecological groups and Pearson rank correlation coefficient (*r*s) for native tree species found in Kibale National Park and recommendations for passive or active restoration.

Species name	Group	Correlation	Passive	Active
Dissotis perkinsiae	PION	-0.449763688		✓
Entada abyssinica	PION	-0.569798646		✓
Entandrophragma angolense	NPLD	-0.449763688		✓
Entandrophragma utile	NPLD	-0.449763688		✓
Erythrina abyssinica	PION	-0.746240612	✓	
Erythrina excels	PION	-0.308921041		✓
Euadenia eminens	SHTO	-0.449763688		✓
Ficus exasperate	PION	-0.449763688		✓
Ficus mucuso	PION	-0.449763688		✓
Ficus natalensis	PION	-0.558376842		✓
Ficus sur	PION	-0.449763688		✓
Flueggea virosa	NPLD	-0.449763688		✓
Funtumia Africana	NPLD	-0.553254196		✓
Funtumia elastic	NPLD	-0.616504683	✓	
Grewia bicolor	PION	-0.449763688		✓
Grewia mollis	PION	-0.449763688		✓
Kigelia Africana	NPLD	-0.522880915		✓
Mangifera indica	SHTO	-0.608941512	✓	
Markhamia platycalyx	PION	-0.663450549	✓	
Maytenus undata	NPLD	-0.308921041		✓
Mimulopsis arborescens	NPLD	-0.240217183		✓
Mimusops bagshawei	NPLD	-0.351994198		✓
Mimusops kummel	NPLD	-0.70565634	✓	
Monodora angolensis	SHTO	0.047914964		✓
Morus alba	PION	-0.315855965		✓

Table 3 Continued. Species, ecological groups and Pearson rank correlation coefficient (*r*s) for native tree species found in Kibale National Park and recommendations for passive or active restoration.

Species name	Group	Correlation	Passive	Active
Neoboutonia macrocalyx	PION	-0.377624899		✓
Phoenix reclinata	NPLD	-0.377624899		✓
Polyscias fulva	PION	-0.449763688		✓
Pseudospondias microcarpa	SWAM	-0.45471946		✓
Psidium guajava	SHTO	-0.377624899		✓
Pterygota mildbraedii	PION	-0.195275676		✓
Sapium ellipticum	PION	-0.745228798	✓	
Spathodea campanulata	PION	-0.711953562	✓	
Strombosia scheffleri	SHTO	-0.334348853		✓
Tabernaemontana holstii	SHTO	-0.559154774		✓
Teclea nobilis	SHTO	-0.449763688		✓
Treculia Africana	NPLD	-0.2524611		✓
Trilepisium madagascariensis	PION	-0.077774249		✓
Uvariopsis congensis	SHTO	-0.56003745		✓
Warburgia ugandensis	NPLD	-0.449763688		✓

CHAPTER FIVE: DISCUSSION

5.1 Criteria for species selection

Studies of communities and individual species combined with classifications of ecological species groups provide a stronger understanding of vegetation-environment relationships than studying either one alone. Natural dominance was the first criterion that was used for species selection. I targeted selection of woody species to initiate forest restoration projects. Although tropical riparian ecosystems contain other than woody species, these species can facilitate the establishment of other plants (Román-Dañobeytia *et al.,* 2012) when their architecture (e.g. leaf and canopy area) buffers harsh abiotic conditions (Meli & Dirzo, 2013), attract seed dispersers when having fresh fruits (Galindo-Rodriguez & Roa-Fuentes, 2017) and outcompete (typically) shade-intolerant grasses through reducing their cover (Dent *et al.,* 2012). They also provide organic matter to the riparian soil and promote shore stabilization in the medium term through their dense roots (Ford *et al.,* 2016). All these characteristics may be also considered as species selection criteria in forest restoration projects, but their inclusion will depend mainly on the ecological condition of the degraded ecosystem, and should be complemented with other criteria (Aerts & Honnay, 2011).

Once the restoration project has been established, it is necessary to consider a wider range of species to fill underrepresented niches with other life forms (e.g. herbs, palms and ferns) and with rare, endangered, endemic and/or riparian specialist species, and thus to improve the structure and function of the riparian forest (Meli *et al.,* 2013) and promote higher diversity and functional redundancy (Fetzer *et al.,* 2015). This will ensure the effectiveness of critical ecological processes that sustain ecosystems (Society for Ecological Restoration International Science & Policy Working Group (SER) 2004. Natural regeneration potential was used as the second criterion. The predictive potential of the abundance–size correlations for selecting target species from disturbed sites could be limited by the small sample size, and hence decrease as their age increases and its species composition starts to resemble that of the reference site(Meli *et al.,* 2013).

However, the typically low species abundance in highly diverse humid tropics makes it difficult to perform accurate correlations without higher statistical power. Assessing some preferred ecological characteristics of target species is a different way to estimate the potential for establishment (Maes *et al.,* 2016). For example, longevity, resistance to herbivores or physical damage, and tolerance to flooding in the case of riparian systems, could also be important features for assessing the potential for establishment (Meli *et al.,* 2014). These features focus on the species responses to particular abiotic or biotic factors. Some of these ecological features are indirectly included in our habitat breadth score, since generalist species may have life-history and functional attributes to cope with biotic and abiotic environmental filters better than specialist species (Devictor *et al.,* 2010). Young fallows such as those we surveyed to estimate the NRP are not always present in areas where restoration is being planned, but they are good sites to identify potential species for passive restoration purposes at the initial stages of restoration efforts (Meli *et al.,* 2013). In subsequent stages of the restoration project, other sites such as older regeneration patches and other ecological species characteristics could be used (Nichol *et al.,* 2017).

My target species list is useful to restore typical disturbed riparian forests in the studied region, including those human-disturbed sites that were abandoned recently (with minimal natural regeneration) or long ago (with substantial natural regeneration) (Meli *et al.,* 2014). Unlike (Brudvig & Mabry, 2008), I did not consider the species of the regional pool that were already established at the disturbed sites because they may not be the most suitable species in social or economic terms when degradation is not very severe, as was the case in my study. The ability of such species to establish naturally in degraded areas is high, and therefore it may be more appropriate to use these species for restoration of severely degraded lands, such as mined sites (Sharma *et al.,* 2016) or sites highly susceptible to erosion on steep slopes (Konz *et al.,* 2010). Seed size and dispersal mechanism syndromes have also been used to understand which species might require active re-establishment and which might passively recolonize degraded sites (Galindo-Rodriguez & Roa-Fuentes, 2017). For example, regenerating species in disturbed sites are frequently those with small seeds, which are widely dispersed (Chazdon *et al.,* 2015). I believe that regeneration indices (Latawiec *et al.,* 2016) are more accurate indicators of these two types of species.

Although not all second-growth forests have recolonized degraded sites, and some species may be adapted to several forms of degradation (e.g. degraded soils, fires and weed infestations), the regeneration potential is a good indicator of the potential use of the species for restoration purposes. It's useful where trees are dominant, but its use would be limited in grasslands or other ecosystem types where species regeneration is difficult to estimate (Meli *et al.,* 2013). Further research is needed to select appropriate species to suit the specific ecological requirements in other ecosystem types.

Finally, the most appropriate method to select target species for restoration will strongly depend on the main objectives of any particular project. Other criteria could be considered in the selection of target species in other case studies, including adaptive capacity to different soils (Singh *et al.,* 2016), other social values (Pardini *et al.,* 2010) or attributes such as dispersal syndromes (Crouzeilles *et al.,* 2017). Technical constraints may be the most useful criterion in practical terms because these can increase the costs (time, labour, materials needed) of the restoration projects, but social criteria should be included in all restoration efforts (Corbin & Holl, 2012).

5.2 Species field performance

Survival and growth rates are important factors in selecting tree species for restoration activities (Aronson *et al.,* 2012; Hall *et al.,* 2011). Early survival and growth of native species in a degraded site were examined which were dominated by pioneers and non pioneer light demanders. Other ecological groups included the shade tolerants, swamp and savanna. The pioneer early successional species (including *Alstonia boonei, Celtis Africana, Erythrina abyssinicaMarkhamia platycalyxSapium ellipticum and Spathodea campanulata*) performed well with high survival rates and rapid growth. These species can reduce the costs of restoration efforts and provide forest products and services in a short period of time. Where land degradation is severe, as in degradation caused by mining (Singh *et al.,* 2016), or with specific problems such as high erosion on steep slopes (Latawiec *et al.,* 2016), the use of pioneer species group adapted to grow on disturbed or degraded ecosystems could be recommended for active restoration.

However, diversity is likely to remain low unless these species are mixed with later successional species ecological groups.

I found that most mid-successional and late-successional ecological groups had slower growth and lower survival rates, but with a large degree of variation among species. This study shows that some mid-successional species, such as *Albizia zygia, Funtumia elastic and Mimusops kummel* are suitable for restoration of highly degraded sites. Meanwhile, some late successional species, such as *Acacia sieberiana, Albizia coriaria, Chrysophyllum albidum and Mangifera indica* are also sound choices. Similarly, several previous studies conducted in tropical areas have demonstrated that some later successional species can perform well in open sites *(Raman et al.,* 2015, Román-Dañobeytia *et al.,* 2012). These findings indicate that with careful selection, some later successional species can be incorporated into restoration plantings. However, the other later successional species in our study performed poorly with low survival rates and slow growth, including several species like *Crassocephalum africanus, Monodora angolensis, Psidiumguajava, Teclea nobilis* and many others that are dominant in the some natural forests (Tang *et al.,* 2013, Meng *et al.,* 2013). Further research is needed in order to determine whether the survival and growth rates of these species can be enhanced using methods such as larger seedlings, direct seeding, inoculation with mycorrhizal fungi, increased post-planting care, planting in less degraded sites or planting after the establishment of a pioneer canopy (Cole *et al.,* 2011). As trees are long-lived organisms, the effectiveness and costbenefit of restoration treatments and the implications for later forest management need longterm field research.

5.3 The role of species from different ecological groups

As ecological information on most native sub tropical tree species is still sparse, this study shows that species successional guides can be used as simple guidelines for species selection. Fast-growing native timber species such as Sapium ellipticum can act as nurse trees for other species in highly degraded sites, and also enhance the business case for restoration by providing early revenues through selective thinning and sustainable harvesting (Tang *et al.,* 2010, Swinfield *et al.,* 2016). Exotic species (e.g.Acacia spp.) are often promoted as nurse tree species in forest restoration practices (Ren *et al.,* 2008). However, our results suggest that native pioneer tree species may be a more appropriate choice for restoration efforts, especially those that regenerate

faster like Sapim ellipticum. Our results also indicate that certain mid-successional non pioneer light demanders and late-successional species like shade tolerants, swamp and savanna ecological groups (such as Albizia zygia, Funtumia elastic, Mimusops kummel, Acacia sieberiana, Albizia coriaria, Chrysophyllum albidum and Mangifera indica) have high potential for restoration planting in this region. Mid- and late-successional species are essential for maintaining productivity, species richness and ecological services in forest ecosystems (Corbin & Holl, 2012).

As forest restoration is a long-term process, long-lived, slower-growing tree species are better suited to long-term carbon sequestration than fast-growing, shortlived species (Stanturf *et al*, 2017). In degraded landscapes, forest fragmentation and defaunation limit the dispersal of seeds, particularly larger seeds, and as a result, natural regeneration of later successional species is often limited (Swinfield *et al.,* 2016). Passive secondary succession is likely to be very slow in many degraded sites and the resulting forest is often dominated by pioneer species, which results in a landscape with little ecological and conservation value (Swinfield *et al.,* 2016, Corlett, 2014). Hence, active planting of later successional species is often required to accelerate forest succession and avoid the creation of "pioneer deserts". Furthermore, later successional species often have higher market values than fast-growing pioneer species, and as supply shortages become more common, their market values are increasing (Galindo-Rodriguez & Roa-Fuentes, 2017). Some studies have shown that forest restoration around remaining natural forests (e.g. isolated protected areas) can balance the benefits of biodiversity conservation and rural livelihoods (Keenelyside *et al.,* 2012). Incorporating economically attractive native species into local restoration plantings may incentivize local people to protect and maintain them. At the same time, producing alternative sources of timber may reduce over-harvesting and disturbance in natural forest. Candidate species for restoration Based on our study, we propose thirteen species with excellent and good performance as candidates for restoration plantings in this area.

Firewood harvesting is regarded as a major and chronic driver of forest degradation in densely populated areas (Specht *et al.,* 2015) and some of these species can provide a sustainable source of firewood for locals, as well as reducing soil erosion (Konz *et al.,* 2010). Restoration with these multi-purpose species could provide economic and ecological benefits to local people, while

restoration success could be enhanced by traditional management experience. Ambitious forest restoration targets can best be achieved through addressing local needs and involving local communities in species selection (Chazdon *et al.*, 2015).

These results offer empirical evidence that some previously ignored native subtropical species have the potential to play important roles in restoration planting. Further studies could evaluate more species and biomes, while information on tree species performance could be made available through user-friendly platforms such as the World Agroforestry Centre's online database "Agroforestry Species Switchboard" and the mobile application "vegetation map for Africa" (Kindt *et al.*, 2016). More importantly, participatory approaches should be applied to promote the use of indigenous knowledge and involve the local community in species selection (Ranjitkar *et al.*, 2016; He *et al.*, 2015). This could enhance information exchange between researchers and practitioners, and enable both local farmers and policy makers to identify and use appropriate native tree species in reforestation and restoration projects (Chazdon *et al.*, 2015, Jacobs *et al.*, 2015). Further, research integrating both scientific and indigenous knowledge is also required to investigate the implications of interactions between native species for multiple species plantin (Ranjitkar *et al.*, 2016; He *et al.*, 2015).

Furthermore, the use of species successional status to aid species selection for forest restoration is a flexible and generalizable tool. The adoption of the approach described in this study, combined with the full participation of local communities, could enable those involved in restoration to select species more reliably and to design their restoration plans using a range of suitable species, including previously neglected native species. This could improve the success of restoration activities in terms of cost, local income generation, enhanced biodiversity and provision of ecosystem services in the subtropics and elsewhere (Aerts & Honnay, 2011).

CHAPTER SIX: CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

This method is useful to select species for restoration because of its relatively low cost and simplicity, which makes it accessible to different stakeholders. It could be applied in other tree-dominated ecosystems, but its use would be limited in grasslands or other ecosystems where species regeneration is difficult to estimate (Meli *et al.*, 2013).

As in any restoration project, the method selected depends on the main objectives. In different conditions, other criteria could be considered in species selection, including soil adaptive capacity (Singh *et al.*, 2016), social values (Pardini *et al.*, 2010), and dispersal syndromes (Crouzeilles *et al.*, 2017). Rare species such as shrubs and herbaceous species are also important, but not necessarily at early stages of restoration.

The high number of woody species found in my study area indicates that the regional species pool for riparian restoration is wide. To facilitate practical restoration, a preliminary list of tree species that are most suitable for their reintroduction into degraded riparian zones of the tropics and similar ecological settings are identified (Ralston & Sarr, 2017). A list of target species must be identified and used for the initial stages of restoration of ecosystems dominated by trees.

6.2 Recommendation

At the early stages of restoration of tree-dominated ecosystems, the combination of species dominance indexes (e.g. IVIi) and abundance–size correlations should be used to select a preliminary list of species suitable for passive or active restoration.

Species that establish by natural regeneration should be used in passive restoration actions when ecosystems are not severely degraded but in severely degraded ecosystems, active restoration is appropriate for some species that fail to regenerate naturally.

My results also suggest that native pioneer tree species may be a more appropriate choice for restoration efforts, especially those that regenerate faster like Sapim ellipticum to act as nurse trees for other species in highly degraded sites.

These results also propose that certain mid-successional non pioneer light demanders and late-successional species like shade tolerants, swamp and savanna ecological species groups (such as Albizia zygia, Funtumia elastic, Mimusops kummel, Acacia sieberiana, Albizia coriaria, Chrysophyllum albidum and Mangifera indica) can be used as nurse trees for other species in highly degraded sites since they have a high regeneration potential in this region.

LITERATURE CITED

Aerts, R., & Honnay, O. (2011). Forest restoration, biodiversity and ecosystem functioning. *BMC Ecology*, *11*, 29.

Andrade, B. O., Koch, C., Boldrini, I. I., & Vélez-martin, E. (2015). Natureza & Conservação Essays and Perspectives Grassland degradation and restoration : a conceptual framework of stages and thresholds illustrated by southern Brazilian grasslands. *Natureza & Conservação*, *13*, 95–104.

Auestad, I., Rydgren, K., & Austad, I. (2015). Near-natural methods promote restoration of species-rich grassland vegetation–revisiting a road verge trial after 9 years. *Restoration Ecology*, *24*, 381–389.

Barnes, B. ., Zak, D. ., Denton, S. ., & Spurr, S. . (2014). John Wiley and Sons Inc., New York. *Forest Ecology*.

Bau, D. E. (2016). Deforestation and forest degradation in southern Burkina Faso : Understanding the drivers of change and options for revegetation. *Department of Forest Science*, 3–80.

Bechara, F. C., Dickens, S. J., Farrer, E. C., Larios, L., Spotswood, E. N., Mariotte, P., & Suding, K. N. (2016). Neotropical rainforest restoration: comparing passive, plantation and nucleation approaches. *Biodiversity and Conservation*, *25*, 2–15.

Bikram, M., Wildlife, S., Relationship, T., Emergence, G., Kunda, P., & Shrestha, M. B. (2017). The History , Causes and Trends of Forest Degradation in Nepal A Review Paper, (September).

Bohlman, S., & Pacala, S. (2012). A Forest Structure Model that Determines Crown Layers and Partitions Growth and Mortality Rates for Landscape-Scale Applications of Tropical Forests. *Journal of Ecology*, *100*, 508–518

Brudvig, L. ., & Mabry, C. . (2008). Trait-based filtering of the regional species pool to guide understory plant reintroductions in Midwestern Oak Savannas, U.S.A. *Restoration Ecology*, *16*, 290–304.

Burak, K. ., Roy, S. ., Matthias, M. ., Craig, M., & Pauline, F. . (2011). Response of Plant Species and Life Form Diversity to Variable Fire Histories and Biomass in the Jarrah Forest of South-West Australia. *Austral Ecology*, *37*, 330–338.

Carandang, P, A., A, L., Bugayong, C, P., Dolom, … Villanueva, M. B. (2013). Analysis of Key Drivers of Deforestation and Forest Degradation in the Philippines. *Department of Environment and Natural Resources*, 1–128.

Chakravarty, S., Ghosh, S. K., & Suresh, C. P. (2011). Deforestation : Causes , Effects and Control Strategies. *Department of Forestry*, 3–29.

Chapman, C. A., Chapman, L. J., Hartter, J., Jacob, A. L., Lwanga, J. S., Omeja, P., … Twinomugisha., D. (2011). Complex responses to climate and anthropogenic changes: an evaluation based on long-term data from Kibale National Park, Uganda. Climate change in the Albertine Rift. *Hauppauge, NY: Nova Science Publishers.*, 70–87.

Chapman, C. A., Chapman, L. J., & Kaufman, L. E. S. (2009). Potential causes of arrested succession in Kibale National Park , Uganda : growth and mortality of seedlings. *Africa Journal of Ecology*, *37*, 81–92.

Chase, J. M., & Myers, J. A. (2011). Disentangling the importance of ecological niches from stochastic processes across scales. *Biological Sciences*, *366*(1576), 2351–2363.

Chazdon. (2013). Making tropical succession and landscape reforestation successful. *J Ournal of Sustainable Forest*, *32*, 649–658.

Chazdon, R. (2015). Chance and determinism in tropical forest succession. In: Carson WP, Schnitzer SA (eds) Tropical forest community ecology. *Wiley-Blackwell, Oxford*, 384–408.

Chazdon, R., Brancalion, P., Lamb, D., Laestadius, L., Calmon, M., & Kumar, C. (2015). A policy-driven knowledge agenda for global forest and landscape restoration. *Conserv Lett.*, *0*, 1–8.

Chazdon, R. L. (2014). The promise of tropical forest regeneration in an age of deforestation. *University of Chicago Press*.

Chazdon, R. L. (2016). Natural regeneration in the context of large-scale forest and landscape restoration in the tropics. *The Association for Tropical Biology and Conservation*, *48*, 709–715.

Chazdon, R. L., & Guariguata, M. R. (2016). Natural regeneration as a tool for large-scale forest restoration in the tropics: Prospects and challenges. *Biotropica*, *48*, 716–730.

Cole, R., Holl, K., Keene, C., & Zahawi, R. (2011). Direct seeding of late-successional trees to restore tropical montane forest. *Fororest Ecolology Management*, *261*, 1590–7.

Corbin, J. D., & Holl, K. D. (2012). Applied nucleation as a forest restoration strategy. *Forest Ecology and Management*, *265*, 37–46.

Corlett, R. (2014). The ecology of tropical East Asia. *Oxford University Press*, *2*.

Crotti, R., & Misrahi, T. (2015). *The Travel & Tourism Competitiveness Report 2015*. (R. Crotti & T. Misrahi, Eds.).

Crouzeilles, R., Ferreira, M. S., Chazdon, R. L., Lindenmayer, D. B., Sansevero, J. B. B., Monteiro, L., … Strassburg, B. B. N. (2017). Ecological restoration success is higher for natural regeneration than for active restoration in tropical forests. *Science Advanced*, 1–8.

Dent, D., DeWalt, S., & Denslow, J. (2012). Secondary forests of central Panama increase in similarity to oldgrowth forest over time in shade tolerance but not species composition. *Journal of Vegetation ScienceVeg Science*, *24*, 530–542.

Devictor, V., Clavel, J., Julliard, R., Lavergne, S., Mouillot, D., Thuiller, W., … Mouquet, N. (2010). Defining and measuring ecological specialization. *Journal of Applied Ecology*, *47*, 15–25.

Eggeling, W. J. (1940). *The indigenous trees of Uganda*. Entebbe: Government Printer.

FAO. (2016). State of the world's forests. Forests and Agriculture: Land-use challenges and opportunities.

Fetzer, I., Johst, K., Schäwe, R., Banitz, T., Harms, H., & Chatzinotas, A. (2015). The extent of functional redundancy changes as species ' roles shift in different environments. *Department of Environmental Microbiology*, *112*, 14888–14893.

Fisher, J. A., Cavanagh, C. J., Sikor, T., & Mwayafu, D. M. (2018). Linking notions of justice and project outcomes in carbon offset forestry projects: Insights from a comparative study in Uganda. *Land Use Policy*, *73*, 259–268.

Ford, H., Garbutt, A., Ladd, C., Malarkey, J., & Skov, M. W. (2016). Soil stabilization linked to plant diversity and environmental context in coastal wetlands. *Journal of Vegetation Science*, *27*, 259–268.

Galindo-Rodriguez, C., & Roa-Fuentes, L. L. (2017). Seed desiccation tolerance and dispersal in tropical dry forests in Colombia: Implications for ecological restoration. *Forest Ecology and Management*, *404*, 289–293.

Ghazoul, J., Burivalova, Z., Garcia-ulloa, J., & King, L. A. (2015). Review Conceptualizing Forest Degradation. *Trends in Ecology & Evolution*, *30*, 622–632.

Gibson, L., Lee, T., & Koh, L. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature*, *478*, 378–383.

Grabherr, G., Reiter, K., & Willner, W. (2013). Towards objectivity in vegetation classification: the example of the Austrian forests. *Plant Ecology*, *169*, 21–34.

Hall JS, Love BE, Garen EJ, Slusser JL, Saltonstall K, M. S. (2011). Tree plantations on farms: Evaluating growth and potential for success. *Restoration Ecology*, *261*, 1675–83.

Hamilton, A. C., Karamura, D., & Kakudidi, E. (2016). Plant Diversity History and conservation of wild and cultivated plant diversity in Uganda : Forest species and banana varieties as case studies. *Plant Diversity*, *38*, 23–44.

He, J., Ho, M., & Xu, J. (2015). Participatory selection of tree species for agroforestry on sloping land in North Korea. *Restoration Development*, *35*, 318–27.

Holl, K. D., & Aide, T. M. (2011). When and where to actively restore ecosystems? *Forest Ecology*, *261*, 1558–1563.

Host, G. ., & Pregitzer, K. . (2010). Ecological species groups for upland forest ecosystems of northwestern Lower Michigan. *Forest Ecology*, *43*, 87–102.

IFER [Institute of Forest Ecosystem Research]. (1994). Field-Map - Tool designed for computer aided field data collection.

Jacob, A. L., Lechowicz, M. J., & Chapman, C. A. (2017). Non-native fruit trees facilitate colonization of native forest on abandoned farmland. *Restoration Ecology*, *25*, 211–219.

Jacobs, D., Oliet, J., Aronson, J., Bolte, A., Bullock, J., & Donoso, P. (2015). Restoring forests: What constitutes success in the twenty-first century? *New Forests*, *46*, 601–14.

Kashian, D. ., Barnes, B. ., & Walker, W. . (2013). Ecological species groups of landform-level ecosystems dominated by jack pine in northern Lower Michigan, USA. *Plant Ecology*, *166*, 75–91.

Katende, A. B., Birnie, A., & Tengnas, B. (1995). *Useful trees and shrubs for Uganda: identification, propagation and management for agricultural and pastoral communities*. Nairobi, Kenya: Regional Soil Conservation Unit.

Kauffman, J., Case, R., Lytjen, D., Otting, N., & Cummings, D. (2009). Ecological approaches to riparian restoration in northeast Oregon. *Restoration Management Note*, *13*, 12–15.

Keenelyside, K., Dudley, N., Cairns, S., Hall, C., & Stolton, S. (2012). Ecological restoration for protected areas: principles, guidelines and best practices. *Gland: IUCN*.

Kelty, M. (2010). The role of species mixtures in plantation forestry. *Forest Ecology and Management*, *333*, 195–204.

Kindt, R., John, I., Ordonez, J., Smith, E., Orwa, C., & Mosoti, B. (2016). Agroforestry species switchboard: a synthesis of information sources to support tree research and development activities. *World Agroforestry Centre*.

Kinyanjui, M. ., Latva-Käyrä, P Bhuwneshwar, P. ., Kariuki, P., Gichu, A., & Wamichwe, K. (2014). an Inventory of the Above Ground Biomass in the Mau Forest Ecosystem, Kenya. *Open Journal of Ecology*, *4*, 619–627.

Knowles, O., & Parrotta, J. (2011). Amazonian forest restoration: an innovative system for native species selection based on phenological data and field performance indices. *Columbia University Press, New York*, *74*, 230–43.

Konz, N., Baenninger, D., Konz, M., Nearing, M., & Alewell, C. (2010). Process identification of soil erosion in steep mountain regions. *Hydrology and Earth System Sciences*, *14*, 675–686.

Lamb, D. (2011). Regreening the Bare Hills: tropical forest restoration in the Asia-Pacfic region.

Latawiec, E., Crouzeilles, R., Brancalion, P. H. S., Rodrigues, R. R., Sansevero, J. B., Santos, J. S., … Strassburg, B. B. (2016). Natural regeneration and biodiversity: A global meta-analysis and implications for spatial planning. *Biotropica*, *48*, 844–855.

Lennox MS, Lewis DJ, Jackson RD, Harper J, Larson S, T. K. (2011). Development of vegetation and aquatic habitat in restored riparian sites of California's north coast rangelands. *Restoration Ecology*, *19*, 225–233.

Liebsch, D., Marques, M., & Goldenberg, R. (2008). How long does the Atlantic rainforest take to recover after a disturbance? Changes in species composition and ecological features during secondary succession. *Biological Conservation*, *141*, 1717–1725.

Lin, H. E. X., & Tian, Y. H. A. N. G. (2010). What determines the number of dominant species in forests ?, *21*, 287–288.

Löf, M., Ersson, B. T., Hjältén, J., Nordfjell, T., Oliet, J. A., & Willoughby, I. (2015). Site preparation techniques for forest restoration. *Restoration of Boreal and Temperate Forests*, 85–102.

Lohbeck, M., Poorter, L., Mart, M., & Rodriguez-, J. (2014). Changing drivers of species dominance during tropical forest succession, 1052–1058.

Ma, K. H., & Letcher, S. G. (2017). Opposing mechanisms affect taxonomic convergence between tree assemblages during tropical forest succession. *Ecology Letters*, *20*, 1448–1458.

Maes, J., Liquete, C., Teller, A., Erhard, M., Luisa, M., Barredo, J. I., … Lavalle, C. (2016). An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. *Ecosystem Services*, *17*, 14–23.

Magurran, A. . (2013). Ecological Diversity and Its Measurement. Chapman and Hall, London.

Mansourian, S., Stanturf, J. A., Afua, M., Derkyi, A., & Engel, V. L. (2017). Forest Landscape Restoration : increasing the positive impacts of forest restoration or simply the area under tree cover ? *Restoration Ecology*, 1–6.

Meli, P., & Dirzo, R. (2013). Effects of grasses on sapling establishment and the role of transplanted saplings on the light environment of pastures: implications for tropical forest restoration. *Applied Vegetation Science*, *16*, 296–304.

Meli, P., Holl, K. D., Benayas, Rey, J. M., Jones, H. P., Jones, P. C., … Mateos, D. M. (2017). A global review of past land use, climate, and active vs. passive restoration effects on forest recovery. *PLOS ONE*, *12*, 171–368.

Meli, P., Martínez-Ramos, M., & Rey-Benayas, J. M. (2013). Selecting Species for Passive and Active Riparian Restoration in Southern Mexico. *Restoration Ecology*, *21*, 163–165.

Meli, P., Martínez-Ramos, M., Rey-Benayas, J. M., & Carabias, J. (2014). Combining ecological, social and technical criteria to select species for forest restoration. *Applied Vegetation Science*, *17*, 744–753.

Meli, P., Rey-Benayas, J. M., Carabias, J., Ruiz, L., & Martınez Ramos, M. (2013). Restauracion de los servicios ecosistemicos ribere~nos. Meta-analisis global y un estudio de caso en Chiapas, Mexico. In: Lara, A., Laterra, P., Manson, R. & Barrantes, G. (eds.) Servicios ecosistemicos hıdricos en America Latina y el Caribe. *Red ProAgua– CYTED, Valdivia, CL.*, *39*, 39–58.

Meng, G., Chai, Y., Yuan, C., Ai, H., Li, G., & Wang, Q. (2013). Community characteristics of the mid-montane humid ever-green broad-leaved forest in Gaoligong Mountains, Yunnan (in Chinese). *Scientia Silvae Sinicae*, *49*, 144–51.

Morrison, E. B., & Lindell, C. A. (2011). Active or Passive Forest Restoration ? Assessing Behavior, *19*, 170–177.

Naghi, M., Pourbabaei, H., & Dey, D. C. (2014). Ecological species group — Environmental factors relationships in unharvested beech forests in the north of Iran. *Ecological Engineering*, *69*, 1–7.

Nichol, J. E., Abbas, S., & Fischer, G. A. (2017). Spatial patterns of degraded tropical forest and biodiversity restoration over 70-years of succession. *Global Ecology and Conservation*, *11*, 134–145.

Norden, N., Chazdon, R., Chao, A., Jiang, Y., & Vı´lchez-Alvarado, B. (2009). Resilience of tropical rain forests: tree community reassembly in secondary forests. *Ecology Letters*, *12*, 385–394.

Oliveira, M. L. F., Daher, R. F., Menezes, B. R. S., & Vivas, M. (2017). Genetic diversity of elephant grass (Cenchrus purpureus [Schumach .] Morrone) for energetic production based on quantitative and multi-category traits. *Agricultural Research*, *77*, 1–10.

Ortega-Pieck, A., Lo´pez-Barrera, F., Ramı´rez-Marcial, N., & Garcı´a-Franco, J. (2011). Early seedling establishment of two tropical montane cloud forest tree species: the role of native and exotic grasses. *Forest Ecology and Management*, *261*, 1336–1343.

Ostertag, R., Warman, L., Cordell, S., & Vitousek, P. (2015). Using plant functional traits to restore Hawaiian rainforest. *Journal of Applied Ecology*, *52*, 805–9.

Pardini, R., Bueno, A., Gardner, T., Prado, P., & Metzger, J. (2010). Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. *PLoS One*, *5*, 1–10.

Pardini, R., de Bueno, A. A., Gardner, T. A., Prado, P. I., & Metzger, J. P. (2010). Beyond the fragmentation threshold hypothesis: Regime shifts in biodiversity across fragmented landscapes. *PLoS ONE*, *5*.

Pereki, H., Wala, K., Thiel-clemen, T., Bessike, M. P. B., Zida, M., Dourma, M., … Akpagana, K. (2013). Woody species diversity and important value indices in dense dry forests in Abdoulaye Wildlife Reserve (Togo , West Africa), *5*, 358–366.

Plumptre, A. J., Ayebare, S., Pomeroy, D., Tushabe, H., Nangendo, G., Mugabe, H., … Nampindo, S. (2017). Conserving Uganda's Biodiversity: Identifying critical sites for threatened species, (September), 25.

Prideaux, B., McNamara, K. E., & Thompson, M. (2012). The irony of tourism: Visitor reflections of their impacts on Australia's World Heritage rainforest. *Journal of Ecotourism*, *11*, 102–117.

Purnamasari, R. S. (2010). Dynamics of small-scale deforestation in Indonesia: examining the effects of poverty and socio-economic development. *Unasylva*, *61*, 14–20.

Putz, F. E., Blate, G. M., Redford, K. H., Fimbel, R., & Robinson, J. (2011). Tropical forest management and conservation of biodiversity. *An Overview. Conservation Biology*, *15*, 7–20.

Ralston, B. E., & Sarr, D. A. (2017). Case studies of riparian and watershed restoration in the southwestern United States—Principles, challenges, and successes. *U.S. Geological Survey Open-File Report*, *1091*, 116.

Raman, T., Mudappa, D., & Kapoor, V. (2015). Restoring rainforest fragments: Survival of mixed-native species seedlings under contrasting site conditions in the Western Ghats, India. *Restoration Development*, *17*, 137– 47.

Ranjitkar, S., Sujakhu, N., Lu, Y., Wang, Q., Wang, M., & He, J. (2016). Climate modelling for agroforestry species selection in Yunnan Province, China. *Environ Modell Softw*, *75*, 263–72.

Ren, H., Yang, L., & Liu, N. (2012). Nurse plant theory and its application in ecological restoration in lower subtropics of China. *Program for Natural Science*, *18*, 137–42.

Rocha, R., Ovaskaine, O., López-baucells, A., Farneda, F. Z., Sampaio, E. M., Bobrowiec, P. E. D., … Meyer, C. F. J. (2018). Secondary forest regeneration benefits old-growth specialist bats in a fragmented tropical landscape. *Scientific Reports*, (October 2017), 1–9.

Román-Dañobeytia, F. J., Levy-Tacher, S. I., Aronson, J., Rodrigues, R. R., & Castellanos-Albores, J. (2012). Testing the Performance of Fourteen Native Tropical Tree Species in Two Abandoned Pastures of the Lacandon Rainforest Region of Chiapas, Mexico. *Restoration Ecology*, *20*, 378–386.

Romero-sanchez, M. E. (2017). Assessing and Monitoring Forest Degradation in a Deciduous Tropical Forest in Mexico via Remote Sensing Indicators. *Forests*, *302*, 1–19.

Sands, R. (2015). Forestry in a Global Context. *CABI Publishing*.

Schmink, M., & Wood, C. (2009). Contested Frontiers in Amazonia. *Columbia University Press, New York*.

Shukla, G. (2010). Biomass production and vegetation analysis of Chilapatta Reserve Forest Ecosystem of West Bengal. Ph. D. Thesis, Uttar Banga Krishi Viswavidyalaya, Pundibari.

Singh, S., Malik, Z. A., & Sharma, C. M. (2016). Journal of Asia-Paci fi c Biodiversity Tree species richness , diversity , and regeneration status in different oak (Quercus spp .) dominated forests of Garhwal Himalaya , India. *Journal of Asia-Pacific Biodiversity*, *9*, 293–300.

Specht, M., Pinto, S., Albuqueque, U., Tabarelli, M., & FP., M. (2015). Burning biodiversity: Fuelwood harvesting causes forest degradation in human-dominated tropical landscapes. *Global Ecology and Conservation*, *3*, 200–9.

Springer, Elliott, S., Dordrecht, Blakesley, D., & Hardwick, K. (2013). Restoring tropical forests: a practical guide. *Royal Botanical Garden, Kew*.

Ssekuubwa, E., Loe, L. E., Sheil, D., Tweheyo, M., & Moe, S. R. (2017). Comparing seed removal rates in actively and passively restored tropical moist forests. *Restoration Ecology*, (November), 1–9.

Suding, K. N. (2011). Toward an Era of Restoration in Ecology : Successes , Failures , and Opportunities Ahead. *Restoration Ecology*, 465–487.

Swinfield, T., Afriandi, R., Antoni, F., & Harrison, R. (2016). Accelerating tropical forest restoration through the selective removal of pioneer species. *Forest Ecology and Management*, *381*, 8.

Tang, C., Chiou, C., Lin, C., JR, L., Hsieh, C., & Tang, J. (2013). Plant diversity patterns in subtropical evergreen broad-leaved forests of Yunnan and Taiwan. *Ecological Restoration*, *28*, 81–92.

Tang, C., Zhao, M.-H., Li, X.-S., Ohsawa, M., & Ou, X.-K. (2010). Secondary succession of plant communities in a subtropical mountainous region of SW China. *Ecological Restoration*, *25*, 149–61.

UWA-FACE. (2011). *Project plan of operation report for January–December 2011*. Kampala, Uganda.

Wamelink, G. W. W., Goedhart, P. W., Josep, Y., & Frissel. (2014). Why some species are rare.

Winterbottom, R. (2014). Restoration: It's About More than Just the Trees. *Restoration Ecology*.

Zahawai, R., Holl, K., & Reid, J. (2014). Hidden costs of passive restoration. *Restoration Ecology*, *22*, 284–287.

APPENDICES

Budget

Item	Unit cost (Ush)
Travel	200,000
Feeding	200,000
Services (Photocopying, Printing, Secretarial, Binding)	130,000
Stationery	150,000
Total	**680000**

Dissertation work plan for the year 2018