

Investigating the impact of fire on the natural regeneration of woody species in dry and wet Miombo woodland

by
Paul Mwansa

*Thesis presented in fulfilment of the requirements for the degree of
Master of Science of Forestry and Natural Resource Science in the
Faculty of AgriSciences at Stellenbosch University*



UNIVERSITEIT
iYUNIVESITHI
STELLENBOSCH
UNIVERSITY

100
1918 · 2018

Supervisor: Prof Ben du Toit
Co-supervisor: Dr Vera De Cauwer

March 2018

Declaration

By submitting this thesis electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the sole author thereof (save to the extent explicitly otherwise stated), that reproduction and publication thereof by Stellenbosch University will not infringe any third party rights and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

March 2018

Copyright © 2018 Stellenbosch University

All rights reserved

Abstract

The miombo woodland is an extensive tropical seasonal woodland and dry forest formation in extent of 2.7 million km². The woodland contributes highly to maintenance and improvement of people's livelihood security and stable growth of national economies. The woodland faces a wide range of disturbances including fire that affect vegetation structure.

An investigation into the impact of fire on the natural regeneration of six tree species was conducted along a rainfall gradient. *Baikiaea plurijuga*, *Burkea africana*, *Guibourtia coleosperma*, *Pterocarpus angolensis*, *Schinziophyton rautanenii* and *Terminalia sericea* were selected on basis of being an important timber and/or utilitarian species, and the assumed abundance. The objectives of the study were to examine floristic composition, density and composition of natural regeneration; stand structure and vegetation cover within recently burnt (RB) and recently unburnt (RU) sections of the forest. The study was conducted at Kanovlei, Caprivi, Sesheke and Kabompo State Forests, along a rainfall gradient from Namibia to Zambia. A systematic random sampling, using a 3-plot cluster, was replicated nine times. A 2 x 4 factorial treatment structure was used (two fire treatment and four rainfall classes) on the research design.

Sixty-eight vascular plant species were encountered in 25 families and 54 genera, dominated by Caesalpinioideae (15.7 %), Papilionoideae (12.9 %) and Combretaceae (11.4 %). Species richness was high and increased with MAP along the rainfall gradient. High species richness translated into high species diversity among the fire treatments and ranged from 1.5 to 2.8 Shannon diversity index units. The selected tree species contributed to stand parameters as important timber species with importance value indices from 4.5 % to 44.8 % across the study sites. Woody species showed high regeneration density across fire treatments and ranged between 3 039 and 18 274 individuals per hectare with tree species contributing up to 55 %. Fire treatment ($p < 0.0001$) and fire frequency ($p = 0.027$) had a statistically significant relationship with regeneration density. All selected tree species were recruiting, evidenced by the > 1 recruitment ratios and negative regression slopes for size class distributions, however the latter also revealed that the populations of some of species are unstable and declining. Basal area was small and not significantly different among fire treatments at all sites except at Sesheke where recently burnt sites had significantly higher basal area. Furthermore, the basal area of the Sesheke ($8.2 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$) which was significantly greater than Kanovlei ($1.2 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$), Caprivi ($1.7 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$) and Kabompo ($3.2 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$) suggested that anthropogenic influences (forest harvesting and disturbance) are having an effect on the stand structure. Canopy cover varied from 20 % to 75 % in the forests showing an increase in forest cover from drier to wetter sites.

The study proved fire treatment did not significantly affect or alter floristic composition of woody vegetation in the short-term. It also showed good recruitment for the selected tree species (as coppice or seedling) but lacking rejuvenation of the population structure. Fire treatment on vegetation cover is potentially useful and could be advantageous to regeneration of tree species for a few years. Hence, the study calls for a long-term investigation of natural regeneration of tree species taking into account the factors that could not be considered in this study.

Key words: Natural regeneration, miombo woodlands, rainfall gradient, fire treatment, size class distributions, floristic composition, fire frequency

Opsomming

Die Miombo is 'n seisoonaal-droë tropiese bosformasie wat strek oor 'n uitgebreide gebied van ongeveer 2.7 miljoen km² in Afrika. Hierdie bosland dra by tot die lewensonderhoud van mense asook die stabiele groei van nasionale ekonomieë. Die bos word bedreig deur 'n reeks van versteurings (o.a. vuur) wat die plantegroei struktuur negatief kan beïnvloed.

'n Ondersoek na die effek van vuur op die natuurlike verjonging van ses boomspesies is gedoen op 'n gradiënt van droë na vogtiger groeiplekke. *Baikiaea plurijuga*, *Burkea africana*, *Guibourtia coleosperma*, *Pterocarpus angolensis*, *Schinziophyton rautanenii* and *Terminalia sericea* is gekies op grond van hulle geskatte verspreidingsdigtheid asook hul belang vir hout en menslike benutting. Die doel van die studie was om die effek van vuur te ondersoek (in onlangs gebrande en onlangs ongebrande gebiede) op die samestelling, vegetasiebedekking, digtheid en struktuur van opstande, asook die digtheid die samestelling van natuurlike regenerasie. Die studie is onderneem op 'n reënval gradiënt wat strek oor Kanovlei, Caprivi, Sesheke and Kabompo staatsbosse van Namibië en Zambië. Sistematiese bemonstering is gedoen m.b.v. gepaarde, ewekansig gekose persele in 9 herhalings. Die oorhoofse behandelingstruktuur is 'n 2 x 4 faktoriaal (2 vuur behandelings en 4 reënval klasse).

Ses en tagtig vaatplante, versprei oor 25 families en 54 genera is waargeneem, met Caesalpinioideae (15.7%), Papilionoideae (12.9%) en Combretaceae (11.4%) die dominante groepe. Daar was 'n ryke spesieverskeidenheid, en dit het toegeneem met toenemende reënval. Die gevolg was hoë diversiteit tussen vuurbehandelings (vanaf 1.5 tot 2.8 eenhede op die *Shannon*-diversiteitsindeks). Die gekose spesies se bydrae tot die indeks van belangrikheid strek van 4.5 tot 44.8% oor studiepersele heen. Houtagtige spesies toon 'n hoë digtheid van regenerasie (saailinge + stomplote), met tussen 3 039 en 18 274 individue per hektaar waarvan boomspesies 55% bydra. Die onlangse teenwoordigheid ($p < 0.0001$) en frekwensie ($p = 0.027$) van vuur het 'n beduidende effek op digtheid van regenerasie. Al die gekose spesies toon tekens van effektiewe hervestiging soos bevestig deur hervestigingsverhoudings > 1 en negatiewe regressie gradiënte, alhoewel grootteklas verspreidings onthul dat die populasie van sommige spesies klaarblyklik onstabiel is of taan.

Die basale oppervlakte het nie beduidend verskil tussen vuurbehandelings nie, behalwe by die Sesheke groeiplek waar onlangs ongebrande groeiplekke 'n beduidend groter basale oppervlak gehad het. Verder, die basale oppervlak van die Sesheke groeiplek ($8.2 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$) was beduidend groter as Kanovlei ($1.2 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$), Caprivi ($1.7 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$) of Kabompo ($3.2 \pm 0.2 \text{ m}^2 \text{ ha}^{-1}$), wat aandui dat mensgemaakte invloede (bos ontginning en versteuring) 'n invloed het op laasgenoemde drie. Die kroondak bedekking varieer van 20 tot 75% met 'n toename van droër na vogtiger groeiplekke.

Die studie het bewys dat vuur nie beduidende korttermyn veranderinge in die floristiese samestelling van die houtagtige plantegroei teweeg gebring het nie. Daar is voldoende aanvanklike regenerasie van die gekose spesies (as saailinge en stomplote), maar daar is nie altyd suksesvolle verjonging (van die populasiestruktuur) nie. Die gebruik van vuur kan bruikbaar wees om plantegroei te bestuur en kan voordelig wees vir regenerasie van sommige spesies vir 'n beperkte aantal jare. Dit is egter noodsaaklik om langtermyn studies op die regenerasie van die spesies te doen ten einde faktore in ag te neem wat nie in hierdie studie gedek kon word nie.

Sleutelwoorde: Natuurlike regenerasie, miombo bosland, reënval gradiënt, vuurbehandeling, grootteklas verspreidings, plantegroei samestelling, brand frekwensie

Dedication

To

My wife Margaret and our daughter Mwenya;

For your prayers, hope, love and support for me. I cannot adequately repay for the loneliness you endured during my absence from home as a husband and father. There was never a moment that went by without constantly thinking about you and the many sacrifices you had to make on daily basis.

And to the memory of

Uncle J C Chikonda;

Even in death your inspirations and values drives me to do what is always best for humanity. You will always be in my heart to the very end of my time.

“ For one man who thanks God that he is not as other men there are a few thousand to offer thanks that they are as other men, sufficiently as others to escape attention.”

John Dewey

Acknowledgements

All praise and glory be to God the Almighty for His gift of life and many graces shown unto me to enable me successfully complete this work.

I would like to express my sincere gratitude to the following persons and institutions, in no particular order, for their contributions to the success of this project. It is virtually impossible to mention everyone that have contributed to the success of my studies. Sincere gratitude to my supervisors, Prof Dr Ben du Toit and Dr Vera De Cauwer, for their mentorship, support and motivation during my studies. I am greatly indebted to them. I am grateful to The Southern African Scientific Service Centre for Climate change and Adaptive Land use (SASSCAL), whose bursary funded my study under TASK ID 038. Mr Martin Kambayi, who brought to light and my attention the opportunity to undertake this study in dry land forests of Namibia and Zambia, I am so thankful chief. Mr Anton Kunneke, for the invaluable input with GIS use in my study. The door was always open to me and always assisted without reservations even at short notice. Prof Daan Nel for the invaluable input on research design and data analysis. Dr Sylvanus Mensah and Mr Stephan van der Westhuizen, for imparting the skills of using R statistical software and always providing “an eye over my shoulder” during my data analysis. I could not have achieved it without your invaluable inputs. Ms Albé Van Niekerk and Ms Ursula Petersen, for the administrative support during my studies; much appreciated for the caring assistance.

I am so grateful to those who helped with fieldwork and who gave me their time, the most thoughtful gift of all: Ms Miya Kabajani, invaluable assistance during the data collection in Namibia and ensured that my work and stay in Namibia was pleasant and enjoyable. Prof Poor Phiri and the Herbarium staff for assisting immensely in species identification of the species. Messrs Chaka Mubita and Werner Mbongo, for all the assistance you provided during the field excursion in Namibia and you opened many doors without which the work could not have been possible. The staff at Kanovlei forest research station led by Mr Wilson Muyenga. My since gratitude also go to Mr Martin Heiki who assisted us dearly with fieldwork at Kanovlei state forest. Ms Justina Ngoma, for coming to my aid when I needed information that has gone a very long way in shaping this work. Thanks to Messrs Shadreck Bwalya and Amon Litondo at Kabompo forest office. My gratitude to Messrs Eric Mukwabataba and Ignatius Sikapoto at Sesheke forest office. I sincerely thank Messrs Bwalya Ponga and Derrick Simukanzye, my immediate supervisors who went out of their way to grant me study opportunity and leave of absence from employment.

I am greatly indebted to the Council of Scientific and Industrial Research (CSIR) on the MODIS burned area product for the study area, Namibia Metrological Service (NMS) on the rainfall and

temperature data for the study sites in Namibia and the Zambia Meteorological Department (ZMD) on the rainfall and temperature data for the study sites in Zambia.

Thank you to all the staff of the Department of Forestry and Wood sciences at Stellenbosch University for the support and creating an atmosphere of camaraderie in the Department especially on the seminar presentations and the birthday cakes we shared every last Friday of the month. To my fellow postgraduate students, especially my office mates, I say thank you so very much for the friendship and support during my studies.

Special appreciations to Gabriel Motsatsi who went out of his way to show us the hospitality of the people of South Africa outside academic circles. To you too Francis Munalula, I say thank so much for providing sense of being at home away from home enjoying every bit of the moments we shared in these lands south of the Limpopo River. Last and not the least, I am humbled and appreciate the kind gesture by my family and friends for all the prayers and support to see to it that I successfully complete my studies.

Table of Contents

Declaration	i
Abstract	ii
Opsomming	iv
Dedication	vi
Acknowledgements	vii
Table of Contents	ix
List of Figures	xii
List of Tables	xvi
List of Acronyms and Abbreviations	xviii
Chapter 1 INTRODUCTION	1
1.1. General background	1
1.2. Significance of the study	3
1.3. Research/problem Statement	4
1.4. Research question	5
1.5. Aim and Objectives of the Study	5
1.6. Scope of study	7
1.7. Assumptions	8
1.8. Brief Chapter Overview	8
1.9. Chapter summary	8
Chapter 2 LITERATURE REVIEW	9
2.1. Introduction	9
2.2. Zambezian Phytoregion	9
2.3. Natural regeneration	15
2.4. Forest fire	17
2.5. Selected important tree species	22
2.6. Chapter summary	29

Chapter 3 RESEARCH DESIGN AND METHODOLOGY	30
3.1. Introduction	30
3.2. Study location.....	30
3.3. Vegetation	32
3.4. Climatic conditions.....	34
3.5. Soil conditions	37
3.6. Research design.....	38
3.7. Plot design	40
3.8. Field work and sampling	42
3.9. Data processing and analyses	48
3.10. Chapter summary.....	57
Chapter 4 RESULTS.....	58
4.1. Introduction	58
4.2. Floristic composition.....	58
4.3. Density and composition of natural regeneration.....	66
4.4. Stand structure	82
4.5. Vegetation cover.....	107
4.6. Chapter summary	111
Chapter 5 DISCUSSION	113
5.1. Introduction	113
5.2. Floristic composition.....	113
5.3. Natural regeneration	116
5.4. Stand structure	118
5.5. Vegetation cover.....	120
5.6. Selected species	122
5.7. Chapter summary	135
Chapter 6 CONCLUSIONS AND RECOMMENDATIONS.....	136
6.1. Introduction	136

6.2. Conclusions and implications for management.....	136
6.3. Recommendations for future research.....	139
REFERENCES.....	141
APPENDICES	157

List of Figures

Figure 2.1: The extent of the Zambebian phytoregion (domain) in relation to other major phytoregions in sub-Sahara (N. Fauvet CIRAD-Forét 1996).	10
Figure 2.2: Approximate distribution of forests and woodlands in Namibia. Land cover of 10% is under forests, 50% under woodlands and 40% under shrub-lands and deserts (Mendelsohn and el Obeid, 2005).....	12
Figure 3.1: The location of the study countries in Southern Africa (top left) and the location of the study sites (state forests) in their respective countries of Zambia (top right) and Namibia (bottom right).....	30
Figure 3.2: The location of sample plots at the respective study sites in Namibia (top left & right) and Zambia (bottom left & right).	32
Figure 3.3: Vegetation structure in all the four study sites where sampling took place showing a very clear distinction in vegetation structure from dry site at Kanovlei state forest to a wetter miombo at Kabompo state forest.	33
Figure 3.4: Extent of the Kalahari sand basin portrayed in the work of De Cauwer et al. (2014) based on the data of Mendelsohn and el Obeid (2004).	38
Figure 3.5: Placement of sampling plots on one cluster as demonstrated in the field instructions of the National Forest Inventories of Namibia (Burkea et al., 2001).....	39
Figure 3.6: The general plot design and layout recommended in the filed instructions manual for forestry inventory in Namibia (Burke et al., 2001).....	40
Figure 3.7: The location of the two subplots within the main sample plot for shrubs and regeneration sampling based on the NFI guidelines (Burke et al., 2001).	42
Figure 3.8: The depiction of some of the materials and methods used in data collection. A) GPS geographical location data capture; B) substrate cover; C) crown and shrub cover measurement; D) tree height reading; E) Collapsible ruler used for shrub and regeneration height estimation; F) dbh measurement; G) vertex transponder for tree distance and H) data capture on record sheets.	44
Figure 3.9: Diagrammatic representation of the line intercept method as demonstrated in Kabajani (2016). The substrates numbered 1, 4 and 5 would be counted and their coverage distance recorded while 2 and 3 would be left out.....	46
Figure 3.10: An illustration of the correct positions where to measure dbh form for possible scenarios that may be encountered in the field (Burkea et al., 2001).	47
Figure 4.1: The abundance of the selected tree species across the study sites.	63
Figure 4.2: Natural regeneration density of all tree and shrub species in the recently burnt and recently unburnt plots across study sites.....	66

Figure 4.3: An interaction plot demonstrating the interaction effect of fire treatment with study site on natural regeneration density.	68
Figure 4.4: Natural regeneration density of all tree and shrub species due to interactive effect of fire frequency on the plot with site. The fire frequencies correspond to 0 burn (none), 1 – 3 burns (low), 4 – 6 burns (medium) and > 6 burns (high) during the period 2001 - 2015.	69
Figure 4.5: The fire treatment effect on natural regeneration density of <i>Baikiaea plurijuga</i> at Sesheke and Caprivi state forests.	70
Figure 4.6: The fire treatment effect on natural regeneration density of <i>Burkea africana</i> at Kanovlei, Caprivi and Kabompo state forests.	71
Figure 4.7: The fire treatment effect on natural regeneration density of <i>Guibourtia coleosperma</i> at Kabompo state forest.	72
Figure 4.8: The fire treatment effect on natural regeneration density of <i>Pterocarpus angolensis</i> at Caprivi and Kabompo state forests.	73
Figure 4.9: The fire treatment effect on natural regeneration density of <i>Schinziophyton rautanenii</i> across the study sites.	74
Figure 4.10: The fire treatment effect on natural regeneration density of <i>Terminalia sericea</i> at Kanovlei and Caprivi state forests.	75
Figure 4.11: The effect of fire frequency on the natural regeneration density of selected species across study sites were highly significant according to fire frequency categories.	76
Figure 4.12: The plot of diameter and height correlation for all the woody species with dbh \geq 5 cm across the study sites.	82
Figure 4.13 Correlation plot of diameter and height of all woody species per study site.	83
Figure 4.14: Height size class distribution of <i>Baikiaea plurijuga</i> at Sesheke and Caprivi study sites with absence of the species at Kanovlei and Kabompo study sites.	86
Figure 4.15: Height size class distribution of <i>Burkea africana</i> at Kanovlei, Caprivi and Kabompo study sites.	87
Figure 4.16: Height size class distribution of <i>Guibourtia coleosperma</i> at Kabompo study site.	88
Figure 4.17: Height size class distribution of <i>Pterocarpus angolensis</i> at Kanovlei, Caprivi and Kabompo study sites.	89
Figure 4.18: Height size class distribution of <i>Schinziophyton rautanenii</i> across the four study sites.	90
Figure 4.19: Height size class distribution of <i>Terminalia sericea</i> at Kanovlei, Caprivi and Kabompo study sites.	91

Figure 4.20: Diameter size class distribution of <i>Baikiaea plurijuga</i> at Sesheke and Caprivi study sites.	92
Figure 4.21: Diameter size class distribution of <i>Burkea africana</i> at Kanovlei and Kabompo study sites.	93
Figure 4.22: Diameter size class distribution of <i>Guibourtia coleosperma</i> at Kabompo study site.	94
Figure 4.23: Diameter size class distributions of <i>Pterocarpus angolensis</i> at Kanovlei, Caprivi and Kabompo study sites.	95
Figure 4.24: Diameter size class distribution of <i>Terminalia sericea</i> at Caprivi study site.....	96
Figure 4.25: The distribution of mean diameter and mean height of all the woody species across study sites comparing the effect of RB and RU fire treatment. The means are plotted with standard errors. Caprivi site was excluded due to heavy influence of anthropogenic effects.	97
Figure 4.26: The mean diameter at breast height (dbh) of all woody species with minimum dbh of 5 cm at Kanovlei, Sesheke and Kabompo sites.	99
Figure 4.27: The mean diameter of the selected species based on the fire frequency at the plot across the study sites.	100
Figure 4.28: The mean height distribution of all woody species as affected by fire frequency at Kanovlei, Sesheke and Kabompo study sites.....	101
Figure 4.29: The mean height distribution of the selected species following the fire frequency at Kanovlei, Sesheke and Kabompo study sites.....	102
Figure 4.30: The basal area of all woody species function of the fire treatment across the study sites	103
Figure 4.31: The distribution of basal area according the fire frequency across the study sites.	104
Figure 4.32: The Mean basal area for each of the selected species across the study sites. None of selected species had adult trees at all the study sites.	106
Figure 4.33: The basal area distribution for the selected species following the fire frequency across the study sites.	107
Figure 4.34: The percentage cover of tree crowns on the forest floor in the recently burnt and recently unburnt plots across the study sites.....	108
Figure 4.35: The crown cover of woody species on the forest floor following the fire frequency at the plots across the study sites	109
Figure 4.36: The substrate cover on the forest floor in the RB and RU plots across the study sites.	110
Figure 4.37: The substrate cover on forest floor following the fire frequency at the plots across the study sites.....	111

Figure 5.1: A dugout canoe under construction at Kabompo state forest - an additional threat to the existence and survival of the species. Four of such canoe construction activities were observed within a distance of 5 km. Photo: Paul Mwansa, 2016. 128

Figure 5.2: An Example of illegal harvest of *Pterocarpus angolensis* that has contributed to rapid decline of the population at Caprivi State Forest. Photo: Miya Kabajani, 2016..... 131

List of Tables

Table 2.1: Summary of the Forest types found in Zambia as defined by the FAO, (Mukosha and Siampale, 2009).	14
Table 3.1: Summary of the mean annual precipitation for the study sites for the period between 1990 and 2015. Source of data used in calculations of the mean is Namibia meteorological service and Zambia meteorological department.	35
Table 3.2: Summary of mean annual maximum temperature across study sites for the period between 1990 and 2015. Source of data used in calculations of the mean is Namibia meteorological service and Zambia meteorological department.....	36
Table 3.3: The average fire return interval and main fire season when most fires occur across study for recently burnt (RB) and recently unburnt (RU) sites.	37
Table 3.4: Summary of the number of sample plots and subplots where data was collected in the four study sites.	45
Table 4.1: A combined list of species richness and relative species richness for all the four study sites.	60
Table 4.2: Species diversity, dominance, evenness and equitability indices on the pooled data showing the lower and upper limits for the entire study region.	61
Table 4.3: Species diversity, dominance, evenness and equitability indices for the fire treatments across study sites.	61
Table 4.4: Summary of the relative frequency of selected species across study sites.	64
Table 4.5: Summary of the relative density of selected species across study sites.....	64
Table 4.6: Summary of the relative dominance of selected species across study sites.....	65
Table 4.7: Summary of the importance value index of the selected species across study sites.....	66
Table 4.8: Indicators of species recruitment (seedlings to sapling ratio) and species survival to the next life stage (sapling to adult ratio).....	77
Table 4.9: The different size class distribution ordinary least-squares regression slopes of selected species recruitment, survival ability to the next life stage and stability of juvenile populations across the study sites.	78
Table 4.10: Distribution of regeneration limitation of selected species across study sites.....	79
Table 4.11: The five woody species dominating regeneration compositions across study sites	81
Table 4.12 Results of correlation analysis between tree diameter and height in RB and RU across study sites	82
Table 4.13: Diameter and height correlation regression results of selected species in study sites where they occurred.....	84

Table 4.14: Results of the mean diameter and mean height of selected species at sites where they occurred. Shown in brackets are standard errors of the mean.98

List of Acronyms and Abbreviations

ANOVA	Analysis of variance
CSIR	Council for Scientific and Industrial Research
DBH	Diameter at breast height (1.3 m)
DoF	Directorate of Forestry
DRC	Democratic Republic of Congo
FAO	Food and Agricultural Organisation
GIS	Geographic Information System
GLM	Generalised linear model
GPS	Geographical Positioning System
IVI	Importance value index
MAP	Mean annual precipitation
MAT	Mean annual temperature
MODIS	Moderate resolution mapping spectrometer
NMS	Namibia Metrological Service
OLS	Ordinary least squares
PAST	Paleontological STatistics
QGIS	Quantum geographic information system
RB	Recently burnt
REDD ⁺	Reducing emissions of greenhouse gases from deforestation and forest degradation
RU	Recently unburnt
SASSCAL	Southern Africa Science Service Centre for Climate change and Adaptive Land management
SCD	Size class distribution
UN	United Nations
ZMD	Zambia Meteorological Department

Chapter 1

INTRODUCTION

1.1. General background

The miombo woodlands form the most extensive tropical seasonal woodland and dry forest formation and cover an area of approximately 2.4 to 2.7 million km² (White, 1983; Campbell, 1996; Mwabumba *et al.*, 1999; Dewees *et al.*, 2011). Namibia is at the southwestern edge of the miombo phytoregion as a tropical dry forest transition zone (De Cauwer *et al.*, 2016). On the other hand, miombo woodlands extensively cover Zambia in the Zambezian phytoregion (Chidumayo, 1997; White, 1983). The name miombo is a Swahili derivative of the *Brachystegia* species (Dewees *et al.*, 2011).

The importance of miombo woodlands cannot be overemphasised in the provision of many products and services. Forests and woodlands contribute highly to the maintenance and improvement of people's well-being, livelihood security and sustainable growth of national economies (Chidumayo & Gumbo, 2010). The woodlands may not provide high yields of quality timber but the life support roles they play have vast importance (Dewees *et al.*, 2011). Miombo woodlands' products range from medicines and food to building timber and fuel, which support rural livelihoods. In addition, towns and cities of the region are supplied with the woodlands' products, most of which is fuel (Campbell, 1996). The 2007 United Nations (UN) population estimates placed 75 million people directly depending on miombo woodlands and an additional 25 million urban dwellers relying on miombo wood or charcoal as source of energy (Dewees *et al.*, 2011). This has been directly linked with an increase of the demand for woodland resources, as a result of a fast growing population that depend on natural resources (Jew *et al.*, 2016). Such huge dependence is leading to heavy disturbance of forests (Dewees *et al.*, 2011) and fast disappearance of forests and woodlands (Chidumayo & Gumbo, 2010).

Forests and woodlands are subjected to a wide range of disturbances, which vary in duration intensity and frequency (Wiafe, 2014). Notable among them is fire, which impacts on vegetation structure (Bond & Keeley, 2005). Fire in miombo woodlands is an important ecological factor requiring management of vegetation to take it into consideration (Kikula, 1986). Generally, fires on the African landscape have been a feature for millions of years. However, fire frequency in woodlands have now increased much more than before (Mendelsohn & el Obeid, 2005). Many authors have attributed the source of fire in forests and woodlands to anthropogenic burning (Trapnell, 1959; Geldenhuys, 1977; Kikula, 1986;

Goldammer *et al.*, 2002; Bond & Keeley, 2005; Mendelsohn & el Obeid, 2005; Ryan & Williams, 2011). Fire impacts tree growth by killing regeneration and crippling young stems (Campbell, 1996), while saplings need several years of fire absence to grow tall and escape burning effects (Mendelsohn & el Obeid, 2005).

The main focus of forest activities in the pre-independence era of Namibia were on regulating commercial harvest of *Pterocarpus angolensis* locally known as Kiaat and to some extent efforts were made to establish exotic species trials at Okahandja and Grootfontein forest stations (Ministry of Agriculture, Water and Forestry, 2011). Those trials focused on species of the genera *Casuarina*, *Eucalyptus*, *Prosopis*, and a number of ornamental species meant for planting in cities and around farmsteads. A slightly different approach was used in Zambia, where organised forestry started in pre-independence era with declaration of the first forest policy in 1947 (Mukosha & Siampale, 2009). Up to now, few efforts have been made to understand regeneration characteristics of the main commercial tree species in natural forests. This research status led to the absence of any management guidelines on natural regeneration including the deliberate use of fire in management of natural forests (Ministry of Agriculture, Water and Forestry, 2011). Early fire studies in Namibia (Geldenhuys, 1977) and Zambia (Trapnell, 1959) pioneered the knowledge on the effect of fire on woodland communities. Fire studies in savannas and miombo woodlands have since followed suit in assessing fire effect on woody communities (Burke, 2002; Graz, 2003; Sankaran *et al.*, 2005; Sheuyange *et al.*, 2005; Ribeiro *et al.*, 2008; Siljander, 2009; Staver *et al.*, 2011b; Ryan & Williams, 2011; Staver *et al.*, 2011a; Nepolo & Mapaure, 2012; Ncube & Mufandaedza, 2013). Studies on natural regeneration of economically important indigenous tree species have emerged within the last few decades in the miombo woodlands. Among others that have looked at the impact of fire on natural regeneration include Kayofa (2015) and Kabajani (2016). Related natural regeneration studies (e.g. Mwabumba *et al.*, 1999; Syampungani *et al.*, 2016) looked at effects from other disturbances including charcoal burning, agriculture and selective harvesting.

Since Namibia has the driest climate in sub-Saharan Africa (Sweet & Burke, 2006), dry conditions have an influence on the regeneration, growth and yield of forest resources, a scenario that is different from Zambia and other much wetter southern African countries (Ministry of Agriculture, Water and Forestry, 2011).

It is against this background of increased pressure on natural resources and the differences in climatic conditions existing in the Zambezian phytoregion that this study aimed to investigate

the impact of fire on the natural regeneration of woody species in dry and wet miombo woodlands.

1.2. Significance of the study

1.2.1. Research in forest/woodland ecosystems

Academic research have arisen over time to include among others the study of natural regeneration of tree species, trees' response and adaption to fire in the deep Kalahari sands (Ministry of Agriculture, Water and Forestry, 2011) as a means of mitigation to forest degradation and contribute to sustainable management of natural resources. The Southern Africa Science Service Centre for Climate Change and Adaptive Land management (SASSCAL) is a joint initiative among five countries of southern Africa (Angola, Botswana, Namibia, South Africa, Zambia) and Germany in response to challenges of global change such as climate change (www.sasscal.org, 2016). SASSCAL has supported numerous academic research projects under Task ID 038, which are looking into 'Forest regeneration, growth, threats and trends in different forest types'(SASSCAL, 2016). This study is one of the main studies under Task ID 038. Thus, it aims to contribute to one of the broader objectives of the task namely "To understand the effects of environmental factors, forest stand factors and fire history on the success of natural regeneration of tree species favoured by the local communities in Namibia."

1.2.2. Ecology of wild fires

In recent times, the high incidence of wild fires turned out to be a significant phenomenon warranting research (Ministry of Agriculture, Water and Forestry, 2011). The study aims to highlight effects of wild fires on natural woody regeneration along a rainfall gradient, and is expected to contribute to the knowledge pool where remedial measures can be designed as response to the observed changes in the woodlands (Ministry of Agriculture, Water and Forestry, 2011). This is because the amount of rainfall influences the effect of fire on woody vegetation (Vera De Cauwer, pers. comm.) Studies in different sites along a rainfall gradient provides a good measure of comparing same fire effect and how it will impact woody species exposed to availability of different amounts of rainfall (Scholes et al., 2004; Sankaran *et al.*, 2005).

1.2.3. Knowledge gap

Extensive studies in dry forests especially in southern African miombo woodlands have been done for decades, focusing mainly on food security, livelihood, community forestry and conservation/development trade-offs. However, there is need for more research to fill the existing gaps such as understanding the resilience of dry forest ecosystems - a vital tool for effective management, providing more information to understand fully the link between urbanisation and dry forests concerning fuelwood and forest food. Another area of importance is adaptation research since not much has been done in southern African miombo woodlands on climate change adaptation (Blackie *et al.*, 2014). Other authors have postulated that the lack of well documented information on plant community patterns and species associations in miombo woodlands exist (Munishi *et al.*, 2011). Study results will contribute to the understanding of fire effects on woody species and their responses along a rainfall gradient. Ultimately, the study's achievements will build upon the earlier knowledge gathered by other researchers within the southern African miombo woodlands.

1.3. Research/problem Statement

Average rainfall and fires are likely to impact on the natural regeneration abilities of tree species. There is little known about the influence of fire on the regeneration of the different tree species along rainfall gradient especially within the Zambezian phytoregion. The majority of the work done along a given gradient in areas including the miombo woodlands focused on woody cover, structure and composition (Scholes *et al.*, 2002; Scholes *et al.*, 2004; Sankaran *et al.*, 2005, 2008, Staver *et al.*, 2011a,b), and species richness (O'Brien, 1993). Other studies were within country - gradients focusing on vegetation structure, (e.g. Shackleton *et al.*, 1994; Banda *et al.*, 2006). Studies on natural regeneration within the miombo woodlands were localised within respective countries, Malawi (Mwabumba *et al.*, 1999), Namibia (Kabajani, 2016; Kayofa, 2015), Zambia (Syampungani *et al.*, 2016), without any given gradient. However, a related study highlighted natural regeneration principles in dry and wet forests of the tropics (Vieira & Scariot, 2006).

The tree species *Baikiaea plurijuga* Harms, *Burkea africana* Hook, *Guibourtia coleosperma* (Benth.) J. Léonard, *Pterocarpus angolensis* DC, *Schinziophyton rautanenii* (Schinz) Radcl.-Sm and *Terminalia sericea* Burch. Ex DC were selected as species of focus for this study. The species were selected on the basis of being important timber and/or utilitarian species most favoured by the local people (Coates-Palgrave, 2002; Mendelsohn & el Obeid, 2005). They are abundant in Namibia (Kabajani, 2016; Mendelsohn & el Obeid, 2005; SASSCAL, 2016) and

are also known to occur across Zambia (Fanshawe, 2010). This is unlike species of the genera *Brachystegia*, *Julbernardia* and *Isoberlinia*, which, even though they are dominant in the miombo woodlands, may not be readily found in the dry woodlands of Namibia. Thus, it was important to select and focus on woody species for the study that could be found in both countries. Local communities have multiple use for these woody species, as explained further under each species in Section 2.5. However, there is also scant information on the best fire management practices that could stimulate natural regeneration so that forests would be able to cope with the pressure of providing timber and non-timber forest products.

Generally, woody cover, species richness and equitability tend to increase with rainfall (O'Brien, 1993; Scholes *et al.*, 2002) and none of these variables have any relationship with fire-return interval (Sankaran *et al.*, 2005). Woody cover revealed a highly linear relationship with rainfall, but only at the extremes of rainfall distribution, i.e. where mean annual precipitation (MAP) < 1 000 mm and again where MAP > 2 000 mm (Staver *et al.*, 2011b). The findings of Sankaran *et al.* (2005) were similar, but had a slightly different range. They revealed woody cover to relate linearly with rainfall within the range of 100 mm to 650 mm. Fire is expected to impact on vegetation within the intermediate rainfall gradients (Scholes *et al.*, 2002), strongly reducing tree cover when present (Staver *et al.*, 2011b) and generally increasing higher woody cover where there is infrequent fire occurrence with fire-return interval > 10.5 years (Sankaran *et al.*, 2005). This study, therefore, aimed at investigating how fire treatment is influencing the density and natural regeneration of woody plants along a rainfall gradient of 500 – 1 300 mm in the miombo woodlands.

1.4. Research question

The study sought to answer the following fundamental question:

Does fire treatment (period since the last fire) impact on the density and composition of natural regeneration of woody plants along a rainfall gradient in miombo woodlands?

1.5. Aim and Objectives of the Study

1.5.1. Aim of the research

The overall aim of the study was to investigate the impact of rainfall and fire treatment on the natural regeneration abilities of socio-economically important woody species of Namibia and Zambia and contribute to the knowledge pool on good fire management practices for natural forests.

1.5.2. Specific Objectives

The following objectives formed the basis of this study in order to answer the research question;

- i. To examine the effect of fire treatment on the floristic composition along the rainfall gradient.
- ii. To determine the effect of fire treatment on the density and composition of natural regeneration of woody plants along a rainfall gradient.
- iii. To assess the effect of fire treatment on the stand structure of selected woody species along a rainfall gradient.
- iv. To determine the effect of fire treatment on vegetation cover along a rainfall gradient.

1.5.3. Hypotheses and key questions

For purposes of fulfilling the specific study objectives, the following hypotheses and key questions were answered.

1. First hypothesis:

H₀: Fire treatment has no effect on floristic composition along a rainfall gradient in the miombo woodlands.

- i. Does fire treatment affect species richness along a rainfall gradient?
- ii. Do species diversity, dominance and equitability indices differ in areas with fire treatment along a rainfall gradient?
- iii. Does fire treatment affect species similarity along a rainfall gradient?
- iv. How much does fire treatment affect species community parameters along a rainfall gradient?

2. Second hypothesis:

H₀: There is no difference in density and composition of natural regeneration of woody species in areas with different fire treatment along a rainfall gradient.

- i. How much does regeneration density of woody species differ along a rainfall gradient due to fire treatment?
- ii. How much does regeneration density of selected tree species differ along a rainfall gradient due to fire treatment?
- iii. What are the differences in recruitment patterns of selected tree species along a rainfall gradient due to fire treatment?
- iv. How does fire treatment affect regeneration limitation of selected tree species along a rainfall gradient?

3. Third hypothesis:

H₀: Stand structure of selected tree species does not differ in areas with different fire treatment along a rainfall gradient.

- i. How does fire treatment affect the correlation of diameter and height of selected tree species?
- ii. How is the distribution of diameter of selected tree species in size classes affected by fire treatment along a rainfall gradient?
- iii. How is the distribution of height of selected woody plants in size classes affected by fire treatment along a rainfall gradient?
- iv. How does fire treatment affect mean diameter and mean height of selected woody plants along a rainfall gradient?
- v. What is the effect of fire treatment on basal area for selected woody species along a rainfall gradient?

4. Fourth hypothesis:

H₀: Vegetation cover is not affected by fire treatment along the rainfall gradient.

- i. What is the effect of fire treatment on canopy cover along a rainfall gradient?
- ii. What is the percentage of substrate cover on the forest floor that is influenced by fire treatment along a rainfall gradient?

1.6. Scope of study

The study focuses on the effect of fire on the natural regeneration of selected woody species along a rainfall gradient from a dry region to a wetter region in the miombo woodlands of Southern Africa. This is to assess if fire affect woody species differently in different rainfall regions. In addition, limiting the study to selected woody species was to focus on generating adequate information that would contribute to the sustainable management and value addition of socio-economic important tree species preferred by local communities in their day-to-day use. The study is limited to the miombo woodlands in both Namibia and Zambia. Similar woody species were found to occur in the miombo woodlands throughout the Zambezi phytoregion though localised variation may occur in different rainfall regions. The study was equally limited to state forests (protected) in the two countries. This was in a bid to limit (mainly) anthropogenic disturbances that may have an effect on natural regeneration.

The study did not include the assessment of soil seed banks, assisted forest regeneration, forest growth, timber volumes and soils even though they could be related to this study. The exclusion

from the study was on an account of high volume of work involved and limited time available. To undertake studies in the above-mentioned fields, one requires undertaking studies over a longer period.

1.7. Assumptions

In this study, the following were the assumptions;

- i. That the state forests in which the study was undertaken were relatively free of human disturbance. Therefore, only factors such as rainfall gradient and fire disturbance have an impact on the natural regeneration and stand characteristics of the forests.
- ii. That the ‘illegal’ selected harvest of forest and non-timber forest products has no significant bearing on the natural regeneration and stand characteristics of the forests.
- iii. That there have not been any commercial logging operations in the protected state forests.

1.8. Brief Chapter Overview

This thesis comprises seven chapters. Chapter one is the introduction containing background information, problem statement, aim and objectives, and scope of the study. Chapter two is on literature review related to the study whereas research design and methodologies used in undertaking this study are highlighted in chapter three. Results based on the objectives of the study are presented in chapter four, which are discussed in chapter five. The thesis ends with conclusion and recommendations in chapter six.

1.9. Chapter summary

In summary, the chapter has outlined the expanse of the miombo woodlands and dry forests in the Zambezian phytoregion. It highlighted the extent to which the miombo woodlands cover Namibia and Zambia including the life support roles the woodlands provide for the people who directly and indirectly depend on the woodlands’ resources. Fire and human pressure for goods and services are some of the wide range of disturbances the woodlands are now facing. In light of the disturbances, there is apparent need to understand the influence of fire on natural regeneration of woody species along a rainfall gradient. Hence the proposed roadmap to investigate the research problem by outlining the research question, research aim and specific objectives for this study.

Chapter 2

LITERATURE REVIEW

2.1. Introduction

This chapter highlights the literature important to this study. The chapter is divided into four sections with the first section focusing on the Zambezan phytoregion and the forest types associated with it. Other sections focusses on natural regeneration, forest fire and the selected tree species for the study.

2.2. Zambezan Phytoregion

The Food and Agriculture Organisation (FAO) uses a classification system based on mean height, crown cover and minimum area to define a forest. Based on the above mentioned parameters, therefore, a forest will constitute a stand size of not less than of 0.5 ha, height of not less than 5 m and a minimum crown cover of 10%, excluding land that is predominantly under agriculture or urban land use (FAO, 2012). The description include plantations and young indigenous stands expected to develop into taller groves (Mendelsohn & el Obeid, 2005). However, a country has some leeway in how to classify its forests to manage for purposes of reducing emissions of greenhouse gasses from deforestation and forest degradation (REDD⁺) (Ministry of Agriculture, Water and Forestry, 2011).

The Zambezan phytoregion is a forest domain found between latitude 3 °S and 26 °S (Figure 2.1), covering the central and southern Africa (Campbell, 1996; Dewees *et al.*, 2011; White, 1983). The region has a continental climate with summer rainfall from November to April ranging between 500 mm and 1 500 mm annually, which increases from south to north, and mean annual temperature of 18 – 24 °C (Campbell, 1996; Chidumayo, 1997). The Zambezan phytoregion was further divided into dry deciduous forest and grassland, dry evergreen forest (*Cryptosepalum*), dry Zambezan miombo woodlands, wet Zambezan woodland, and *Colophospermum mopane* woodland and scrub woodland. The phytoregion is home to 8,500 tree species of which more than 54% are endemic (Campbell, 1996; Chidumayo, 1997; Dewees *et al.*, 2011; White, 1983).

The vegetation types important to this study in the Zambezan phytoregion are the deciduous forests and woodlands commonly known as miombo. Dry deciduous forests occur in deep, freely drained soils that have adequate moisture supply during dry season (Chidumayo, 1997). The author also points out that in areas with annual rainfall of 600 to 900 mm, deciduous forest

replaces dry evergreen forest. *Baikiaea* forest on the Kalahari sands is the most extensive dry deciduous forest with *B. plurijuga* as dominant species, *Pterocarpus lucens* as subdominant and *S. rautanenii* being dominant at a local level. Woodlands are the most widespread in the Zambezan phytoregion and give it the general vegetation characteristics. They are the climax vegetation in many places of the phytoregion and secondary in other places where they are greatly modified by cultivation and fire resulting in four distinct types, namely miombo, mopane, munga and chipya (Chidumayo, 1997). Miombo woodlands are typically composed of broad-leaved trees of the subfamily Caesalpinioideae that are dominated by few species mainly from the genera *Brachystegia*, *Julbernardia* and *Isoberlinia* (White, 1983; Campbell, 1996; Chidumayo, 1997; Campbell *et al.*, 2007; Dewees *et al.*, 2011; Ministry of Agriculture, Water and Forestry, 2011; Timberlake & Chidumayo, 2011).

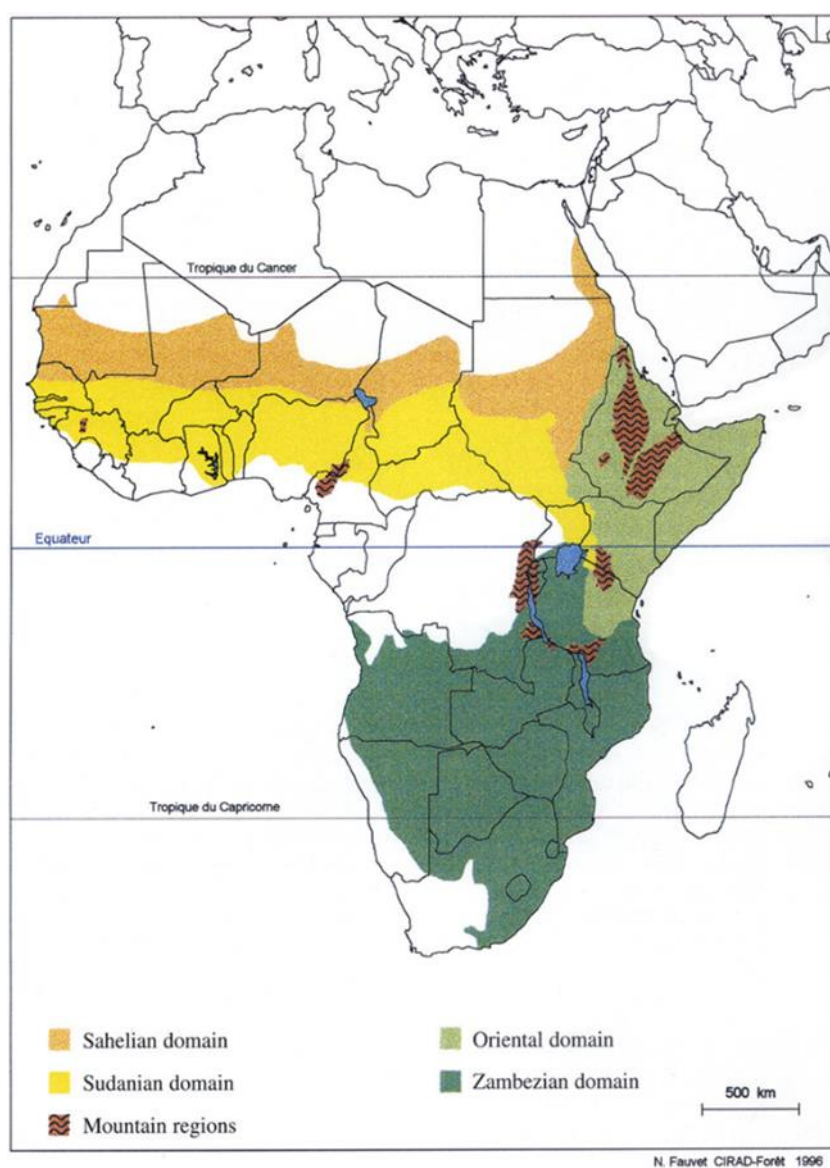


Figure 2.1: The extent of the Zambezan phytoregion (domain) in relation to other major phytoregions in sub-Saharan (N. Fauvet CIRAD-Forêt 1996).

The dominance of species from these genera distinguishes the miombo from other African Savanna woodlands and provide a basis for beekeeping. This form of land use is highly significant for cultural, social and economic purposes in miombo woodlands (Campbell *et al.*, 2007). Grass enables fire to be a constant feature in miombo woodlands, which is frequent and widespread (Campbell, 1996; Chidumayo, 1997; Mendelsohn & el Obeid, 2005; Timberlake & Chidumayo, 2011). Additional features distinguishing the miombo woodlands are the unimodal rainfall pattern, prolonged dry seasons, coupled with the generally leached and impoverished soils. The above environmental factors regulate the extent of the miombo woodlands in southern Africa (Timberlake & Chidumayo, 2011).

2.2.1. Forest types of Namibia

Giess (1998) broadly classified the vegetation of Namibia into three main zones of deserts, savannas and woodlands. The forest in Namibia is found on the aeolian Kalahari sands in the north-central and northeastern parts, described as dry, semi-open woodlands (Giess, 1998; Mendelsohn & el Obeid, 2005; Ministry of Agriculture, Water and Forestry, 2011). The areas with forests include eastern Zambezi region, Kavango East and West, eastern Ohangwena and in the hills around Tsumeb, Otavi and Grootfontein (Mendelsohn & el Obeid, 2005). More open woodlands are located in the west and south of the forested area, while shrubland and desert mainly cover the western and southern parts of the country (Figure 2.2) (Giess, 1998; Mendelsohn & el Obeid, 2005). Thus, 20% of Namibia's surface area is covered by a contiguous woodland and 66% of this is open *Acacia-Combretum* woodland representing about 830,000 km² (Ministry of Agriculture, Water and Forestry, 2011).

Mendelsohn and el Obeid (2005), asserted that woodland is a much broader term encompassing forests and landscapes where reasonable tall trees are conspicuous, while in shrublands trees are few and scattered. They defined savannas as woodlands that are distinguished by a prominent grass layer under a stratum of widely spaced trees often giving an appearance of an open parkland. Important environmental factors such as aridity, soils and fire, operating alone and in combination, determine the structure and composition of plants in any given place. Consequently, these factors affect the mix of grasses, shrubs and trees resulting into an area being a forest, woodland, shrubland or desert. The subfamily Caesalpinioideae dominates in the Kalahari sands woodlands for typical species ecosystems of *B. plurijuga*, *B. africana*, *G. coleosperma* and *C. mopane*. Other important species found in the woodland include *P. angolensis*, *Sclerocarya birrea*, *T. sericea* and *S. rautanenii* (Burke, 2006; Chidumayo, 1997; Ministry of Agriculture, Water and Forestry, 2011). The forest sector in Namibia is managed

by the Directorate of Forestry (DoF), who works in partnership with forestry stakeholders comprising forestry resource users (communities, farmers, private sector) and institutions that provide financial resources (FAO, 2010). According to FAO (2010), the DoF has two main divisions; the forestry management division and forestry research division. The forestry management division has the responsibility of policy and legal framework development, protection and management of classified forests, promotion of farm community and environmental forestry, providing extension services, and maintaining an efficient sector-wide management information system. The responsibility to conduct integrated forestry research encompassing indigenous knowledge and dissemination of forestry research information is vested with the forestry research division.

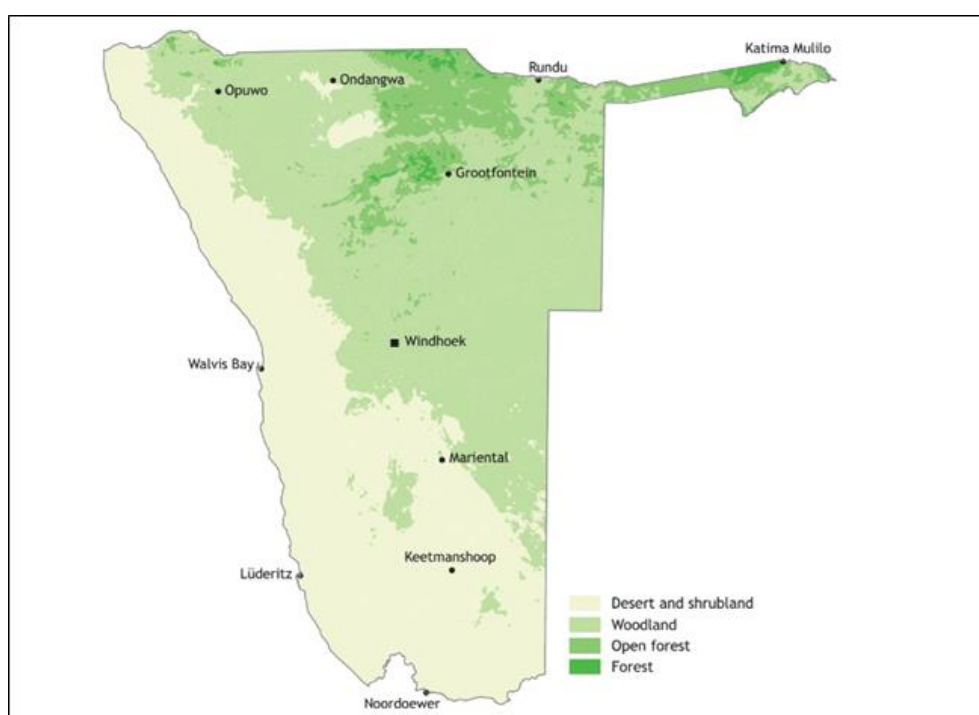


Figure 2.2: Approximate distribution of forests and woodlands in Namibia. Land cover of 10% is under forests, 50% under woodlands and 40% under shrub-lands and deserts (Mendelsohn and el Obeid, 2005).

In the 2011 population and housing census, the population of Namibia stood at 2.113 million people (National Statistics Agency, 2011). At the average annual growth rate of 1.9%, the population of Namibia is projected to be at 2.542 million at the end of 2017, out of which 1.124 million of the population are projected to live in woodland areas directly or indirectly depending on forest resources (National Statistics Agency, 2006).

2.2.2. Forest types of Zambia

Miombo woodlands are the most widespread vegetation type in the Zambebian phytoregion extending from Angola, the southern Democratic Republic of Congo (DRC), Malawi, Mozambique, Tanzania, Zambia and Zimbabwe (Campbell, 1996; Chidumayo, 1997; Dewees *et al.*, 2011; Ministry of Agriculture, Water and Forestry, 2011; White, 1983). The miombo woodlands cover 70% of the Zambebian phytoregion (Chidumayo, 1997), which is characterised by *Caesalpinioideae* leguminous tree species in the subfamily *Caesalpinioideae* as earlier pointed out by various authors under Section 2.1.

Sixty six percent of Zambia's land mass is covered by evergreen, semi-evergreen and deciduous forests (Mukosha & Siampale, 2009). Further, it is stated that the largest coverage is by the semi-evergreen forests, dominated by miombo woodland at 45.4% of the total land cover. The deciduous forests are second at 19.8% comprising Kalahari mopane, munga woodland and *Baikiaea* forests whereas the evergreen forests stand at 1.1% of the total land cover. White (1983) classified the miombo woodlands broadly into dry type (< 1 000 mm) and wet type (> 1 000 mm). In addition, (Chidumayo, 1987a), further divided the miombo into subtypes as follows:

- Northern wet miombo – this miombo type predominantly consists of *Brachystegia spiciformis* and *Brachystegia utilis* woodlands having *Julbernardia paniculata* and *Parinari curatellifolia* in co-dominant canopy. The understory taxa belong to *Monotes africanus*, *Syzygium guineense macrocarpum* and *Uapaca* species. These woodlands are mostly found in the Northern, Luapula and parts of the Central Provinces of the country.
- Northwestern wet miombo – this miombo type consist of *Brachystegia* (*B. spiciformis*, *B. longifolia*) woodlands in which *Isoberlinia angolensis* and *J. paniculata* form common canopy co-dominants. *Anisophyllea boehmii*, *Diplorhynchus condylocarpon*, *S. guineense macrocarpum* and *Uapaca species* are found in the understory canopy. The woodland is found in Northwestern province, northern parts of Western province and the Copperbelt province.
- Central dry miombo - consists of *Brachystegia* (*B. boehmii*, *B. spiciformis*, and *B. utilis*) woodland with *Julbernardia globiflora* as a common canopy co-dominant. The understory comprise *D. condylocarpon*, *Lannea* species, and *Pseudolachnostylis maprounefolia*. This type of woodland is found in Central province and parts of Southern province.

- Eastern dry miombo - consists of *Brachystegia manga* – *Julbernardia* woodland. The understory species are *D. condylocarpon*, and *P. maprounefolia*. This type is found in the Eastern province and the parts of the Central and Northern provinces bordering the Eastern province.
- Western dry miombo – consists of the *B. spiciformis* – *J. paniculata* having *B. africana* as a common canopy co-dominant, with *D. condylocarpon* as a common understory species. This type of woodland is found in Western Province and parts of Southern province. The woodland is referred to as Kalahari miombo (White, 1983) or Kalahari sand miombo (Fanshawe, 2010).

Based on the FAO land use/forest type classification for integrated land use assessment for Zambia, Mukosha & Siampale (2009) summarised the forest types as indicated in Table 2.1.

Table 2.1: Summary of the Forest types found in Zambia as defined by the FAO, (Mukosha and Siampale, 2009).

Land uses	Brief description	Code
Forest	Area \geq 0.5 ha, tree crown cover	
Forest with natural or natural assisted regeneration		
Evergreen Forest	Includes: <ul style="list-style-type: none"> • Mavunda forests (<i>Cryptosepalum exfoliatum</i>) • Mufinsa (<i>Syzygium guineense</i> subspecies <i>afromontanum</i>) • Mofu (<i>Entandrophragma devevovi</i>) • <i>Parinari</i> species, <i>Syzygium</i> species • Riverine/riparian forests 	EF
Semi-evergreen forest	Includes: <ul style="list-style-type: none"> • Miombo woodland (<i>Brachystegia</i>, <i>Isoberlinia</i>, <i>Julbernardia</i> and <i>Marquesia macropura</i> species) 	SEF
Deciduous forest	Includes: <ul style="list-style-type: none"> • <i>Baikiaea</i> forests (<i>Baikiaea plurijuga</i>) • Kalahari woodland (<i>Baikiaea</i>, <i>Brachystegia</i>, <i>Isoberlinia</i>, <i>Julbernardia</i> and <i>Schinziophyton</i> species) • Mopane woodland (<i>Colophospermum mopane</i>) • Munga woodland (<i>Acacia</i>, <i>Combretum</i> and <i>Terminalia</i> species) 	DF
Other		OF
Forest Plantations		
Broadleaved forest plantations		FPB
Coniferous forest plantations		FPC

Forest sector management in Zambia is the responsibility of Forest Department, a government agency with a mandate to administer the forest estate (Mackenzie, 2014). Mackenzie (2014) citing Forest Department (2009) listed functions of the Forest Department to include among others reforestation of degraded areas, sustainable management and utilisation of forest

resources, undertaking smooth, efficient and operations, formulating forest policy and legal framework, protection and management of forest estates, carrying out forest surveys and demarcations, conduct research, extension and publicity. The Forest Department is divided into four divisions, namely administration and human development, forest management, forest research and forest extension (Mackenzie, 2014).

In the 2010 population and housing census, Zambia's population was 13.719 million people (Central Statistical Office, 2011). With a projected 2.8% average annual growth rate, the midterm population in 2017 is projected to be at 16.405 million people, with 57.5% rural based population which is directly and indirectly dependent on forest resources (Central Statistical Office, 2013).

2.3. Natural regeneration

Natural regeneration is broadly defined as the renewal of the forest by self-sown seed, by coppice of root suckers or layering seeds deposited by wind, birds or mammals without the help of humans (Chidumayo, 1997; Pardos *et al.*, 2005). These processes can either fall into sexual or vegetative (asexual) categories (Chidumayo & Gumbo, 2010). In short, it is a process by which a forest naturally renew itself through a recruitment of young plants (Borghetti & Giannini, 2005).

Regeneration processes in natural forests are stimulated by open patches created by disturbances, and by death or felling of mature trees (Borghetti & Giannini, 2005). These authors further state that the success of natural regeneration is dependent on sufficient amount of growing space. Mature trees highly regulate the understory light regime and the tree canopy tend to reduce the growing space for the recruitment of young trees into the canopy layer.

The changes in species composition following a disturbance are derived from the different ways in which specific species withstand the disturbance (Kennard *et al.*, 2002). The authors add that mode of regeneration dictates the survival and influences the growth of a species after a disturbance. In many cases forest disturbances arise from wild fires, cutting/harvesting or browsing by wild or domestic animals (Kayofa, 2015). At optimum, natural regeneration of miombo tree species, from seeds or coppicing and root suckering, is well attained when the occurrence of fire and grazing are sufficiently limited for certain periods of time (Mwabumba *et al.*, 1999). Subsequently, survival and growth of a given species after a disturbance is dictated by the mode of regeneration and the rates differ from one species to the other (Kennard *et al.*, 2002).

2.3.1. Importance of natural regeneration

According to Borghetti and Giannini (2005), in most cases when natural regeneration is compared to artificial regeneration it proves to be superior because seedlings which are locally dispersed from seeds adapt well to locally harsh environmental conditions. Secondly, the processes are reliable to restore large areas after a disturbance, and lastly it is a valuable means in bringing about forest types that are efficient for providing important forest products and guarantying multipurpose forest functions. Through natural regeneration, the forest is able to recover very fast and promotes species richness depending on the suitability of the environmental factors (Chen *et al.*, 2014). However, the authors also argues that for timber production, paper industry and construction activities, artificial regeneration maybe be found suitable depending on the local climate and soil conditions, as it is faster for selective restoration of the forest.

2.3.2. Sexual regeneration

Sexual regeneration is when seedlings establish themselves from germinating seeds leading to recruitment of small trees into tree phase (Chidumayo & Gumbo, 2010). Sexual regeneration is considered the primary forest regeneration mechanism (Deiller *et al.*, 2003). It has an important role in the establishment of trees exposed to frequent fires even when vegetative regeneration is more advantageous by expectation (Munkert, 2009).

Generally, seedlings take long to establish for species that undergo a suffrutex stage, commonly known as shoot dieback, and even those that do not (Campbell, 1996). The author further argues that in the first year of seedling establishment, most growth takes place belowground as little happens aboveground. These activities take place mostly during the rainy season. However, shoot dieback often does not result in seedling mortality, as roots are capable of producing a shoot the following season. Fire frequency is a major obstacle for sexual reproduction in most tree species. To overcome this obstacle, seedlings must establish and grow to fire tolerant size fast enough within the short period between fire intervals to guarantee a successful reproduction regimen (Hoffmann, 1998). Depending on the phenology of the species, fire destroys dispersed seeds, developing fruit or flower buds and the author concluded from the study, conducted in the tropical rain forest of Brazil, that fire increases mortality significantly.

2.3.3. Asexual regeneration

Asexual regeneration is a vegetative regeneration resulting from coppicing of existing trees that were cut or damaged leading to recruitment of sprouts or resprouts into the tree phase.

Production of secondary trunks can also be induced by profound changes in growing conditions such as severe injury caused by herbivory, fire, floods, logging or drought. (Chidumayo & Gumbo, 2010). They classified types of sprouts as being trunk sprouts, specialised underground stems (lignotubers and rhizomes) and root sprouts (root suckers).

Many tree species have the ability to resprout, best utilising vegetative and sexual reproduction, enabling them to dominate in many ecosystems (Bond & Midgley, 2001). The majority of the tree species in the miombo woodland persist because of the ability to coppice after a disturbance (Luoga *et al.*, 2004), for which coppicing ability is influenced by the species, stump height and diameter (Handavu *et al.*, 2011). It is further argued that sprouting ability for individuals after severe injury from disturbance enable each individual to produce vigorous shoots than seedlings because sprouts are supported by extensive root system and food storage from the parent individuals (Luoga *et al.*, 2004; Busby *et al.*, 2010; Chidumayo & Gumbo, 2010). Further, Pardos *et al.* (2005) stated that sprouts are able to resist the stress caused by lack of water or nutrients because of the support from a well-established root system. According to Bond and Midgley (2001), sprouts are in a better position to fill the gap left by a mature tree than seedlings when it is blown over. This is because sprouts are able to grow faster than seedlings and quickly fill the gaps. However, recruitment opportunities reduce in both time and space by trees with long established persistence. Hence, forest diversity is reduced, since resprouts tend to be multi-stemmed in most instances (Kruger *et al.*, 1997). Kruger *et al.* (1997) postulated that resprouters are rare in tall forests and produce few seedlings due to allocation of resources to areas of vegetative reproduction. Since resprouters divert resources to developing, maintaining and protection of bud-banks, they are likely to be disadvantaged in resource allocation and structural reasons. In addition, resprouters are unable to maximise vertical growth as resources are shared amongst several stems, poorly supporting each stem because of being multi-stemmed. Multi-stem species dominate in short forests and thickets, and noticeably seedlings are absent (Kruger *et al.*, 1997). It has been concluded from various studies that resprouting ability tend to differ from one species to the other and decline with the age, size, frequency and severity of disturbance (Chidumayo & Gumbo, 2010; Kennard *et al.*, 2002).

2.4. Forest fire

Various authors have pointed out that fire is an important natural phenomenon affecting savanna systems and its use date way back to early Stone Age (Chidumayo, 1997; Nepolo & Mapaire, 2012). The authors also stated that occurrence of fire in forests is determined by environmental conditions (climate), source of ignition and sufficient fuel (flammable biomass)

and these constitute three factors required for fire to burn (Bond & Van Wilgen, 1996; Chidumayo, 1997). Fire is ranked third on the scale of global disturbances consuming huge quantities of plant biomass nearly in all forests types of the world such as boreal forests, tropical grasslands and the savanna (Bond, 2001; Bond & Van Wilgen, 1996). For many millions of years, fire has been a feature on the African landscape occurring more wide and frequently compared to any other continent (Mendelsohn & el Obeid, 2005). 20 - 40 million hectares of tropical forests including open forests, tropical and sub-tropical savannas were estimated as burnt annually (Goldammer *et al.*, 2002). The miombo region has a climate with a long dry season from April to November while high temperatures and lowest humidity occur during August to November, creating a hot-dry season, which is the period of high fire risk (Chidumayo, 1997).

The majority of wild fires are man-made resulting from the use of fire as a tool for various purposes (Bond & Keeley, 2005; Chidumayo, 1997; Geldenhuys, 1977; Goldammer *et al.*, 2002; Kikula, 1986; Mendelsohn & el Obeid, 2005; Ryan & Williams, 2011; Trapnell, 1959). Among others this include using fire to prepare agricultural fields (new and old), honey collection, triggering new grass for livestock, hunting, bush and undergrowth clearing to improve visibility around settlements, maintenance of foot-paths and roads, and scaring away dangerous animals such as snakes and carnivores (Chidumayo, 1997; Goldammer *et al.*, 2002; Kikula, 1986; Mendelsohn & el Obeid, 2005). It is from such human activities that wild fires ensue, contributing to the shrinking of evergreen forests to increase grasslands and savannas (Bond, 2001; Bond & Van Wilgen, 1996).

2.4.1. Fire regime

Fire regime can be described by its frequency (how often it occurs), intensity (how fiercely it burns) and season (when it occurs) and these factors determine fire variation (Bond, 2001; Bond & Van Wilgen, 1996). It is now proven that fire intensity is largely determined by fuel load and most fuel that is easily ignitable in savanna is made up of standing dead grass and litter (Gambiza *et al.*, 2000).

Fire occurrence is known to be in three forms. The first type is ground fire, which burns underground in organic layers of the soil, second type is surface fires burning just above the ground surface, and the third type is crown fire which burn in the canopies killing or severely damaging crowns, boles and whole trees (Bond & Van Wilgen, 1996; Kozłowski & Ahlgren, 1974). According to Bond & van Wilgen (1996), in most cases crown fire is supported by surface fire. However, there are instances when crown fire burns independent of the surface

fire. These authors further pointed out that fire can burn with the wind upslope that is called head fires and fire can equally burn against the wind downslope called back fires. Furthermore, it was deduced that different ecosystems have different conditions resulting into distinct fire regimes.

As clearly stated by Bond & van Wilgen (1996), the fire regime in African savanna has a fire frequency generally ranging from annual to every 30 years or more in different places. It is dependent on rainfall and grazing pressure, season of fire during dry seasons when grasses are dormant. Fire intensities could range from < 100 to $> 5,000 \text{ kWm}^{-1}$ depending on the combination of the mentioned factors. In any particular region, fire regime is determined by grass productivity and it is a function of rainfall (Mapaure, 2013), agreeing with the earlier authors on fire regime.

2.4.2. Fire return interval

Fire return intervals varies from place to place and may be measured in decades or even centuries depending on the fire history of the place (Bond, 2001; Bond & Van Wilgen, 1996). Heinselman (1973), as cited by Bond & van Wilgen (1996), is known to have initiated the development of a method to determine fire return intervals. This was done by mapping past fires as careful as possible, referring to written records, assessing fire scars and stand origins. Bond & van Wilgen (1996) stated that the work of Heinselman (1973) and other studies conducted elsewhere contributed to the development of Stochastic models which describes fire history on site, citing Johnson (1978), Johnson & van Wagner (1984) and van Wagner (1978). These authors concluded that fire maps and written records of fire have reliably been used in reconstructing and analysing short-term disturbance history. As cited in Bond & van Wilgen (1996), Whitney (1986) reconstructed a fire disturbance regime for pre-settlement pine forests of Michigan using 150-year old survey records. Whitney showed that the nature and flammability of the vegetation controlled fire and also that there was a close association of forest type and fire frequency. Therefore, Bond & van Wilgen (1996) concluded that for one to clearly understand the current vegetation composition, one has to understand the fire history of the site and not just the post-fire age. Therefore, fire return interval for any particular area can only be derived in the context of fire occurrence history in a given period.

2.4.3. Fire effects on tree species

The effect of fire on plants is dependent on time when the fire occurs, its fire frequency and flammable biomass (Chidumayo, 1997). Chidumayo went on to state that mature plants are

more susceptible to fire damage than buried seeds in the soil. Dormant plants are also less susceptible to fire damage as compared to actively growing or reproducing plants. Chidumayo's argument concurred with the findings stating that plant growth, survival and reproduction are directly affected by fire which in turn can impact seeds and seedling dynamics (Bond & van Wilgen, 1996). It is further stated that fire alone or with another agent regularly kills mature plants, making it very important in structuring communities as a natural disturbance. According to Mapaure (2013), fire has the largest impact on woody vegetation in the lower vegetation strata such as shrubs, saplings and small trees. The author further stated that plant mortality especially of seedlings, shrubs and saplings, whose fire tolerance levels may generally be low, result from severe or more frequent fire.

Fire effects on plant reproduction are complex with many species being stimulated for enhanced flowering, seed germination and seedling recruitment as others are obligate dependent on fire for reproduction and may die if fire is suppressed (Bond, 2001; Bond & Van Wilgen, 1996). Additionally, plant species adapt to respective fire frequency, season or intensity. As such, fire survival of plants is largely influenced by thick bark, crown architecture and resprouting from bark protected buds or other subterranean positions. Chidumayo (1997) adds that fire tolerance by most tree species is achieved through having active phenological phases during low fire risk, insulating vital tissues from high fire temperatures and the ability to recover vegetatively from fire damage. A lower level of damage is attained in wetter or more wooded landscapes compared to open and drier landscapes (Smit *et al.*, 2010). Bond & van Wilgen (1996) asserted that species ability to survive a fire is dependent on the nature of fire and varies among species and within size classes. These authors noted fire regime variation to have an influence on species co-existence and if successive fires differ, different species may appear on the site.

Gambiza *et al.* (2000) carried out a model experiment where it was discovered that frequent fires affected negatively on woodlands through the reduction of woody basal area and densities of harvestable trees. On the opposite extreme, very infrequent fire supported promotion of bush encroachment of smaller woody plants that inhibit the transition of harvestable trees. They concluded that complete fire protection and long fire-free periods promote woody plant development, unlike frequent fires, which has the opposite effect. These findings concurred with the earlier works (Geldenhuys, 1977; Trapnell, 1959) and were later on supported by the research in other areas (Bond, 2001; Mendelsohn & el Obeid, 2005; Sheuyange *et al.*, 2005; Govender *et al.*, 2006; Smit *et al.*, 2010; Ryan & Williams, 2011; Nepolo & Mapaure, 2012; Verma & Jayakumar, 2015). Fire has less influence on species with the most effective

vegetation fire-survival characteristics while species without such characteristics, is subject to mortality, thus regulating population growth and community composition (Bond, 2001; Bond & Van Wilgen, 1996). Fire is a natural selection force that determine the development of savanna species by synchronising flowering, seed germination and replacing of former species by opportunistic ones leading to the altering of plant composition and community; especially in understorey, small trees and shrub layers (Trapnell, 1959; Lavorel *et al.*, 1999).

Fires are a fundamental part of most forests largely impacting on ecosystems and soils (Fisher & Binkley, 2000; Reinhart *et al.*, 2016). In the woodland burning experiment, Trapnell *et al.*, (1976) assessed the effect of fire and termites on the Zambian woodland soil. The soil comprised 87% sand, 4.8% silt and 6.8 % clay in the top 15 cm layer. It also had very low organic content and extremely low base saturation. Trapnell *et al.* (1976), concluded that fire has positive effects on soil nutrients status, especially in the top 15 cm soil layer, because soil pH and exchangeable phosphorus, calcium and magnesium improved due to long term burning effect in comparison with complete protection from fire. In other observations, fire tend to increase nutrient availability in soil top layer (Fisher & Binkley, 2000; Singh *et al.*, 2017). An increase in soil nutrient availability to plants after a fire results from rapid mineralisation of nutrients, reduced competition among plants, and to some extent the sustained changes in soil conditions in form of temperature and water content (Fisher & Binkley, 2000). Significant effects of fire on soil available nutrients occur a month after a fire and progressively weakening over a course of a year to the state of an unburned in a few years (Singh *et al.*, 2017). The study of post-regeneration low fire frequency in Spain concluded that species richness increases in the first years after fire due to the availability of good supply of nutrients and light (Ohl, 2005). Thus, the study agreed with other authors on fire's positive effect on soil nutrient availability in the short term. However, Mendelsohn and el Obeid (2005) asserted the contrary, saying that repeated annual fires accelerate the rate of soil nutrient loss from the soil thus reducing soil fertility. Soil nutrient loss takes place in two ways, according to Mendelsohn and el Obeid (2005), firstly, fire cause carbon, nitrogen and sulphur to become volatile molecules and are then lost to the atmosphere. Secondly, roots tap nutrients from below surface, translocating them to leaves, twigs and wood, which are later burnt after they fall to the ground as dead material thus, increasing the rate of nutrient loss through burning. Overall observations on soil nutrient availability after fires indicate that there is generally a depletion of nutrients over time from the ecosystem as a whole (Ben du Toit, pers. comm.).

Generally, some species are fire tolerant, others are fire sensitive or medium tolerant (sensu Trapnell, 1959; Geldenhuys, 1977). Among the fire tolerant species are *P. angolensis*, *Boscia albitrunca*, *B. africana*, *A. boehmii*, *D. condylocarpon*, *S. Innocua*, *U. nitida*, *P. polyandra*, *P. curatellifolia*, *E. africanum*, *S. madagascariensis*, *S. cocculoides*, *S. spinosa*, *T. sericea* and *Xylopia odoratissima*. Medium fire tolerant species include among others *B. bequaertii*, *P. maprouneifolia*, *S. pungens*, and *U. kirkiana* whereas the fire sensitive species include among others *Brachystegia species*, *Julbernardia species*, *Isoberlinia species*, *A. erioloba*, *C. mopane*, *B. plurijuga*, *O. pulchra* and *Commiphora species* (Burke, 2006; Geldenhuys, 1977; Trapnell, 1959).

2.5. Selected important tree species

2.5.1. Importance of selected tree species

Natural resources, among others dry forests and woodlands (Campbell, 1996; Chidumayo & Gumbo, 2010; Mendelsohn & el Obeid, 2005) have several types of economic values attached to them, “commonly classified under total economic value” (Barnes *et al.*, 2010). The authors classified the values in three categories of direct use, indirect use and non-use values; giving example of harvesting for timber, fuel wood, poles, non-timber forest products (NTFPs) as some of the direct use.

Forest and woodland resources, despite being limited in Namibia, provide valuable ecological, social and economic resources that are important among the many livelihoods in rural communities (Kabajani, 2016). Generally, Campbell (1996) states that the woodlands are highly significant in southern, central and eastern Africa. Major rivers, notably the Zambezi, Luangwa, Limpopo, Save, Cuando, Kavango, Rufiji, Rovuma and Luapula (part of the Upper Congo), have their sources in the areas covered by the woodlands. Subsequently, the activities in the woodlands control the quality and quantity of water in the region (Campbell, 1996; Timberlake & Chidumayo, 2011).

Tree species are valued according to utilisation, magnitude of use, products and services including the intended markets such as local or continental (Campbell, 1996; Kabajani, 2016; Mendelsohn & el Obeid, 2005). Species use is very selective depending on the wood products either being industrial wood for international and domestic markets or non-industrial wood meant for local distribution in local markets (Chidumayo & Gumbo, 2010). Thus, selection criteria are mainly on the height, diameter and density of species taking into account access to the forest and woodlands not far from paths and roads, resistance of species from insect attack

and competing alternative use. Most preferred high timber species include among others *B. plurijuga*, *P. angolensis*, *G. coleosperma*, *B. africana* and *Afzelia quanzensis* (Maliondo *et al.*, 2005; Mendelsohn & el Obeid, 2005; Barnes *et al.*, 2010; Timberlake & Chidumayo, 2011; Mackenzie, 2014). Other species such as *Brachystegia* species, *Terminalia* species, *Acacia* species, *Pericorpsis angolensis* and *C. mopane* are widely used in construction timber or fuelwood (Timberlake & Chidumayo, 2011). According to Chidumayo and Gumbo (2010), local use (non-industrial) has less impact on forest degradation since there is a wide spectrum to choose from for local purposes. However, there is very little evidence to show that the current levels of timber exploitation are sustainable (Chidumayo & Gumbo, 2010). More plant foods are also gathered from domesticated miombo fruit species of the genera *Sclerocarya*, *Strychnos*, *Adansonia*, and *Berchemia*, and from plants growing on scarce, clay-rich soils in the genera that include *Carissa*, *Ficus*, *Garcinia*, *Pappea*, *Syzygium* and *Parinari* (Deweese *et al.*, 2011). The authors pointed out two major fruit producing species of significant economic importance in miombo woodlands, namely *U. kirkiana* and *S. rautanenii*, occur naturally in large stands.

2.5.2. *Baikiaea plurijuga*, Harms

B. plurijuga in general is a medium-sized to large tree, with a normal height range of 8 - 16 m (Coates-Palgrave, 2002). However, others have recorded tree heights of 20 – 27 m and up to 120 cm diameter at breast height (Fanshawe, 2010; Ngoma *et al.*, 2015). *B. plurijuga* in different areas is referred to by various names such as Zambezi teak, Rhodesian teak, Mukusi or Zambezi kiaat (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Ngoma *et al.*, 2015). Coates-Palgrave (2002) describes the crown of the tree as being large, dense and spreading. Studies of the species have established that it is less tolerant to fire and sensitive to water availability (Burke, 2006; Ngoma *et al.*, 2015).

The occurrence and distribution of the species is limited to soils characteristic of deep Kalahari sands in open deciduous woodlands (Coates-Palgrave, 2002), with environmental parameters that conform to more than 300 mm mean annual precipitation and less than 1 700 mm water deficit. Other parameters include less than 10 frost days, greater than 4°C annual minimum temperature and an average temperature of greater than 20°C (Burke, 2006). The range of distribution is mainly in the open *Baikiaea* forests (De Cauwer *et al.*, 2016) especially south-eastern Angola, northern Botswana, northeastern Namibia, southwestern Zambia and western Zimbabwe (Mendelsohn & el Obeid, 2005; Ngoma *et al.*, 2015). The distribution of the species in Namibia is largely found in the far northern and north eastern part of the country (Curtis &

Mannheimer, 2005; Mendelsohn & el Obeid, 2005), whereas in Zambia the species distribution is in northwestern and western parts of the country (Ngoma *et al.*, 2015).

The species is regarded as important as it is a source of hardwood timber for local, national and international markets (Barnes *et al.*, 2010; Ngoma *et al.*, 2015). This is because its wood has characteristics of even texture, hardness, strength, durability and beautiful dark-brown colour. Subsequently, the uses range from furniture, flooring, mining props, canoes and rail sleepers (Coates-Palgrave, 2002; Mendelsohn & el Obeid, 2005). Further, the wood is able to withstand abrasion without splintering.

2.5.3. *Burkea africana*, Hook

B. africana is a medium leguminous tree of 8 – 12 m height, occasionally reaching 20 m (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Mendelsohn & el Obeid, 2005; Wilson & Witkowski, 2003). The species is also known as red syringa or wild syringa (Mendelsohn & el Obeid, 2005; Wilson & Witkowski, 2003). The species has a dark grey bark, which is rough and flaking. Leaves are bipinnate, which crowd at the end of branchlets; flowers are creamy white occurring between September and November, which appear just before the new leaves. Further, the species can be confused with *Albizia antunesiana* and *E. africanum* during the season when the species lose leaves. However, in both cases, the stubby, velvety, rusty red to maroon tips of the branchlets of *B. africana* set it apart from the other two species (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005). The species is further distinguished by other qualities of having a rooting depth of 15 – 60 cm and passive seed dispersal, including having relative high fire tolerance (Mendelsohn & el Obeid, 2005; Wilson & Witkowski, 2003).

The species grow in various woodlands that can be found over a range of altitudes and habitats (Coates-Palgrave, 2002). The distribution range is generally on sandy soils up to 1 500 m a.s.l. throughout Africa, extending from Ethiopia in the north and Nigeria in the west (Wilson & Witkowski, 2003) to southeastern South Africa (Mendelsohn & el Obeid, 2005). The habitats, according to Burke (2006), should meet certain environmental parameters such as less than 30 days of frost, greater than 19°C average temperature, more than 300 mm annual mean rainfall and less than 1 900 mm water deficit.

The wood from the species has qualities such as being hard, heavy and tough. Colour has a variety of pale yellow to reddish brown (Coates-Palgrave, 2002). In the past, the primary use of the species was for commercial harvest of saw timber, which is not currently being done in Namibia (Barnes *et al.*, 2010). General uses of the species by the local communities include

tanning with bark and pods, treatment of septic sore by way of using chewed bark and applying it as poultice on the sore, fuelwood, construction of kraal fences, stools as well as mortars and pestles (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Mendelsohn & el Obeid, 2005).

2.5.4. Guibourtia coleosperma, (Benth.) J. Léonard

G. coleosperma (large false Mopane) is an evergreen tree species belonging to Fabaceae family; of the subfamily Caesalpinioideae (Coates-Palgrave, 2002). The species in general attain 6 – 20 m height with a large, rounded and drooping canopy. The bark is smooth, cream to pinkish cream colour with patches that are often dark brown to black flakes. Other distinct features of the species are leaves that form two leaflets, which are curved, ovate and shaped like a sickle (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005).

The species is known to occur in open woodlands and dry forests (Coates-Palgrave, 2002), broad-leaved deciduous woodlands (Scholes *et al.*, 2002) or wooded bushlands (Sheuyange *et al.*, 2005) that are mostly Kalahari sandy soils (Barnes *et al.*, 2010; Burke, 2006; Coates-Palgrave, 2002; Mendelsohn & el Obeid, 2005; Scholes *et al.*, 2002; Sheuyange *et al.*, 2005). Burke (2006) outlines suitable conditions to include among others less than five frost days, average temperatures that is greater than 21°C, mean annual precipitation of greater than 450 mm and less than 1 500 mm water deficit. In addition, the species has low fire tolerance and mainly its seed dispersal is by way of animals. The species is known to grow in close association with *B. plurijuga*, *B. africana* and *P. angolensis* (Mendelsohn & el Obeid, 2005; Ncube & Mufandaedza, 2013).

The species is commercially harvested for export timber in Zambia (IRDNC, 2015; Mackenzie, 2014) and to a lesser extent in Zimbabwe where it is selectively done at rotation age of 50 – 100 years for various purposes including domestic and commercial products (Ncube & Mufandaedza, 2013). In Namibia, harvesting for saw timber took place previously but there are no more extraction of the species (Barnes *et al.*, 2010). Other notable use of the species are construction, furniture, ornamentals, dug-out canoes and seeds as source of food and seed oil (Curtis & Mannheimer, 2005; Mendelsohn & el Obeid, 2005). Consumption of its seeds especially the arils in form of processed drink and pounded (roasted seeds) meal are known in Botswana, Zambia (Coates-Palgrave, 2002) and Namibia (Mendelsohn & el Obeid, 2005).

2.5.5. Pterocarpus angolensis, DC

P. angolensis is a leguminous tree species in the Fabaceae family, subfamily Papilionoideae (Vermeulen, 1990; Coates-Palgrave, 2002; Therrell *et al.*, 2007; De Cauwer *et al.*, 2014). The

species has many localised names in respective areas where it is found and include among others African teak, Bastergreinhout, Bloodwood, Dolfhout, Girassonde, Kiaat, Mninga, Mukwa, Transvaal teak, Umbila, (Vermeulen, 1990; Coates-Palgrave, 2002; Mendelsohn & el Obeid, 2005; De Cauwer *et al.*, 2014; De Cauwer *et al.*, 2017).

The species is a medium to a large sized deciduous tree reaching 10 – 20 m in height, with a fairly cylindrical trunk and an open flat-topped and spreading crown (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Graz, 2004a; Therrell *et al.*, 2007). Tree individuals with height of up to 25 m occur in Mozambique, Tanzania and Zambia (Vermeulen, 1990). Other descriptive features of the species include a dark grey to brown bark that is rough and fissured resembling that of crocodile skin. Leaves occur in 5 – 9 pairs of sub-opposite to alternate leaflets. Flowers are pea shaped, orange in colour whereas fruits are very distinctive found in circular pods occurring between January and April (Coates-Palgrave, 2002). Flowering and leaf flush in the species occur during the months of August to December (Stahle *et al.*, 1999). Phenology is well synchronised with the beginning of rainy season, closely followed by flowering with the beginning of the rains (Kayofa, 2015). The species experiences shoot dieback (suffrutex) to the depth of 2 – 36 cm below surface during dry season as a measure to protect meristem from fire damage (Van Holsbeeck *et al.*, 2016). During the suffrutex stage, the plant is able to produce shoots every growing season that may reach between 1 and 3 metres (Graz, 2004a). Further, the suffrutex stage can last for a decade and may extend to 25 years if the frequencies of fire occurrence are high (Graz, 2004a; Van Holsbeeck *et al.*, 2016). Graz (2004a) points out that the species has high resistance to fire but only after it has reached the sapling stage.

The species occurs in woodlands and wooded grassland (Coates-Palgrave, 2002; Fichtler *et al.*, 2004). The species grows best in open dry forest where it faces less competition and no other factors that can limit its growth such as high temperature seasonality (De Cauwer *et al.*, 2017). The species' occurrence is closely associated with open and closed savannas, often as dominant canopy species (Shackleton, 2002). Shackleton (2002) states that the species is widely distributed throughout Africa south of the equator, whereas De Cauwer *et al.* (2017) narrow the distribution to most tropical Southern Africa, others narrowing it to central and southern Africa with limits being northern South Africa and northern Tanzania as the southern and northern limits respectively (Thunström, 2012). However, other authors point to the fact that the range of altitude on which the species occurs varies as low as the sea level on the Mozambican coast up to 1 650 m a.s.l in parts of Tanzania (Stahle *et al.*, 1999). The distribution range of the

species include growing on a wide variety of soil formations that are deep well-drained in Angola, DRC, Malawi, Namibia, Tanzania, South Africa, Zambia and Zimbabwe (Vermeulen, 1990). The ideal conditions that fit the growing environment for this species, according to Burke (2006), are arenosols (sandy soils), less than 10 frost days per year, 20 °C average temperature, greater than 400 mm mean annual rainfall, and less than 1 700 mm water deficit. The conditions are within the mean annual rainfall and mean temperature ranges presented in the earlier work of Stahle *et al.* (1999) and Mwitwa (2004). The growing of the species is closely associated with *Baikiaea* forests in the low rainfall regions (southeast Angola, northeast Namibia and western Zambia) and with Miombo woodland (*Brachystegia*, *Isoberlinia*, *Julbernardia*) in the high rainfall regions (Mwitwa, 2004).

The species is the best known, most valuable and mostly widely used wood in Southern Africa for timber and other uses (Coates-Palgrave, 2002; Van Holsbeeck *et al.*, 2016). The valuable timber is highly sought after for wood carving and carpentry (Curtis & Mannheimer, 2005; Graz, 2004a; Mendelsohn & el Obeid, 2005; Shackleton, 2002). According to Coates-Palgrave (2002), the species has a multipurpose use including utilitarian and traditional folk medicine. It is also an effective dye when the red, sticky and blood-like sap is used on the cloth. Other qualities that make it preferred for high quality furniture production include among others golden or reddish brown heartwood, minimal knots, easy workability, glues and screws well, and fine polishing. In addition, very little shrinking coupled with high durability, making the species ideal for boat building and canoe construction (Coates-Palgrave, 2002; Mendelsohn & el Obeid, 2005). Excessive exploitation for its valuable timber and land use change are a threat to the existence of the species (Mojeremane & Lumbile, 2016). Shoot dieback associated with high occurrence of forest fires, elephant damage (Mmolotsi *et al.*, 2012), and failure of seedlings to survive are contributing to low population densities, a threat of extinction due to lack of young trees in the wild (Mojeremane & Lumbile, 2016). Other authors add that low recruitment success coupled with logging pressure to satisfy expanding market demand are contributing to reduction in growing stock across its distribution range (Schwartz *et al.*, 2002).

2.5.6. *Schinziophyton rautanenii*, (Schinz) Radcl.-Sm

S. rautanenii is a medium to large-sized deciduous tree that can attain height of 15 – 20 m (Coates-Palgrave, 2002). Other attributes are smooth bark, which is yellow-grey to golden brown and has tendency of peeling off sometimes, and digitate leaves forming 5 – 7 leaflets in a spiral arrangement (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Mendelsohn & el Obeid, 2005). According to Coates-Palgrave (2002), the species has a thickset and stout trunk;

wide spread crown (Chidumayo, 2016), which is described by Mendelsohn and el Obeid (2005) to be one that varies from rounded shape for isolated trees and flat topped for trees in dense stands. In addition, flowers are yellow in colour, fruits are green–grey containing one or two nuts covered in extremely hard shell. The tree is mostly multi-stemmed resulting from forking within 2 m above the ground (Chidumayo, 2016). The species is locally known by other names such as Manketti-tree, Mankettiboom, Mugongo or Umunkete (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Mendelsohn & el Obeid, 2005).

The species largely grows on wooded hills and among sand dunes, sometimes forming a pure stand (Coates-Palgrave, 2002) and in broad-leaved semi-deciduous woodlands on Kalahari sands (Scholes *et al.*, 2002). The habitats include being found on sub-humid and semi-arid regions of Southern Africa (Chidumayo, 2016). The species is patchily distributed in north-east Namibia (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005) but the range outside Namibia include narrow belt across southern Angola, northern Botswana, Malawi, Mozambique, Zambia and Zimbabwe (Mendelsohn & el Obeid, 2005). Distribution of the species in Zambia is throughout but dominating in the southwest region bordering Botswana, Namibia and Zimbabwe (Chidumayo, 2016). Growing conditions for the species among others include soils dominated by Kalahari sands, as pointed out by other authors, annual rainfall of 625 – 700 mm, annual minimum and maximum temperatures of 4-19 °C and 25-35 °C respectively.

Wood of the species is pale yellowish white, with grains that are wavy and coarse texture. The wood, though comparatively very light, is strong and its use include in light packing cases, insulating materials, floats, toys, drawing boards and dart boards (Coates-Palgrave, 2002). The author include the consumption of porridge made from dry fruit pulp scrapped from the kernel as another use of the species in Namibia. Overall, the species is a multi-purpose tree ranging from being food source for animals, such as elephants, to humans for fruits with a sweet and edible pulp. Local communities extract edible oil from seed kernels, use the tree for shed, and make hunting bows, spoons, drums and art crafts from the light timber (Chidumayo, 2016; Curtis & Mannheimer, 2005).

2.5.7. *Terminalia sericea*, Burch. Ex DC

Various authors have described *T. sericea* as a small to medium deciduous tree (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Mendelsohn & el Obeid, 2005; van Wyk & van Wyk, 1997). Mendelsohn and el Obeid (2005) put the height of the species at 3 – 15 m, with crown of the species slightly rounded to flattish and layered (van Wyk & van Wyk, 1997). The work of Coates-Palgrave (2002) put the height estimates at 4 – 6 m and occasionally reaching

10 m. However, some individual trees may reach up to 23 m in height (Orwa *et al.*, 2009). Other factors that distinguish the species as outlined by Curtis and Mannheimer (2005) include a dark grey and deeply fissured bark, purplish branchlets, which peels and flakes in rings exposing a tan under-bark. Leaves tend to cluster towards the tip of branchlets that are dark brown or purplish but leaves themselves are obovate-elliptic and thinly textured (Coates-Palgrave, 2002; Curtis & Mannheimer, 2005; Orwa *et al.*, 2009; van Wyk & van Wyk, 1997). Flowers are small spikes; cream to pale yellow appearing before the first rains in September and October, while fruits are pale-red, pinkish or flesh-coloured having two wings on either side of a single kernel. Localised names for the species are Sand geelhout, Muhorono, Silver cluster-leaf, Vaalbos (Mendelsohn & el Obeid, 2005).

Habitats for the species are generally those on deep Kalahari sandy soils, often in open areas but also in open woodlands growing as bushes under other tree species (Mendelsohn & el Obeid, 2005). It is sometimes dominant or co-dominant in mixed deciduous forests (Orwa *et al.*, 2009).

The wood qualities for species are distinctively yellow colour, with fine grains, hard and durable (Mendelsohn & el Obeid, 2005). The species is also heavy, tough and resistant to both termites and borers, thus, giving the species a general purpose timber use and suitable for furniture (Coates-Palgrave, 2002; Orwa *et al.*, 2009; van Wyk & van Wyk, 1997). As stated by Mendelsohn and el Obeid (2005), uses of the species varies to include leaves providing fodder for various wild and domestic animals, including caterpillars. Others uses include glazing of pottery, manufacture of domestic utensils, furniture, poles for construction of huts and fences, and host of traditional medicinal applications. Roots of the species have wide applications in traditional medicinal use to cure a number of ailments in local communities (Coates-Palgrave, 2002; Orwa *et al.*, 2009; van Wyk & van Wyk, 1997).

2.6. Chapter summary

Chapter two has presented the various works relating to this study done by other authors and researchers. It highlighted the extent and range of the Zambezian phytoregion, which lies between 3 °S and 26 °S, including the forest types of Namibia and Zambia as part of the forest biome in the phytoregion. Natural regeneration, its importance and the type were described for the forests and woodlands. Forest fire was also looked at describing the sources, fire regime, frequency, interval and its effect on woody vegetation. The last section of the chapter highlighted the selected tree species by describing their physical properties, occurrence and distribution range.

Chapter 3

RESEARCH DESIGN AND METHODOLOGY

3.1. Introduction

This chapter highlights the study locations, and their vegetation, soil and climatic conditions. It also describes the research design, methods and materials used in data collection, processing and analyses.

3.2. Study location

The study took place in two countries of Southern Africa; Namibia and Zambia as shown in Figure 3.1 and described in Section 3.2.1, following a rainfall gradient from a dry to a wetter area.

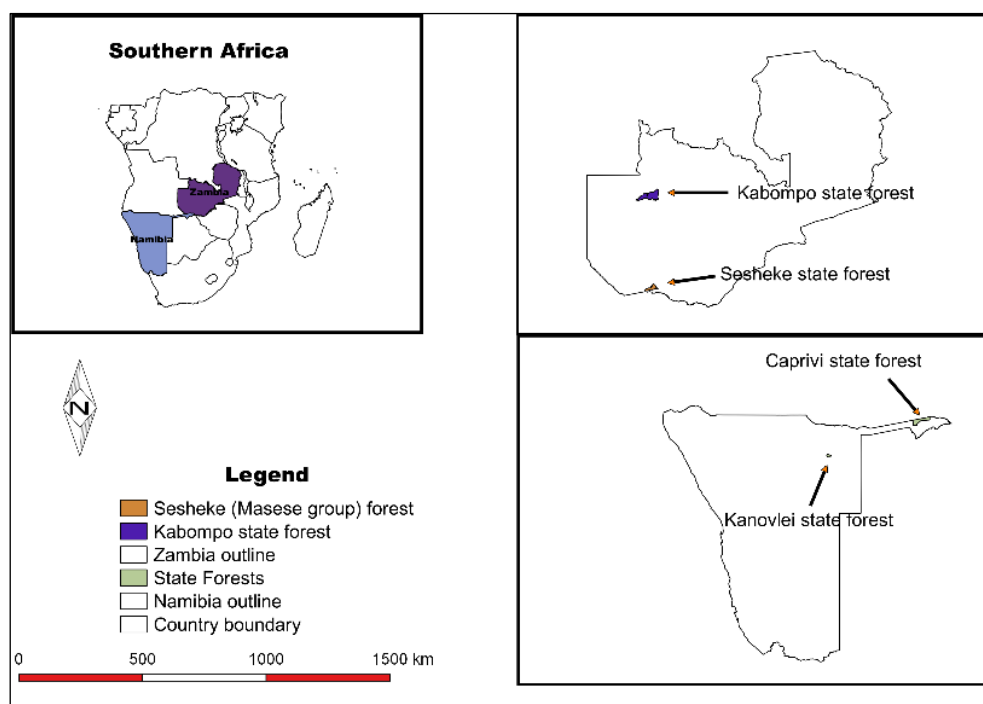


Figure 3.1: The location of the study countries in Southern Africa (top left) and the location of the study sites (state forests) in their respective countries of Zambia (top right) and Namibia (bottom right).

3.2.1. Study countries

Namibia - is located in the southwestern corner on the African continent between latitudes 17° and 29° S, and longitudes 11.7° and 25.4° E covering a total land area of 830 000 km² (FAO, 2003). The country shares borders with Angola in the north, Zambia in the northeast, Botswana in the east, South Africa in the south and the Atlantic Ocean in the west (Sweet & Burke, 2006).

Isaacson (1995) cited in (Sweet & Burke, 2006) described three main physiographic regions of the country. The first is the western coastal plain of the Namib Desert covering 12% of the total land mass. The second is the central plateau, which cover half of the country from the southern to the northern border, and the third is a stretch of semi-arid Kalahari zone on the eastern side of the country. As Sweet & Burke (2006) pointed out, western coastal plains comprises mobile dunes, gravel and sandy plains whereas, central plateau is made up of highland areas and the great western escarpment. Sand of varying thickness cover the Kalahari zone. Generally, the country has soils that are shallow, with low nutrient content and poor moisture retention (Mendelsohn & el Obeid, 2005; Moses, 2013). Namibia's area of 82.429 million ha is divided into 7.290 million ha of forests, 8.290 million ha of other wooded land, 66.749 million ha of other land-use and 100 thousand ha of inland water (FAO, 2010).

Zambia - is a landlocked country located in Southern Africa between latitudes 8° and 18° S, and longitudes 22° and 34° E with a total land mass of 752 614 km². The country shares borders with the Democratic Republic of Congo (DRC) in the north, Tanzania in the northeast, Malawi to the east, Mozambique to the southeast, Zimbabwe and Botswana in the south, Namibia in the southwest and Angola in the west (Mukosha & Siampale, 2009).

Zambia is largely a high plateau with elevations between 900 m and 1 600 m above mean sea level (a.m.s.l) (Mackenzie, 2014; Mukosha & Siampale, 2009) with exceptions of Muchinga Mountain with the highest point at 1 800 m a.m.s.l in the northeast and the lowest points are found at 400 a.m.s.l in major river valleys (Mackenzie, 2014). Zambia's land mass of 75.261 million ha is divided into 49.968 million ha of forests, 6.055 million ha of other wooded land, 15.771 million ha of other land use and 3.467 million ha of inland water (Mackenzie, 2014; Mukosha & Siampale, 2009).

3.2.2. Study sites

Four study sites were chosen for data collection. Each country had two study sites based on mean annual precipitation (MAP) and the primary land use of being a protected forest area. This was to minimise impact of human and domestic animals' disturbances on the study since primary objective is to assess the impact of fire treatment period along a rainfall gradient.

Study sites in Namibia were Kanovlei State Forest (21 590 ha) in the Otjozondjupa region and Caprivi State Forest (146 100 ha) in the Zambezi region. Kanovlei State Forest is located at 612 km northeast of Windhoek (nation's capital city) whereas Caprivi State Forest is located at 1 230 km northeast of Windhoek and 10 km west of Katima Mulilo town.

Study sites in Zambia were Sesheke (Masese group of forests as they are officially known) State Forest (960 835 ha) and Kabompo State Forest (97 374 ha), which are located at 676 km and 670 km, southwest and northwest of Lusaka (nation's capital city) respectively. The trajectory from Kanovlei (in the southwest of the study area) to Kabompo in the northeast marks the longest distance along the rainfall gradient. These two study sites are located at 1 897 km apart and represent a rainfall gradient from approximately 500 to 1 000 mm per annum. Figure 3.2 shows the location of sample plots in their respective state forests.

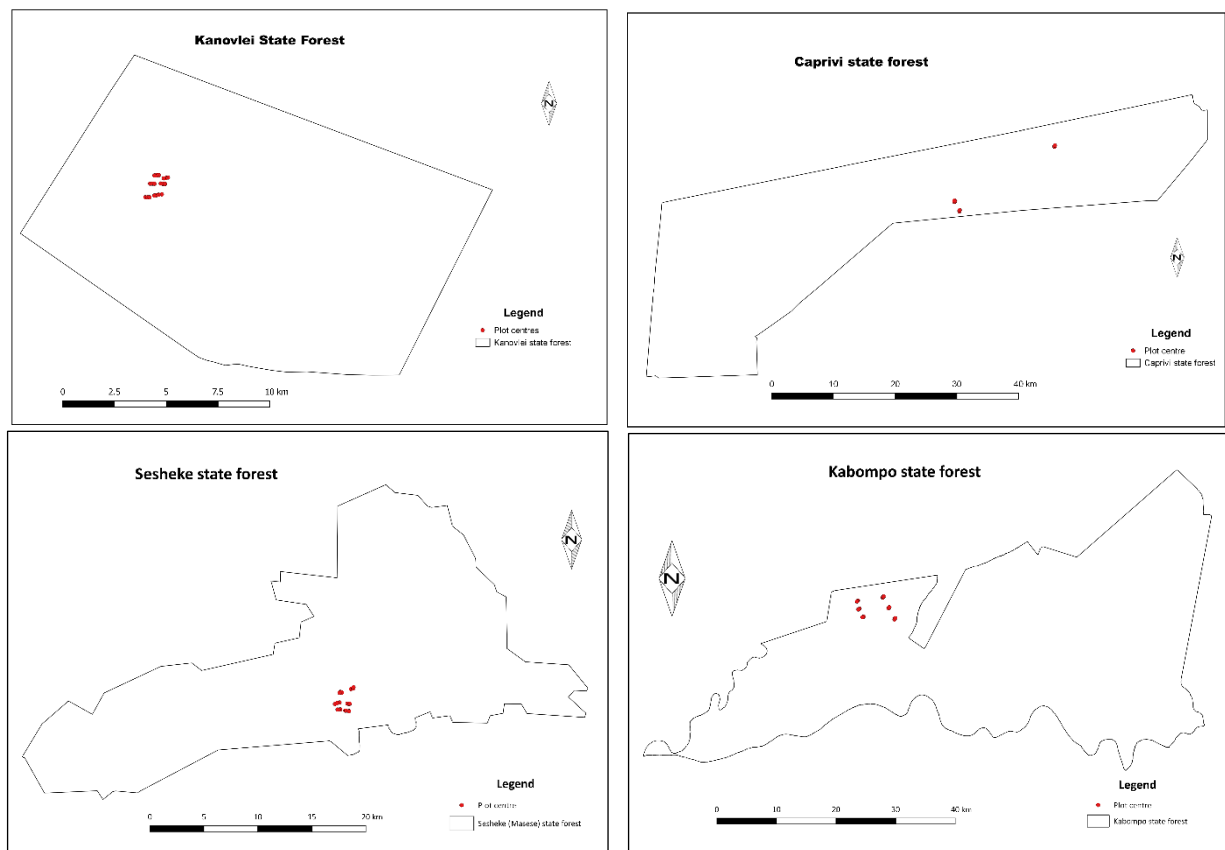


Figure 3.2: The location of sample plots at the respective study sites in Namibia (top left & right) and Zambia (bottom left & right).

3.3. Vegetation

The vegetation type found in Kanovlei state forest is open forest dominated by *B. africana* and *P. angolensis* accounting for half of the total stem volume. The major woody species are *B. africana*, *P. angolensis* and *T. sericea* as the most abundant while *O. pulchra*, *S. rautanenii* and *S. cocculoides* are also found in small numbers (Muhoko & Kamwi, 2014).

The vegetation type of Caprivi state forest belong to the woodlands that were once dominated by *B. plurijuga* and *P. angolensis*, which is part of upland *Baikiaea* woodlands (Chakanga *et al.*, 1998). These woodlands form part of the savanna woodlands covering the Otjozondjupa, Omaheke, Kavango and Zambezi regions of Namibia while extending into southern Angola,

northwest Botswana, southwest Zambia and northwest Zimbabwe. The inventory of Chakanga et al. (1998) found that the most common tree species of Caprivi state forest in the canopy were *B. plurijuga*, *Combretum collinum*, *T. sericea*, *B. africana*, *Philenoptera nelsii* and *P. angolensis* with *B. plurijuga* being the most abundant per hectare. The shrub level comprised mainly the species that were found at canopy level with an exception of *Baphia massaiensis* and *Bauhinia petersiana* that are shrubs.

Vegetation in Sesheke state forest is a two-storey *Baikiaea* forest. The forests are either open or closed canopy in the range of 9 – 19 m height with maximum height reaching 24 – 27 m (Fanshawe, 2010). Further, the author describes the dominant and co-dominant species found in this forest as *B. plurijuga* and *Pterocarpus lucens* with variations in proportion in localised groups of *Commiphora angolensis* and *S. rautanenii*. Sub-dominant species are *Boscia albitrunca*, *C. angolensis*, *Croton gratissimus*, *Philenoptera nelsii*, *Strychnos innocua*, and *Ficus sycomorus* whereas *B. massaiensis*, *B. petersiana*, *Combretum celastroides*, *Grewia avellana*, *C. gratissimus*, *Strophantus welwitschii* and *Dalbergia nitidula* belong to the shrub layer.

The vegetation type in Kabompo state forest is miombo woodland that is dominated by the *Brachystegia-Isoberlinia-Julbernardia* genera (Fanshawe, 2010). Common dominant and co-dominant species among others are *B. boehmii*, *B. floribunda*, *B. longifolia*, *B. spiciformis*, *E. africanum*, *I. angolensis*, *J. paniculata* and *M. macroura*. According to Fanshawe (2010), the small trees at middle canopy level belong to *Diospyros kirkii*, *P. maprouneifolia*, *S. cocculoides*, *U. kirkiana* and *U. nitida* whereas *B. petersiana* and *Ekebergia benguelensis* belong to the shrub layer. Figure 3.3 shows the vegetation types in the respective study sites during the data collection period between June and November.

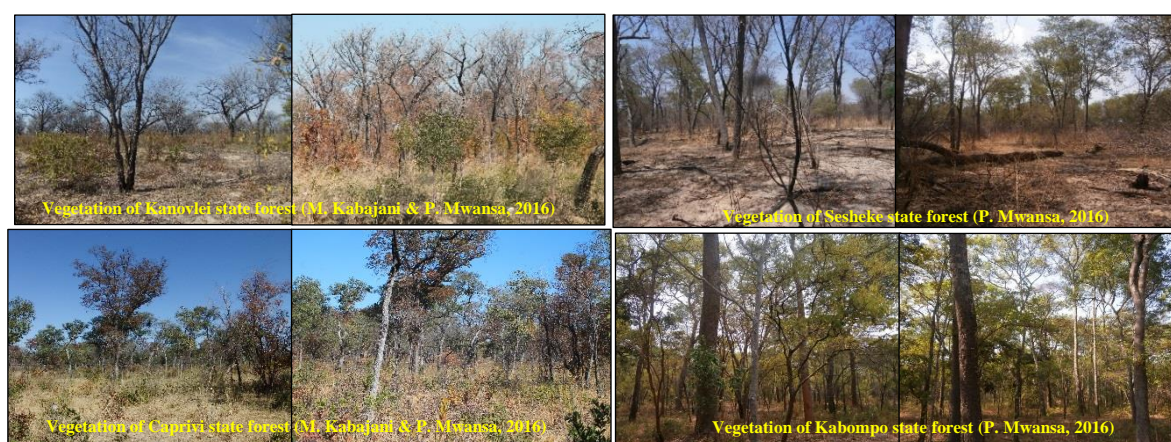


Figure 3.3: Vegetation structure in all the four study sites where sampling took place showing a very clear distinction in vegetation structure from dry site at Kanovlei state forest to a wetter miombo at Kabompo state forest.

3.4. Climatic conditions

Namibia has very distinct wet and dry periods with the rainy season running from end of September to mid-March (Graz, 2004b) citing Amakali (1992). Significant annual variations in both the amount and timing of precipitation exist across Namibia forming a clear gradient that emerges from southwestern parts of the woodlands way up to the northeast of the region (Graz, 2004b). Annual rainfall variation range from less than 20 mm in the western Namib and coastal areas to more than 700 mm at the eastern end in Zambezi region (Sweet & Burke, 2006). Sweet & Burke (2006) stated that the major part of the country receive summer rainfall with the bulk of it falling between the months of November and March while winter rains occur at the very southwest of the country. Generally, temperature range fluctuates according to the region throughout the year (Okitsu, 2005). Okitsu affirmed that mean daily maximum temperatures of 34 – 36 °C occur in October, which is the hottest month. On the extreme end, less than 2 °C to more than 10 °C is the average daily minimum temperature in the coldest months of July and August for most part of the country and on the coast, respectively.

Zambia is divided into three agro-ecological zones based on rainfall and soil characteristics. Region I cover most of the southern and part of the eastern parts of the country receiving annual rainfall less than 800 mm. Region II cover the western, central and eastern parts of the country receiving annual rainfall between 800 mm and 1 000 mm. Region II is further divided into II-a (predominantly original fertile soils) and II-b (predominantly sandy soils). Region III is a high rainfall area covering the northern and northwestern parts of the country. The region receives annual rainfall ranging between 1 000 mm and 1 500 mm (JAICAF, 2008). Zambia receives summer rainfall between November and March averaging 500 mm – 1 500 mm (FAO, 2009; JAICAF, 2008; Mukosha & Siampale, 2009). MAP decreases from 1 200 mm to 600 mm from north and northwestern to south and southwest (FAO, 2009; Mukosha & Siampale, 2009). The Sesheke and Kabompo study sites are found in two different agro-ecological zones. Sesheke is in region I and receives an annual rainfall of 670 mm, which is scattered and variable while Kabompo is in region III, with a MAP of 1 120 mm (Fanshawe, 2010). Both sites receive summer rainfall from mid-November to March for Sesheke and from late October to March for Kabompo. Fanshawe (2010) also pointed out that heavy dew is a common occurrence during winter months in Sesheke. Temperature is extreme reaching a maximum of 38 °C from September to February while the rest of the months the maximum is around 31 °C. Night radiation causes night temperature to drop drastically resulting into frost in low lying areas and depressions during winter. Frost in Kabompo occur in hollows and associated complex shallow

wetlands commonly known as dambos during the cold months of June and July (Fanshawe, 2010).

3.4.1. Rainfall gradient

The Zambezian phytoregion experiences general climatic conditions that favour dry forests and woodlands (Chidumayo & Gumbo, 2010). The region is classified as the Southern sub-humid tropical zone of Africa (Campbell, 1996). As earlier stated in Section 2.1, the region receives rains during summer rainy season ranging from 3 to 7 months (Campbell, 1996; Chidumayo & Gumbo, 2010).

The rainfall gradient considered in this study was within the south to north gradient, which increase in MAP and approximated from 500 mm y⁻¹ to 1 000 mm y⁻¹. Thus, Kanovlei in the south had the lowest MAP and Kabompo in the north had the highest MAP, as shown in Table 3.1.

Table 3.1: Summary of the mean annual precipitation for the study sites for the period between 1990 and 2015. Source of data used in calculations of the mean is Namibia meteorological service and Zambia meteorological department.

Study site	Annual precipitation (mm)			
	Mean	Standard error	Minimum	Maximum
Kanovlei State Forest	551	±46.8	275	944
Sesheke State Forest	633	±9.6	523	733
Caprivi State Forest	741	±26.7	566	982
Kabompo State Forest	1 024	±28.4	819	1 278

3.4.2. Temperature range

The mean annual temperature (MAT) readings for the study sites are summarised in Table 3.2. The data sets were based on the readings from the nearest meteorological stations at Grootfontein and Katima Mulilo in the case of the Kanovlei and Caprivi study sites whereas, that of the Sesheke and Kabompo study sites were based on the meteorological stations at Sesheke and Kabompo towns.

Table 3.2: Summary of mean annual maximum temperature across study sites for the period between 1990 and 2015. Source of data used in calculations of the mean is Namibia meteorological service and Zambia meteorological department.

Study site	Annual temperature (° C)	
	Mean daily	Mean maximum
Kanovlei State Forest	21	29
Sesheke State Forest	-	31
Caprivi State Forest	23	30
Kabompo State Forest	-	28

3.4.3. Fire frequency, fire return period and main fire season

The number of fires that occurred in the study sites between 2001 and 2015 were determined from the moderate resolution imaging spectroradiometer (MODIS) MCD45 burned area product. MODIS burned product has a spatial resolution of 500 m pixel size on a monthly basis that includes the burn day (Stellmes, Frantz, Finckh, Revermann, Roder & Hill, 2013).

The pre-processed burned area shapefiles of 2001 – 2015 were obtained from the Council for Scientific and Industrial Research (CSIR). The fire data in geographic information system (GIS) format was then processed using quantum geographic information system (QGIS) (Quantum GIS Development Team, 2016) to generate fire maps for the study sites. The fire frequency for each plot and the main fire season per respective study, when most fires occur, were determined from the fire maps. Most of the plots at all the four study sites burned nearly on annual basis from 2001 to 2015. If the plot burned more than once in a particular year, the fire occurrence was regarded as one for the plot in that year. Thus, fire frequency on the plots was classified according to the number of times a plot burned during the 14-year period as none (0), low (1 – 3), medium (4 - 6) and high (> 6). The actual number of burns on each plot is shown in Appendix 3.1. The plots in areas at the study sites that burnt in 2014 and 2015 were classified as recently burnt (RB) while plots in areas that burnt in 2013 or beyond were classified as recently unburnt (RU). The mean fire return interval for the RB and RU was derived by dividing the sum of fire return intervals by the number of annual fires experienced during the period 2001 – 2015. Table 3.3 provides a summary of an average fire return interval for RB and RU areas across the study sites.

Table 3.3: The average fire return interval and main fire season when most fires occur across study for recently burnt (RB) and recently unburnt (RU) sites.

Study site	Fire return interval (years)		Fire season
	RB	RU	
Kanovlei	2.3	2.0	August - October
Sesheke	1.0	1.2	May - October
Caprivi	1.4	2.0	June - October
Kabompo	1.1	2.0	May - October

3.5. Soil conditions

Generally, all the four study sites are in the Kalahari sand basin stretching from the north of South Africa to the southwest of DRC.

The Kanovlei and Caprivi study sites both have the same soil types, deep sands of arenosols and dunes (Burke, 2006; Mendelsohn & el Obeid, 2005). The soils are generally made up of more than 70% sand and the rest consisting of clay and silt particles. Due to the sandy texture of the soil, water is allowed to drain through rapidly leaving little moisture and few nutrients at depths where plant roots can reach (Mendelsohn *et al.*, 2002).

The soil type at the Sesheke study site is Kalahari sands in nature that could probably be underlain by Karoo beds. The sands are from the Pleistocene deposits varying from pale grey to orange colour, moderately acidic, having a 3 – 12 % fraction of silt and clay with fine sand content varying at 15 – 60% (Fanshawe, 2010). In Kabompo, Fanshawe (2010) classified the soil type as Kalahari sands of upland sands. These sands are pale grey to reddish brown colour, deep, well drained and has fairly coarse grains with a 10 -12% fraction of silt and clay. Figure 3.4 shows the general extent of the Kalahari sand basin.

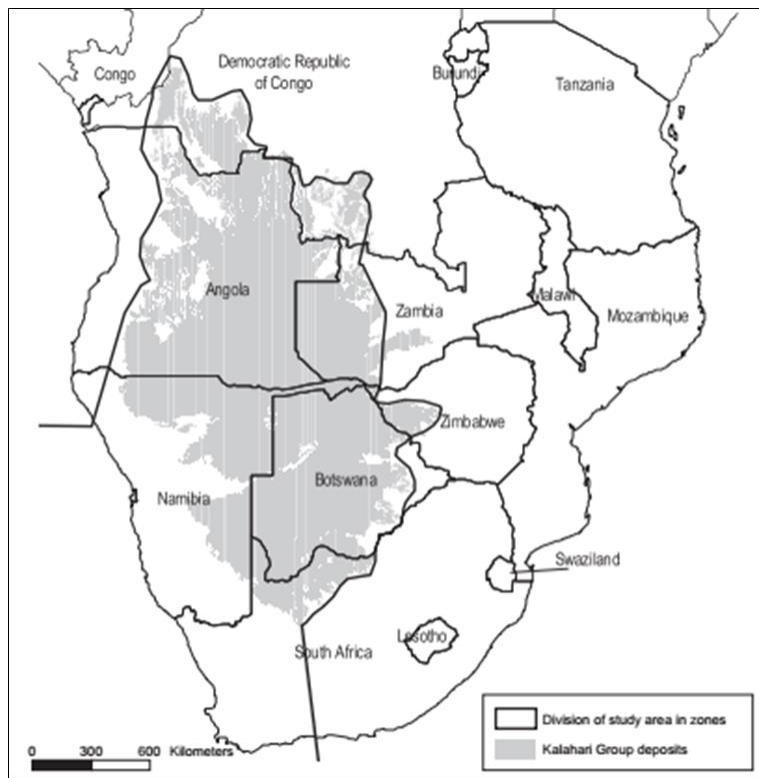


Figure 3.4: Extent of the Kalahari sand basin portrayed in the work of De Cauwer *et al.* (2014) based on the data of Mendelsohn and el Obeid (2004).

3.6. Research design

This study employed a systematic random sampling design, adapting from the field instructions module for the National Forest Inventory (NFI) of Namibia (Burke *et al.*, 2001). Systematic sampling refers to the way the study sites were selected along a rainfall gradient and to the selection of the sample area that were recently burnt and recently unburnt. Random sampling was in the siting of the sample plots that were placed randomly within the fire treatment areas. The NFI highly recommends the use of systematic sampling. However, stratification method can equally be used in the inventories if there is prior knowledge of the area (Burke *et al.*, 2001). Use of systematic random sampling is necessary because of the method's ability to increase sampling accuracy, valid test of significance and keeps the random error at the lowest possible level (Jayaraman, 1999). The method is highly suitable for sampling large areas, flexible for heterogeneity, and respond to change in a quick way (Brown *et al.*, 2013). Further, the method is a sampling strategy known for being efficient when sampling vegetation cover and other

natural resources (Aune-Lundberg & Strand, 2014). Sample plots in a cluster are spaced at 100 metres apart, from one plot centre to the next (Figure 3.5).

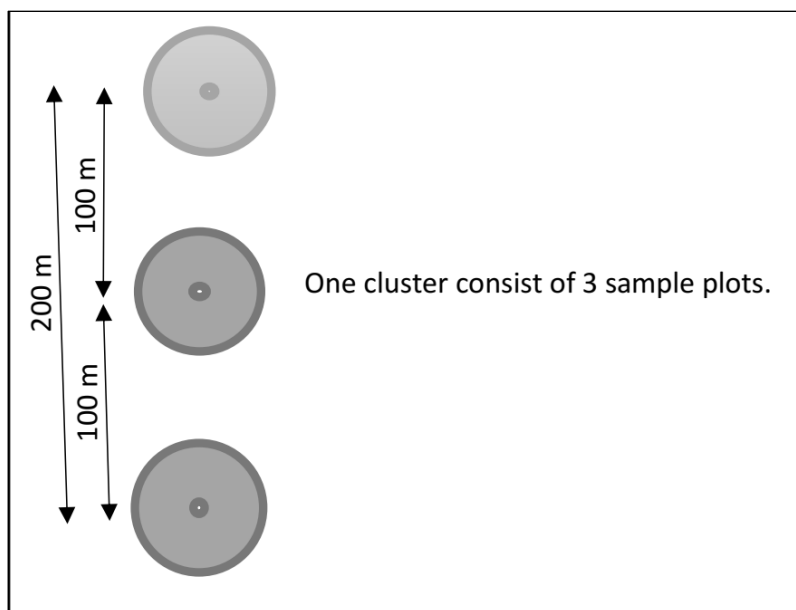


Figure 3.5: Placement of sampling plots on one cluster as demonstrated in the field instructions of the National Forest Inventories of Namibia (Burkea *et al.*, 2001).

Madow & Madow (1944) in their postulation on the theory of systematic sampling justified that the use of cluster sampling reduces biasness, enhances consistence and efficiency in sampling.

3.6.1. Plot layout

The area in which sampling took place were pre-selected based on the fire history using GIS fire data and maps. The areas on the map were categorised into recently burnt (RB) and recently unburnt (RU) depending on how recent the plot burned as stated in Section 3.4.3. Accessibility was an additional factor in determining which areas to sample from, even though it could have led to some bias in measurements (De Cauwer, 2012).

The layout of plots in this study followed the field instructions of the NFI (Burke *et al.*, 2001) by using a three plot-cluster in the south - north direction. The first plot was the south most followed by the second plot at 100 m north of the first and the third plot was located at 200 m north of first plot. The distance between one cluster and the next in the study sites varied between 1 km and 2.5 km depending on the extent of either RB area or RU area.

Nine (9) replication plots were sited in each of the RB and RU areas, in the Kanovlei, Sesheke and Kabompo study sites following a 2 x 4 factorial treatment structure (2 levels of fire treatment representing RB and RU, and 4 levels of MAP representing each study site). However, the desired sampling design could not be attained in Caprivi as there was too much

human disturbance. Therefore, with the exception of Caprivi state forest, the other sites had three clusters per sample site in RB and RU. Sampling in the RB and the adjacent RU plots determine successional patterns after a disturbance (Ohl, 2005). A reconnaissance survey was carried out to verify the selected sites on the ground and establish the most suitable sampling sites (Chidumayo, 1987b).

3.7. Plot design

A concentric circular plot of 30 m radius with an area of 2 827m² is the recommended plot design (Kabajani, 2016). Other concentric circles used for sampling within the main plot are 3.99 m radius with an area of 50 m², 10 m radius (314 m²) and 20 m radius (1 257 m²) (Burke *et al.*, 2001). Kabajani (2016), citing Kuru *et al.* (2015) justified the use of concentric circular plots as being relatively better than square plots because they reduce the potential of skewing of boundary lines by obstacles.

The general design of the sampling plot is illustrated in Figure 3.6 and each of the concentric circle was used to sample different variables as explained in the following subsections.

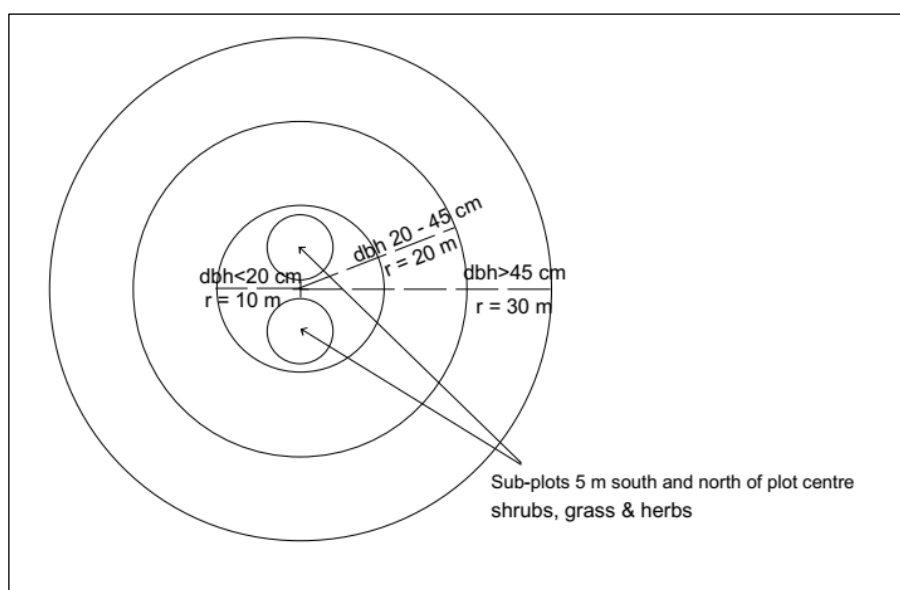


Figure 3.6: The general plot design and layout recommended in the filed instructions manual for forestry inventory in Namibia (Burke *et al.*, 2001).

3.7.1. Circular Plot – 10m radius

In this circle, all trees with diameter at breast height (dbh) greater or equal to 5 cm were measured and identified at species level. Diameter, height and other tree attributes (see Appendix 1.2) were entered on the record sheet. The area of this circle is 314 m² that is

equivalent to 0.0314 ha. Therefore, to up scale the number of individuals to a hectare, a factor of 31.831 was used.

3.7.2. Circular Plot – 20m radius

This is an area in the sample plot outside the circle with a 10 m radius in which all trees whose dbh was greater than 20 cm, was measured. In this area, trees with dbh less than 20 cm were not sampled. All other measurements and recording were done as applied in the 10 m radius plot. This circle has an area of 1 257 m² (0.1257 ha). Thus, a factor of 7.958 was used to up scale the number of individuals to a hectare.

3.7.3. Circular Plot – 30m radius

Sampling of trees in this area was done outside the second circle with a 20 m radius where all trees with dbh equal to 45 cm or more were measured. The area covered the distance between the 20 m and 30 m radii for dbh measurements whereas other measurements such as biophysical data, stand description, crown coverage, land use and land type, considered the whole 30 m radius from the plot centre (Burke *et al.*, 2001). The number of individuals in this circle were up scaled to a hectare using the factor 3.537 based on its area of 0.2827 ha (2 827 m²).

3.7.4. Circular subplot – 3.99 m radius

A subplot was established within the main plot where all woody vegetation (seedlings, shrubs) of dbh less than 5 cm were counted (Burke *et al.*, 2001; De Cauwer, 2012). Shrubs and regeneration subplot had a 3.99 cm radius (Burke *et al.*, 2001). A similar subplot radius was used in the national forest inventory conducted in Zambia. The 3.99 cm radius subplot was used to measure regeneration and shrubs, and they also used the subplot centre to carry out the edaphic and topographic measurements on the subplot (Mukosha & Siampale, 2009). Burke *et al* (2001) advised to locate two subplots within the sample plot at 5 m north and south of the main sample plot centre as shown in Figure 3.7. In this study, two subplots were used and set up accordingly for regeneration sampling. Each subplot has an area of 50.01 m², which is equivalent to 0.005 ha. Subsequently, individual counts were up scaled to a hectare using the factor of 200.

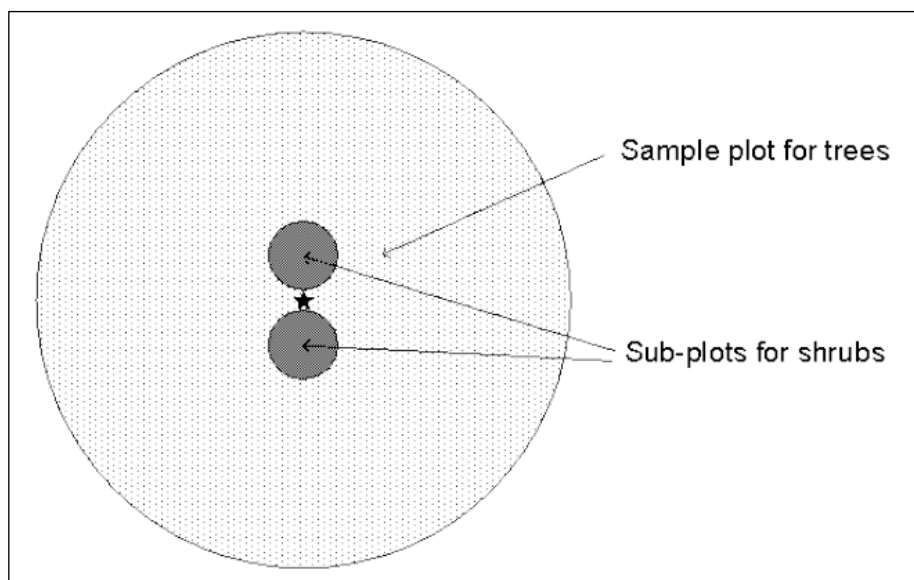


Figure 3.7: The location of the two subplots within the main sample plot for shrubs and regeneration sampling based on the NFI guidelines (Burke *et al.*, 2001).

3.8. Field work and sampling

Fieldwork and sampling were conducted in two phases. The first phase of sampling was in Namibia with the first visit at Caprivi state forest from 23rd June to 29th June 2016. Three days were spent on a reconnaissance survey to verify the sampling area on the ground. Sampling at Kanovlei state forest took place from 1st July to 6th July 2016. Only a day was spent on reconnaissance survey. The second phase of sampling was in Zambia with Kabompo state forest visited first from 21st to 27th October 2016. Sampling at Sesheke state forest was from 03rd to 10th November 2016.

3.8.1. Materials and methods

Field work requires a variety of tools and equipment (Burke *et al.*, 2001; De Cauwer, 2012; Condit, 2008; Mukosha & Siampale, 2009). The following materials and equipment were used for this study.

- i. 360° Compass – direction and orientation of plots
- ii. GPS set and accessories – navigation and geographical location of plots
- iii. 2 diameter tapes and calliper – dbh measurements
- iv. Laser vertex clinometer – tree distance from plot centre, tree height, plot radius of the plot circles and verifying if a tree is within the plot circle
- v. 50 m self-rolling tape – assessment of substrate cover on the plot
- vi. Suunto Clinometer/Hypsometer – backup for tree height measurements

- vii. Dendrometer – basal area estimation on the plot
- viii. Bitterlich gauge – vegetation (canopy and shrub) cover estimation
- ix. Collapsible ruler (5 m) – estimation of regeneration, small trees and shrubs' height
- x. Digital camera – capturing images of the plot surroundings
- xi. Topography and field maps – verifying of fire maps
- xii. Field manual – reference on methods
- xiii. Flora and species list – identification of tree species
- xiv. Clip board, pens & inventory sheets – data capture
- xv. First Aid kit – field emergencies

Before any sampling could be undertaken in the field, potential sample sites were pre-selected after processing fire data in GIS format into fire maps at Stellenbosch University. Each study site had at least three potential sample sites that had RB and adjacent RU areas for possible sampling. Upon arrival at the study site, a reconnaissance survey was adequately conducted in order to find the suitable sites (Chidumayo, 1987b). The highest number of reconnaissance survey days were spent at Caprivi state forest (3 days), followed by Sesheke with 2 days, while only 1 day was spent at both Kanovlei and Kabompo. Caprivi and Sesheke state forests are heavily disturbed by humans to the extent of failing to meet the desired number of sample plots free of human disturbance at Caprivi. However, the required number of sample plots at Sesheke state forest could be established.

Once the sample site was established, the location of plots were about 50 – 200 m away from fire cut lines and/or forest roads (De Cauwer, 2012). It was further suggested to use either coordinates of the potential plot site or throwing an object, such as a twig, over one's shoulder as a means of selecting randomly the plot centre. The latter method was used in this study. After locating the first plot centre, stand information was recorded. Among the first things to be recorded were the geographical location of the plot using a handheld Geographical Positioning System (GPS). Other information included biophysical data and stand description of the plot. This included taking of photographs of the surroundings on the plot. Crown and shrub cover were recorded using a Bitterlich gauge while substrate cover was determined using the line intercept method. Estimation of substrate cover is an assessment of herbs, grass and shrub cover on the plot. Figure 3.8 highlights some of the materials and methods used during the data collection exercise.



Figure 3.8: The depiction of some of the materials and methods used in data collection. A) GPS geographical location data capture; B) substrate cover; C) crown and shrub cover measurement; D) tree height reading; E) Collapsible ruler used for shrub and regeneration height estimation; F) dbh measurement; G) vertex transponder for tree distance and H) data capture on record sheets.

After completing sampling on the first plot, then the second plot was set up at 100 m north of the first plot and the third plot at 200 m north of the first plot to complete a three plot-cluster. All measurements and recordings were repeated exactly as in the first plot on the subsequent plots. The distance and the direction from one plot centre to the next were determined using the GPS and the compass respectively. It took on average of 2 hours and 30 minutes to complete sampling of one plot. Table 3.4 shows summary of the number of sample plots and subplots for all the four study sites.

Table 3.4: Summary of the number of sample plots and subplots where data was collected in the four study sites.

Study site	Sampling Month	No. Sample plots	No. Subplots
Kanovlei state forest	July	19	38
Sesheke state forest	November	18	36
Caprivi state forest	June	10	20
Kabompo state forest	October	18	36
Total		65	130

3.8.2. Field team composition

A field team comprises all those working in one plot at any given time and the number may differ, ranging from 1 to 10 (Condit, 2008). In this study, the team comprised a minimum of two members, for Caprivi and Kabompo state forests, and three members for Kanovlei and Sesheke state forests. Team members were the lead student (study owner) and assistant(s) who were very knowledgeable in the respective state forests.

Supervisor/Co-supervisor – according to Condit (2008), are University professionals cum experienced scientists, with an intimate knowledge of forest study and methods, overseeing the study from planning to execution and writing. Responsibilities included giving expert guidance to the lead student, field orientation of the team on materials and methods, checking for quality assurance of the study. The main supervisor accompanied the team on the Namibian excursion during the reconnaissance survey at Kanovlei and Caprivi state forests and left the team to continue with work after a few days.

3.8.3. Measurements and data parameters

Various measurements and data parameters were recorded as described in the preceding sections using record sheets adapted from Burke *et al.* (2001) by De Cauwer (2012). See Appendices 1.1 – 1.5 for the design of the data inventory sheets.

Stand information relating to each plot was recorded. This included general information describing the area within and surrounding the sample plot, coordinates of the plot, elevation, land type, land use, disturbance type on the plot and cause of damage to trees. Vegetation cover was assessed and recorded in form of crown and shrub coverage. Crown and shrub cover was estimated based on the variable-plot method (Cooper, 1957) using a Bitterlich gauge. The method does not require actual measurements in the field. It only take an observer to hold the

equipment close to the eye and swoop around the plot, starting from the south moving clockwise, counting the crowns of the individual trees that are outside the cross hairs of the Bitterlich gauge (Cooper, 1957; Holck, 2008). All individuals counted in were identified to species level. In this study, a constant of 0.05 (percentage cover value of 5%) was multiplied by the number of individuals counted in to determine the percentage of crown cover in the forest (Friedel & Chewings, 1988). The same method was used in an earlier work of SASSCAL Task 038 (Kabajani, 2016). Dimensions of the Bitterlich gauge used in the study were: cross arm length of 74 cm and 33.54 cm width between two pins of the cross hairs. Ground vegetation cover (substrates) were estimated on the plot in four categories to determine the extent to which bare ground, shrubs, herbs and grasses covered the plot. Ground vegetation is equally as important as crown vegetation as it provides animal species with food and habitat (Roberts-Pichette & Gillespie, 1999), provides shelter to woody seedlings and can also compete for available resources (Seifert *et al.*, 2014). The line intercept method (Figure 3.9) was used to calculate the percentage of bare ground and other substrate cover on the plot.

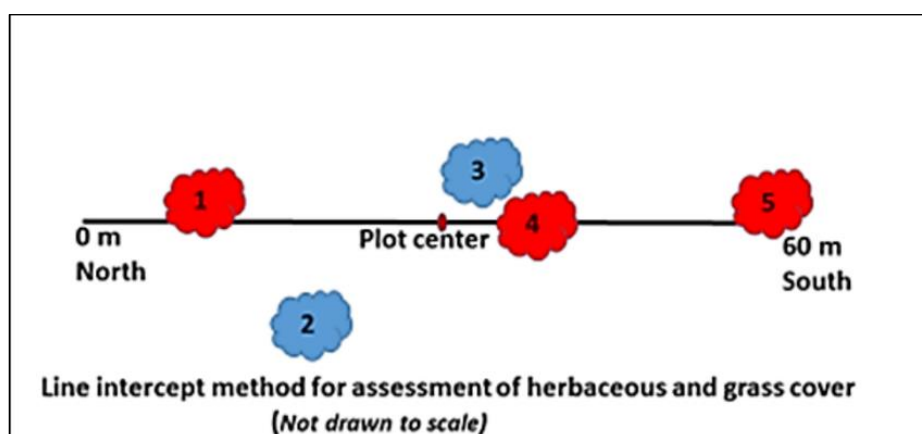


Figure 3.9: Diagrammatic representation of the line intercept method as demonstrated in Kabajani (2016). The substrates numbered 1, 4 and 5 would be counted and their coverage distance recorded while 2 and 3 would be left out.

In this study, members of the Poaceae family relating to natural grasslands, cultivated laws and pasture were recorded as grasses. All substrates of forb in nature were recorded as herbs; all woody species with less than 5 cm dbh were recorded as shrubs; any length of line not covered by any live substrate as described above was recorded as bare ground. There was no species identification for this kind of vegetation assessments.

All woody vegetation with dbh greater than 5 cm were recorded on a tree parameter sheet. Parameters recorded include among others species name, dbh, height, distance from plot centre, tree status, timber or pole quality (visual assessment in classes), log length, crown class (visual assessment), phenology, damage and cause of damage (visual assessment in classes). All trees

were identified to the species level on site and those that could not be identified were later taken to the herbarium for identification. All species encountered in Namibia were identified on site, whereas, in Zambia 7 individuals were taken to the herbarium where they were later identified accordingly. In case of multi-stemmed trees (a tree with two or more stems) were physically counted and all parameters recorded on each individual stem of the tree (Gandiwa, 2011). Figure 3.10 demonstrates possible scenarios and the correct recommended positions where diameter could be measured from.

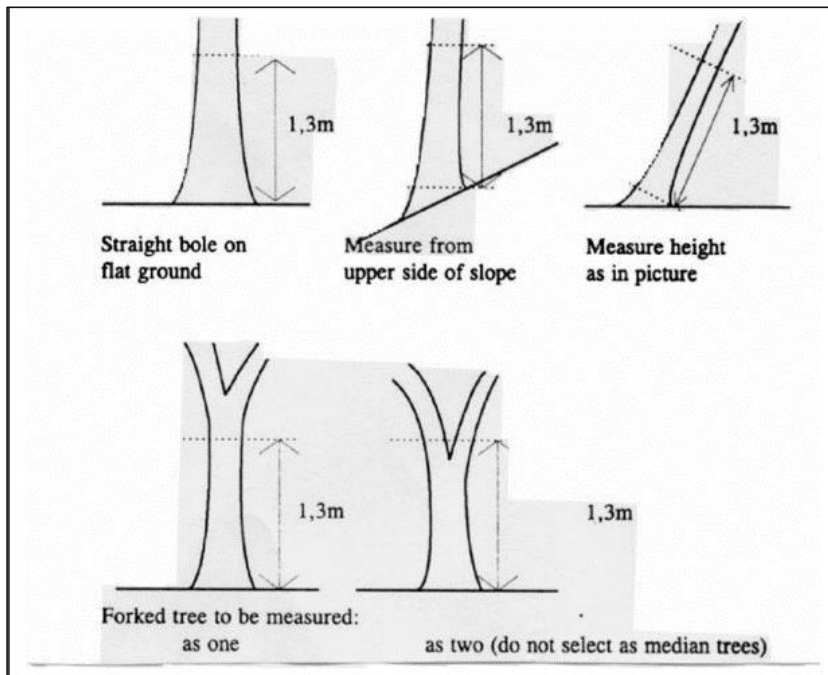


Figure 3.10: An illustration of the correct positions where to measure dbh form for possible scenarios that may be encountered in the field (Burkea *et al.*, 2001).

All woody vegetation with dbh less than 5 cm were recorded on the shrub and regeneration data sheet (Appendix 1.3). Density of individuals were counted and recorded in height classes of 0-50 cm, 51-100 cm, 101-200 cm, 201-300 cm, and greater than 300 cm (Burke *et al.*, 2001). Crown classes were also assessed using the collapsible ruler, categorised as 0-50 cm, 51-100 cm, 101-200 cm, 201-300 cm and greater than 300 cm. Individuals were identified to species level on site. It was only in Sesheke state forest were two individuals could not be identified. Similarly, the unidentified were taken to the herbarium where they could not identify them. The reason of failure to identify them by the herbarium staff was because that the collected leaves and stems were small and not distinct enough to rule several identifications out and not misclassify them. The two individuals were retained and recorded by their local names. Mode of regeneration could not be assessed, as it was practically impossible to undertake the exercise due to the hardness of the soils at the time of sampling and lack of adequate time for such a

mammoth task. Therefore, every regeneration was assumed to be from seedling unless where there was a clear evidence of coppicing. Regeneration was also assumed to concern newly emerged seedlings after the growing season and those that survived one or more growing seasons.

In this study, individuals with dbh less than 5 cm were considered as regeneration, dbh of 5 – 10 cm were regarded as saplings, 10.1 – 20 cm as juvenile trees and dbh greater than 20 cm as canopy trees.

3.9. Data processing and analyses

Data was organised in a spreadsheet for analysis using R statistical software version 3.4.0. (R Core Team, 2017), Minitab statistical software version 17.3.0 (Minitab, 2016) and PAST statistical software version 3.14 (Hammer, 2016).

Statistical analysis was initially by way of linear modelling. This is because linear modelling explains or models the relationship between response (dependent) variable and one or more explanatory (independent) variables (Faraway, 2015). The data violated either one or all the assumptions necessary to validated the results of linear modelling such as normality, homoscedasticity, independence and fixed X (Clewer & Scarisbrick, 2001; Zuur et al., 2009; McDonald, 2014; Faraway, 2015), as is the case with most data sets in ecology and evolution (Bolker *et al.*, 2009). In most cases, data obtained from unbalanced sampling is highly like to violate assumptions of normality (Clewer & Scarisbrick, 2001) and also normal distribution is rarely shown in tree stem densities and basal areas (Vera De Cauwer, pers. comm.). Before accepting results of linear model and making inferences from them, all the assumptions were verified through a model validation process (Zuur *et al.*, 2009) explained as follows;

Normality: assumes that residuals are randomly distributed, fitting a bell-shaped curve (McDonald, 2014). The test for normality was done through the assessment of the diagnostic Q-Q plot of the model in order to check if the residuals follow a straight line. This was further confirmed by plotting the histograms of residuals. Finally, the Shapiro-Wilk test was run in R statistical software to obtain the mathematical p-value. If the assumption is violated, the residuals did not follow a straight line on the Q-Q plot, distribution on the histogram was skewed and the Shapiro-Wilk test would give a $p < 0.05$ at 5% level.

Homoscedasticity: assumes that all the random variables have equal variance (Faraway, 2015; McDonald, 2014). The validation of this assumption was done from the assessment of residuals plotted against fitted values of the model. A random pattern of residuals was expected on either

side of zero for homoscedastic data. However, if the assumption was violated random pattern was missing in the distribution of residuals. This assumption was further confirmed from the Levene test (on the response variable's mean and median) and Bartlett test. Both Levene and Bartlett would produce p-values < 0.05 implying that data lacked homogeneity, hence violating this assumption.

Independence: assumes that the residual value of one observation is not affected by that of another observation (Clewer & Scarisbrick, 2001; Faraway, 2015; McDonald, 2014). Three methods of validating the independence of the data were used; (i) plotting pairs of residuals where they were expected to have a random pattern around zero, (ii) modelling residuals in R statistical software, and (iii) the use of Durbin-Watson test. The plot of residuals from the first method indicated independence in the data and was mathematically confirmed by the last methods with $p > 0.05$. The plotting of residuals against time was not done for this study since data collection was not a time series.

Fixed X: assumes that explanatory variables are deterministic, implying that the values are known in advance at each sample (Zuur *et al.*, 2009). This assumption was not applicable to this study and hence no test for the assumption were carried out.

If the assumptions were found to have been seriously violated during model validation, data sets were transformed. If the assumptions were still violated in the transformed model, generalised linear model (glm) or a non-parametric method was then used. For all statistical significant tests, the interaction of fire treatment (RB & RU) with study site was probed. In addition, the fire frequency on the plots was probed on selected parameters that were assumed to be influenced by fire frequency. These were regeneration density, basal area, vegetation cover, mean diameter and mean height.

3.9.1. Species richness

Species richness is a measure of species encountered in a given taxon per unit area sampled (Magurran, 2004; O'Brien, 1993). Number of species were calculated from the sum of all species recorded. Relative species richness, a measure of one family's contribution to the sample population was calculated using the formula,

$$RS = \left(\frac{\text{Number of a species in a family}}{\text{Total number of species in the sample}} \right) \times 100, \text{ (Wiafe, 2014)}$$

Where; RS is species richness of individual's family relative to other families in the sample.

Species richness and relative species richness were determined across study sites and at individual study site level. In this study, the tree large subfamilies of the Fabaceae were treated as families for purposes of categorisation. These were Mimosoideae, Caesalpinioideae and Papilionoideae families.

3.9.2. Species diversity, dominance, and equitability indices

The diversity, dominance, evenness and equitability of the species were analysed with PAST statistical software version 3.14 (Hammer, 2016). Species diversity is highly important in being an environmental and geographical indicator in combination with other information, and characterises different types of communities (Hammer & Harper, 2006). The analysis focused on non-parametric indices of Shannon (entropy), Simpson's dominance and equitability (Magurran, 2004). Shannon index assess the diversity of the community by taking into account the number of individuals and the number of taxa as well, giving a range of values from zero for a community comprising a single taxon to high values for community comprising many taxa that have few individuals each (Hammer & Harper, 2006). The Shannon indices mostly range between 1.5 and 3.5 (Magurran, 2004) and rarely exceeding more than 5 (Margalef, 1972). Hammer (2016) defines Simpson dominance as an assessment of dominance with a range of values from 0 where there is an equal presence of taxa to 1 in which one taxon is dominating the community completely. Dominance should be stated in relation to Simpson index 1-D, a measure of evenness which ranges between 0 and 1, as they are opposite of each other. Equitability index gives a measure of evenness on the distribution of individuals among the taxa in the community derived from the division of Shannon diversity by the logarithm of the number of taxa (Hammer, 2016). Furthermore, diversity and dominance indices were analysed for statistical significance for effect of fire return interval along a rainfall gradient in addition to comparing of species richness and diversity using diversity curves. Levene's test for homogeneity of variance ($p < 0.011$) indicated unequal variances in the sample. Thus, the unequal variance (Welch) ANOVA version (Hammer, 2016) was used instead.

3.9.3. Similarity index

The multiple-site similarity index and the Sørensen coefficient similarity index were computed to compare species similarity across study sites and between sites (Diserud & Ødegaard, 2007). Sørensen coefficient similarity index is a good and widely used index (Magurran, 2004) but it fails to address the comparisons of more than two sites (Diserud & Ødegaard, 2007) hence, the development of multiple-site similarity index. Adapting from the work of Diserud and

Ødegaard (2007) and Magurran (2004), the two formulae used in the calculation similarity coefficients were;

$$C_s^T = \frac{T}{T-1} \left(\frac{ab + ac + ad + bc + bd + cd - abcd}{a+b+c+d} \right), \text{ multiple similarity index (Diserud \&}$$

Ødegaard, 2007)

$$C_s = \frac{2ab}{a+b}, \text{ between two sites (Magurran, 2004), where;}$$

C_s^T = multiple – site similarity index, C_s = Sørensen coefficient similarity index, T = the number of sites, a = number of species at site A, b = number of species at site B, c = number of species at site C, d = number of species at site D, ab = number of species shared by two sites and abcd = the number of species shared at all four sites. C_s^T will be equal to 1 if sites are found to contain the same species. Thus, C_s^T will decrease with an increase of species observed in or a few sites, and the opposite is true with the increase in number of shared species by several sites (Diserud & Ødegaard, 2007).

3.9.4. Plant community parameters

The stratum biodiversity was analysed across study sites and for individual study sites using vegetation monitoring protocols (Roberts-Pichette & Gillespie, 1999). This is in order to ascertain stand characteristics of selected species in relation to other species in the community. In this study, all tree species at the level of regeneration, shrub, small tree and canopy-tree stratum were combined for the various analyses. Based on Roberts-Pichette & Gillespie (1999), formulae (Appendix 3.2) were used to calculate abundance, frequency, density, and dominance. Thereafter, relative frequency, relative dominance and importance value index were calculated for each selected species' live individuals per respective study site and across study sites.

Abundance was the sum of individual stems of each species found in the total area of the sample. Frequency was the distribution of the species expressed as a percentage of plots in which the species occurred. Density was an average number of individuals of a species per unit area while basal area was the cross section area of tree stems as calculated under Section 3.9.7.

Relative frequency:

$$RF = \left(\frac{\text{Frequency of a species in the sample}}{\text{Total frequency all species in the sample}} \right) \times 100$$

Where; RF in percentage is the distribution of an individual species in a sample relative to the distribution of all species.

Relative density:

$$RD = \left(\frac{\text{Number of a species in the sample}}{\text{Total number of individuals of all species in the sample}} \right) \times 100$$

Where; RD is density of individual species relative to the density of all species.

Relative Dominance:

$$RDom = \left(\frac{\text{Basal area of species in the sample}}{\text{Total basal area of all trees}} \right) \times 100$$

Where; RDom is the area a species occupies relative to the total area occupied by all tree species.

Importance value index (IVI): is a useful tool for profiling the structural role of an individual species in a given stand. Further, the index permits to make comparisons among stands with reference to species composition and stand structure (Roberts-Pichette & Gillespie, 1999).

IVI = Relative density + Relative Dominance + Relative Frequency

3.9.5. Regeneration density

Regeneration densities of woody (tree and shrub) species were computed as number of plants per hectare for all study sites. The densities were calculated per plot then averaged for the study site and for each selected species. These were then analysed graphically and quantitatively for statistical significance. Since data failed to meet the assumption tests, it was initially natural logarithmic (ln) transformed (Clewer & Scarisbrick, 2001; Faraway, 2015; Jayaraman, 1999; McDonald, 2014) and results reported in back transformation with a bias correction (Baskerville, 1972). However, transformation was discarded because count-data regularly perform poorly when transformed (O'Hara & Kotze, 2010). The generalised linear model (glm) with a negative binomial family was then used since the data showed overdispersion (Faraway, 2006; Warton & Hui, 2011). Overdispersion is when the variance is larger than the mean in the data (Bolker *et al.*, 2009; Zuur *et al.*, 2009). Overdispersion for this data was tested by running a code in R statistical software for a model with a Poisson glm, which was then tested for

overdispersion. Phi (ratio of variance to mean) is expected to be around 1 but if it is greater than 1, then it provides evidence of overdispersion. Glm uses a link function to offer a very natural extension of a classical linear model. Classical linear model is obtained by least squares including the tests of significance of the models on comparisons of minimum sums of squares using F-tests. Glm on the other hand is based on deviance of errors and likelihood ratios statistic to test the global significance of the model. It provides a uniform method of estimation and inference equal to the case with normality in error distribution (Venables & Ripley, 2002).

3.9.6. Recruitment pattern

Recruitment patterns for selected species, listed under Section 1.3, were assessed to determine the impact of recent and non-recent fires on recruitment abilities of species from seedling to sapling (5 – 10 cm dbh) and beyond sapling (> 10 cm) level. Two ratios were evaluated; first, the ratio of recruitment from seedling to sapling stage and second the ratio of sapling to juvenile trees and beyond (non-sapling stage). Seedling to sapling ratio was calculated by dividing the seedling density with the density of saplings while the sapling to adult ratio was given by dividing the density of saplings with the density of non-saplings (all size classes aggregated) for each species. The individual stem counts were used in the densities even in cases of multi-stem plants (Gandiwa, 2011). The assessment of recruitment was done across study sites and within individual sites for each of the selected species. Species with recruitment ratio greater than 1 were considered to be recruiting while species with recruitment ratio of less or equal to 1 were considered to have a low recruitment, hence low representation in juvenile size classes (Mwavu & Witkowski, 2009; West et al., 2000).

Recruitment patterns were further assessed by obtaining slopes in ordinary least-squares regression of size classes (Condit et al., 1998; Lykke, 1998; Mwavu & Witkowski, 2009; Sop et al., 2011). Height size classes were divided into 10 units; the first five classes followed the classes for data capture of shrubs and regeneration explained in Section 3.8.3 whereas the last five classes had intervals of 5 m. The classes were, ≤ 0.5 m, 0.51 – 1 m, 1.01 – 2 m, 2.01 – 3 m, 3.01 – 5 m, 5 – 10 m, 10.01 – 15 m, 15.01 – 20 m, 20.01 – 25 m and > 25 m. A total of 1 059 individuals from the selected species (regeneration and tree data) in all the four study sites were analysed for height distribution. Dbh was divided into 12 size classes of 5 cm interval; 5 – 10 cm, 10.1 – 15 cm, 15.1 – 20 cm, 20.1 – 25 cm, 25.1 – 30 cm, 35.1 – 40 cm, 40.1 – 45 cm, 45.1 – 50 cm, 50.1 – 55 cm, 55.1 – 60 cm and > 60 cm. The 0 – 5 cm size class was not measured but the individuals were included as counts for the height size classes. A total of 604 stems (dbh > 5 cm) of selected species from the four sites were analysed for dbh size class distribution

(SCD). This was to make inference on the population structure of the selected species to get an indication of the species that have good recruitment and survival to the next stage (Condit *et al.*, 1998; Lykke, 1998; Mwavu & Witkowski, 2009; Sop *et al.*, 2011). The analysis methods followed the work of these authors where SCD parameters were analysed by calculating ordinary least-squares regression for each species in study sites. The SCD midpoint (m_i) was used as independent variable and the average density (N_i) in each SCD as the dependent variable. These variables were natural logarithmic transformed, $\ln(N_i)$ and $\ln(m_i + 1)$, adding 1 to average density transformation in order to compensate for zero values (Lykke, 1998). The SCD slopes were used for describing population structure of each species (Condit *et al.*, 1998; Lykke, 1998). Negative slopes are indicators of more individuals being found in small size classes (Condit *et al.*, 1998; Mwavu & Witkowski, 2009) as well as an indicator of good rejuvenation and recruitment (Sop *et al.*, 2011). Thus, better recruitment by species is indicated by steep negative slopes (Lykke, 1998; Mwavu & Witkowski, 2009). Flat distribution (slopes closer to zero) are interpreted as a show of a small number of individuals in small and large classes, whereas positive slopes indicate poor recruitment with more individuals in large classes (Condit *et al.*, 1998; Sop *et al.*, 2011). SCD slopes from diameter were used to make inference on population structure and dynamics (Sop *et al.*, 2011; Traoré *et al.*, 2012). Negative height SCD slopes were used as indicators of the species recruitment, ability to survive to the next life stage and stability of juvenile populations while positive SCD slopes are indicators of disturbed and unstable recruitment (Obiri *et al.*, 2002).

3.9.7. Regeneration limitation

Regeneration limitation was assessed using the formula suggested by Muller-Landau *et al.*, (2002);

$$RL = \left(\frac{\text{Total number of subplots where the species was not present}}{\text{Total number of subplots in the sample}} \right) \times 100$$

Where RL is regeneration limitation in percentage.

Regeneration limitations for selected species were assessed across study region and within study sites. It highlighted the percentage of plots on which selected species failed to occur. It equally gives an understanding of species that have hindered regeneration.

3.9.8. Correlation of dbh and height

Diameter and height were assessed for association at different levels; across study sites, RB & RU areas, and species of interest in the respective study sites. This was in order to investigate

three things (McDonald, 2014); firstly, if diameter and height associated with each other, i.e. if increase or decrease in one would affect the increase or decrease in the other. Secondly, was to estimate the strength of the two variables, and thirdly, to find the equation line that would fit the scatter points well. Three models by way of simple linear, exponential and polynomial regressions were assessed for goodness of fit of the data sets. The model by simple linear regression performed better than the other two. Hence, settling for the simple linear regression for the final analysis of diameter and height correlation.

The assumptions for normality, homoscedasticity and independence were validated as described above and the assumptions were not strongly violated (Appendix 4.3). Thereafter, statistical significance tests were carried out using classical simple linear regression regardless of a non-normal distribution was detected. This is because models with independent variables are robust at 0.05 probability level even when assumption of normality is violated (Edgell & Noon, 1984; McDonald, 2014).

3.9.9. Size class distributions

Diameter and height size class distributions (SCD) of selected species were analysed across study sites. Statistical significance analysis was carried out using glm (Bourne *et al.*, n.d.; Faraway, 2006; Warton & Hui, 2011) with a negative binomial distribution (Zeileis *et al.*, 2008) due to overdispersion of data.

Lykke (1998) deemed SCD to be a good indicator of disturbance and successional dynamics in Savannas and tropical dry forests. On the contrary, Condit *et al.* (1998) deemed SCD not to be necessarily a predictor of population change in the forest.

3.9.10. Mean diameter and height

Mean diameter of the trees on the plot were assessed in order to get the idea of the structural parameters in the state forests (Glèlè Kakaï & Sinsin, 2009). The mean diameter was calculated in Minitab statistical software version 17.3.0 (Minitab, 2016) using the formula:

$$D = \sqrt{\frac{1}{n} \sum_{i=1}^n d_i^2} \quad (\text{Glèlè Kakaï \& Sinsin, 2009}), \text{ where;}$$

D is the mean diameter in cm, n is the number of trees on the plot, and d_i is the diameter of the i^{th} tree in cm.

The mean height on each plot was also assessed from the sum of all individual heights, which were then divided by the total number of tree stems (Gandiwa, 2011). The formula used to calculate mean height on the plot in Minitab was;

$$h_r = \frac{1}{n_r} \sum_{i=1}^{n_r} h_i \text{ (Glèlè Kakai \& Sinsin, 2009), where;}$$

h_r is the mean height on the plot, h_i is the height of the i^{th} tree, and n_r is the total number of trees on the plot.

The test of significant differences on mean diameter and mean height were carried out using classical linear modelling for the data that met the assumptions. The data sets that did not meet the assumptions were natural log transformed and if there was still violation of assumptions then non-parametric methods were used on the interaction effect of fire treatment with the sites.

3.9.11. Basal area

Basal area (g) for each tree was calculated using the formula $g = \frac{\pi(\text{dbh})^2}{4}$ for all tree species per plot and selected tree species per plot (Wiafe, 2014), where;

g is basal area in metre squared (m^2), π is 3.142 and dbh is diameter at breast height (m).

Basal area in all plots was then summed up and averaged to estimate for the whole study site after which it was up scaled using the formula $G = \frac{\Sigma g}{A}$ to obtain mean basal area per hectare, where;

G is mean basal area (m^2) per hectare, Σg is mean basal area (m^2) per individual tree and A is the area of the circle in a plot; 0.0314 ha, 0.1257 ha and 0.2827 ha for the 10 m radius, 20 m radius and 30 m radius plots. The factors of 31.821, 7.958 and 3.537 were used to upscale dbh from plot level to a hectare in the respective plots with radius of 10 m, 20 m and 30 m.

The basal area data did not meet the assumptions of normality and homoscedasticity as explained in 3.9. Data showed non-normal distribution even after square root transformation (Clewer & Scarisbrick, 2001; McDonald, 2014). Thus, non-parametric methods were used in the analysis and post – hoc test were done with the Dunn test (Pohlert, 2014) where there was statistical significance difference with the Kruskal- Wallis test. The Mann-Whitney test was used for the effect of fire return interval within study sites.

3.9.12. Canopy cover

The vegetation cover for canopy was analysed in R statistical software 3.4.0 (R Core Team, 2017). Canopy cover was analysed within and between sites taking into account the effect of fire treatment. Cover percentage were calculated as a response variable based on the variable plot method (Cooper, 1957). Data validation in 3.9 showed non-normal distribution. Hence, glm with binomial family was used to analyse canopy cover.

3.9.13. Substrate cover

The substrate cover on the plot was analysed in R statistical software 3.4.0 (R Core Team, 2017). Cover proportions (%) for each substrate (bare ground, grass, herbs, and shrubs) were calculated as a response variable based on the line intercept method (Kabajani, 2016). Each substrate was then analysed in relation to the effect of fire treatment along a rainfall gradient. Glm with binomial family was used to analyse substrate cover since data showed non-normal distribution and the data was proportional (Warton & Hui, 2011).

3.10. Chapter summary

In this chapter, study sites have been described in detail relating to location, vegetation, soil and the prevailing climatic conditions. The study sites were located along a rainfall gradient stretching over 1 897 km from a dry site at Kanovlei in Namibia to a wetter site at Kabompo in Zambia. Systematic random sampling with a 3-plot cluster was used to collect data on stand and tree parameters. The collected data sets were analysed using spreadsheets and statistical software (R, Minitab, and PAST) with various parametric, semi-parametric and non-parametric investigations.

Chapter 4

RESULTS

4.1. Introduction

This chapter provides the results of the data collected from field sampling plots in four study sites described in Section 3.2.2. The results mainly focus on the six selected tree species of interest to this study mentioned in Section 1.3. However, in some instances, a holistic picture is presented before narrowing in on the individual study sites and selected species.

4.2. Floristic composition

4.2.1. Species richness

Floristic composition is rightly described by its species richness, abundance, dominance and frequency of the species (Lamprecht, 1989). A total of 68 vascular plants in the shrub and woody species categories were identified. The identification level included family, genera and species across the study sites between the altitudinal range of 967 m and 1 237 m a.s.l. The identified species belonged to 26 families and 54 genera. In this study, the three large subfamilies of the Fabaceae were treated as families. Table 4.1 presents the species richness and relative species richness across study sites. The dominant families belonged to Caesalpinioideae (15.7%) represented by 11 species in 10 genera, Papilionoideae (12.9%) represented by 9 species in 6 genera, Combretaceae (11.4%) represented by 8 species in 3 genera, Mimosoideae (8.6%) represented by 6 species in 3 genera and Euphorbiaceae (5.7%) represented by 4 species in 4 genera. Two species at Sesheke study site were not identified; however, they are listed by their local names as shown in Appendix 4.1. Caesalpinioideae is the most dominant family in the miombo woodlands (Jew *et al.*, 2016).

The Kanovlei site had 11 species from 11 genera and 8 families. There were 11 species in RB and 9 in RU. The dominant families were Caesalpinioideae (18.2%), Combretaceae (18.2%) and Papilionoideae (18.2%), all contributing 2 species and 2 genera each to the population.

At the Sesheke site, there were 15 families contributing to 19 genera and 21 species in the overall site. In addition, there were two unidentified species that could belong to any of the families and genera. The species richness in RB was 19 and in RU was 17. The dominant families were Papilionoideae (13%), Euphorbiaceae (8.7%), Mimosoideae (8.7%) and Strychnacea (8.7%), each represented by 2 species in 2 genera.

The species richness was 26 species from 23 genera and 13 families at the Caprivi site. The species richness in RB was 17 and 23 in RU. The dominant families included among others Caesalpinioideae (15.4%), Mimosoideae (15.4%), Papilionoideae (15.4%) and Combretaceae (11.5%). The first 3 families were each represented by 4 species in 4 genera while Combretaceae family was represented by 3 species in 3 genera.

The Kabompo site had 17 families and 34 genera represented by 39 species on its overall species richness. The species richness in RB was 32 and in RU was 35. The highest contribution to relative species richness were 23.1 % Caesalpinioideae, 15.4% Papilionoideae, 10.3% Combretaceae and 7.7% Euphorbiaceae. 9 species in 9 genera represented the Caesalpinioideae family, 6 species in 6 genera for Papilionoideae, 4 species in 4 genera for Combretaceae and 3 species in 3 genera for Euphorbiaceae.

Table 4.1: A combined list of species richness and relative species richness for all the four study sites.

Family	No. Of Genera	Species richness	Relative species richness (%)
<i>Caesalpinioideae</i>	10	11	15.7
<i>Papilionoideae</i>	6	9	12.9
<i>Combretaceae</i>	3	8	11.4
<i>Mimosoideae</i>	3	6	8.6
<i>Euphorbiaceae</i>	4	4	5.7
<i>Rubiaceae</i>	4	4	5.7
<i>Annonaceae</i>	3	3	4.3
<i>Strychnaceae</i>	1	3	4.3
<i>Anacardiaceae</i>	2	2	2.9
<i>Apocynaceae</i>	2	2	2.7
<i>Apiaceae</i>	1	1	1.4
<i>Capparaceae</i>	1	1	1.4
<i>Clusiaceae</i>	1	1	1.4
<i>Ebenaceae</i>	1	1	1.4
<i>Melastomataceae</i>	1	1	1.4
<i>Moraceae</i>	1	1	1.4
<i>Ochnaceae</i>	1	1	1.4
<i>Olacaceae</i>	1	1	1.4
<i>Oleaceae</i>	1	1	1.4
<i>Passifloraceae</i>	1	1	1.4
<i>Polygalaceae</i>	1	1	1.4
<i>Proteaceae</i>	1	1	1.4
<i>Rhamnaceae</i>	1	1	1.4
<i>Sapindaceae</i>	1	1	1.4
<i>Sapotaceae</i>	1	1	1.4
<i>Tiliaceae</i>	1	1	1.4
Others	2	2	2.8

4.2.2. Species diversity, dominance and equitability

Results of the pooled data to assess species diversity across study sites in Table 4.2 shows that the study region as a whole was diverse with a high Shannon diversity index of 3.34. The Simpson dominance index (0.05) indicates that there was no single taxon dominating the community but there was uniformity in taxa evenness in the community with a high Simpson_1-D index of 0.95. The distribution of 7 875 individuals among the 70 taxa was also high with an Equitability index of 0.79.

Table 4.2: Species diversity, dominance, evenness and equitability indices on the pooled data showing the lower and upper limits for the entire study region.

Diversity Index	Number	Lower	Upper
Species	70	70	70
Individuals	7 875	7 875	7 875
Dominance_D	0.051	0.049	0.053
Simpson_1-D	0.949	0.947	0.951
Shannon_H	3.339	3.316	3.361
Equitability_J	0.786	0.7804	0.791

The results of species diversity indices in respective study sites are presented in Table 4.3. It is apparent from the table that communities in both RB and RU of respective sites were diverse as indicated by the Shannon indices. The Simpson dominance and Simpson evenness indices both show that there was little dominance and higher evenness of taxa in the respective communities except for RB in Caprivi site. The equitability index indicated good distribution of individuals among the respective taxa per community.

Table 4.3: Species diversity, dominance, evenness and equitability indices for the fire treatments across study sites.

Diversity Index	Kanovlei		Sesheke		Caprivi		Kabompo	
	RB	RU	RB	RU	RB	RU	RB	RU
Species	11	9	19	17	17	23	32	35
Individuals	395	407	1 706	859	381	639	1 586	1 902
Dominance_D	0.199	0.173	0.152	0.188	0.428	0.199	0.105	0.082
Simpson_1-D	0.801	0.827	0.848	0.812	0.572	0.801	0.896	0.918
Shannon_H	1.878	1.935	2.192	2.154	1.511	2.02	2.643	2.841
Equitability_J	0.783	0.881	0.744	0.760	0.533	0.644	0.763	0.799

The statistical test on differences of diversity indices using the Welch Anova among the communities proved that the differences were non-significant ($p=0.399$, $df=54$).

4.2.3. Species similarity measure

The multiple-site similarity coefficient of 0.51 indicated some level of similarity across the study sites. The coefficient was not high enough to show strong similarity among study sites meaning that the sites had as much of shared species as the rare species.

Similarity comparisons between any two sites revealed low Sørensen coefficient similarity indices. Species similarity between Kanovlei and Sesheke had Sørensen coefficient similarity index of 0.18, Kanovlei and Caprivi had 0.54, Kanovlei and Kabompo had 0.28, Sesheke and Caprivi had 0.24, Sesheke and Kabompo had 0.13 while Caprivi and Kabompo had 0.28. The results indicate that only Kanovlei and Caprivi exhibited more species observed at both sites. The other between site similarity comparisons showed that there were more species observed at each site than those observed at both sites under comparison. However, intra site similarity comparisons showed that RB and RU were very similar. The species similarity indices were 0.80 at Kanovlei, 0.67 at Sesheke, 0.70 at Caprivi and 0.84 at Kabompo.

4.2.4. Plant community parameters

This section presents results on the assessment of species community parameters focusing on living individuals for selected tree species based on abundance, relative frequency, relative density, relative dominance and importance value index.

The species with most abundant stems of the six selected species across study sites was *B. plurijuga* followed by *B. africana* while *S. rautanenii* had the least abundant 76 stems. The selected species accounted for 1 109 stems out of the 7 875 stems recorded across study sites. Abundance of selected species in respective study sites differed highly. Figure 4.1 outlines the species with with the least abundant stems across the study sites and at the respective study sites.

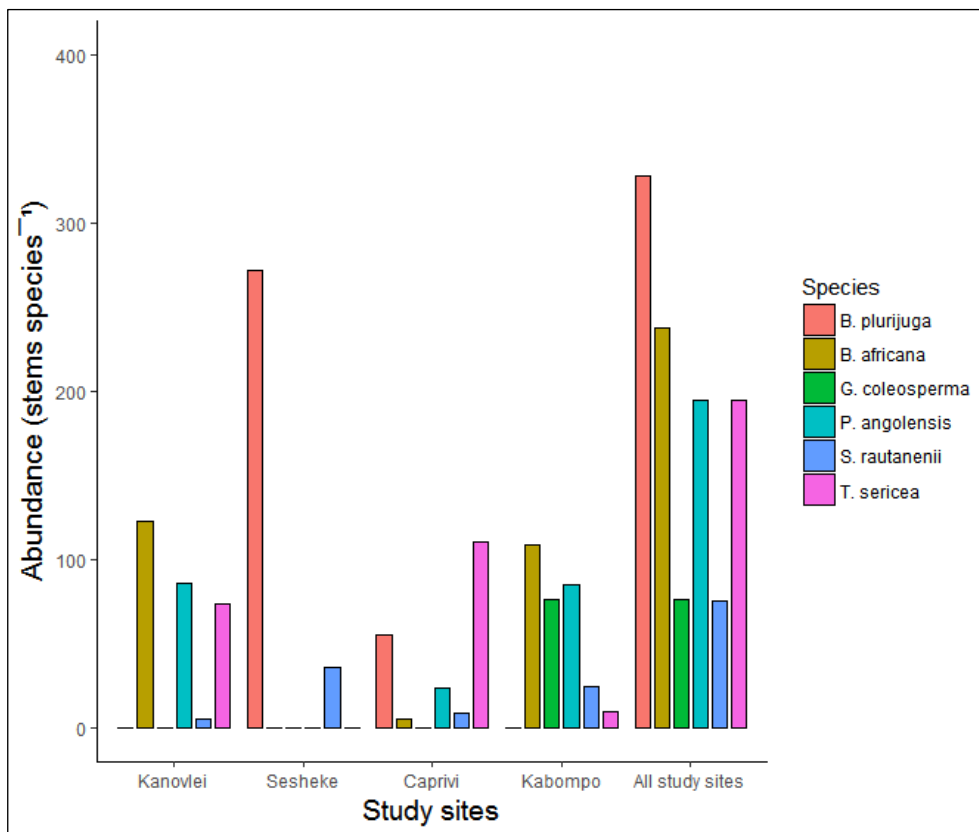


Figure 4.1: The abundance of the selected tree species across the study sites.

Distribution of selected species in relation to other species across study sites ranked *P. angolensis* to have the highest relative frequency followed by *B. africana* and *T. sericea*. Other species, non-selected for this study, shared the highest relative frequency percentage within study sites and the whole study region. Table 4.4 gives a summary of relative frequency in the study sites and whole study region.

Table 4.4: Summary of the relative frequency of selected species across study sites.

Species	Relative Frequency (%)				
	Kanovlei	Sesheke	Caprivi	Kabompo	All sites
<i>P. angolensis</i>	13.4	0	4.4	5.2	5.6
<i>B. africana</i>	11.2	0	2.2	5.8	5.1
<i>T. sericea</i>	14.2	0	9.9	0.3	4.3
<i>B. plurijuga</i>	0	12.3	8.8	0	3.8
<i>S. rautanenii</i>	1.5	1.4	2.2	2.6	2.1
<i>G. coleosperma</i>	0	0	0	3.5	1.6
Others	59.7	86.3	81.3	82.6	77.5

The relative density of each selected species across study sites indicated that *B. plurijuga*, *B. africana* and *P. angolensis* were ranked in the first 3 positions (Table 4.5). Relative frequency matched with abundance in Table 4.5 in ranking of selected species across study sites.

Table 4.5: Summary of the relative density of selected species across study sites.

Species	Relative Density (%)				
	Kanovlei	Sesheke	Caprivi	Kabompo	All sites
<i>B. plurijuga</i>	0	10.6	5.5	0	4.17
<i>B. africana</i>	15.3	0	0.6	3.1	3.02
<i>P. angolensis</i>	10.7	0	2.4	2.4	2.48
<i>T. sericea</i>	9.2	0	10.9	0.3	2.48
<i>G. coleosperma</i>	0	0	0	2.2	0.98
<i>S. rautanenii</i>	0.7	1.4	0.9	0.7	0.97
Others	64.1	88	79.7	91.3	85.9

Results in Table 4.6 reveal *B. plurijuga* to dominate across study sites followed by *P. angolensis* and *B. africana*. The 3 species together account for 51.5% of relative dominance across study sites. *T. sericea* had the least percentage of relative dominance because it is a smaller type of

tree found in lower tree layers and not the typical canopy tree hence, the species is highly limited in transcending regeneration stage into sapling, eventually into juvenile and adult stages.

The summary in the table shows the level of dominance by selected species in their respective study sites. *B. plurijuga* showed dominance in Sesheke and Caprivi sites whereas *P. angolensis* showed dominance in Kanovlei site. Kabompo site was dominated by species that were not part of the selected species to the study. These species shared the highest relative dominance percentage followed by *G. coleosperma* among the selected species in the study site.

Table 4.6: Summary of the relative dominance of selected species across study sites.

Species	Relative dominance (%)				
	Kanovlei	Sesheke	Caprivi	Kabompo	All sites
<i>B. plurijuga</i>	0	75.3	57.2	0	26.5
<i>P. angolensis</i>	54.1	0	6.3	3.7	9.2
<i>B. africana</i>	35.7	0	0	9.0	6.9
<i>G. coleosperma</i>	0	0	0	11.6	2.4
<i>S. rautanenii</i>	0	2.8	6.5	0.3	1.7
<i>T. sericea</i>	0	0	1.1	0	0.8
Others	10.2	21.9	28.9	75.4	52.5

The importance value index (IVI) profiled the structural role of selected species across study sites ranking *B. plurijuga* as the most important valued tree species. The ranking of selected tree species across study sites according to IVI values followed the order of *B. plurijuga*, *P. angolensis*, *B. africana*, *T. sericea*, *G. coleosperma* and *S. rautanenii* as shown in Table 4.7.

What stands out in Table 4.7 is that different selected species are ranked differently at each of the study sites based on species composition and stand structure as evidenced by their IVI. Sesheke site had the highest number of selected species recording 0% IVI values whereas the other sites had only 1 or 2 of the selected species with 0% IVI.

Table 4.7: Summary of the importance value index of the selected species across study sites.

Species	Importance Value Index %				
	Kanovlei	Sesheke	Caprivi	Kabompo	All sites
<i>B. plurijuga</i>	0	98.3	71.5	0	44.8
<i>P. angolensis</i>	78.3	0	13.1	11.3	15.9
<i>B. africana</i>	62.2	0	2.8	17.9	14.9
<i>T. sericea</i>	23.4	0	21.9	0.6	7.0
<i>G. coleosperma</i>	0	0	0	17.3	6.8
<i>S. rautanenii</i>	2.2	2.8	9.6	3.6	4.5

4.3. Density and composition of natural regeneration

4.3.1. Regeneration density for all species across study sites

Regeneration density across study sites for all species showed significant differences among sites in RB and RU as shown in Figure 4.2.

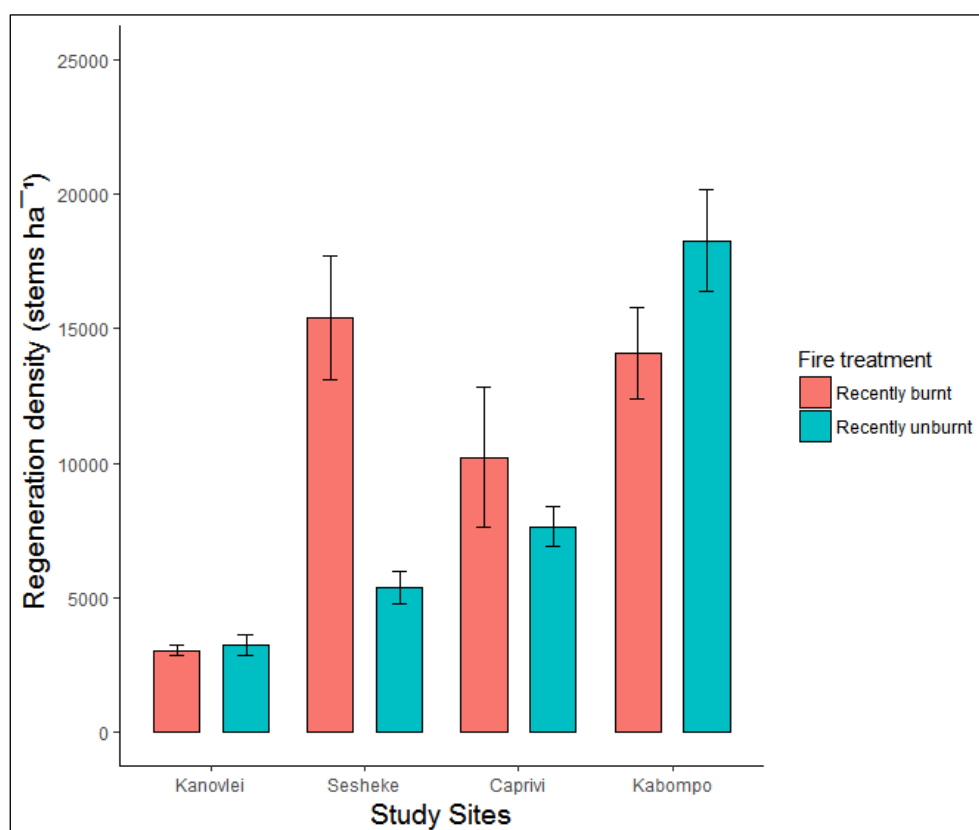


Figure 4.2: Natural regeneration density of all tree and shrub species in the recently burnt and recently unburnt plots across study sites.

The fire treatment influenced natural regeneration across study sites which differed significantly ($p < 0.0001$, $df=3$). There were no significant differences within Kanovlei and Caprivi sites. The regeneration density, with standard error in parenthesis, at Kanovlei was 3 039 (± 191) individuals per hectare in RB and 3 266 (± 383) individuals per hectare in RU. At the Sesheke site, regeneration density was 15 419 ($\pm 2 287$) individuals per hectare in RB and 5 387 (± 628) individuals per hectare in RU whereas at the Caprivi site regeneration density was 10 231 ($\pm 2 625$) in RB and 7 641 (± 743) individuals per hectare in RU. The Kabompo site had regeneration density of 14 097 ($\pm 1 685$) individuals per hectare in RB and 18 274 ($\pm 1 883$) individuals per hectare in RU.

Figure 4.3 demonstrates the effect of fire treatment on natural regeneration along the rainfall gradient. The interaction plot revealed that there was no significant effect of fire treatment at the Kanovlei study site (MAP 551 mm). The Sesheke (MAP 633 mm) and Caprivi (MAP 741 mm) study sites showed that natural regeneration increased with recent fire in RB plots and decreased with non-recent fire in RU treatments. An opposite effect on natural regeneration density was observed at the Kabompo study site (MAP 1 024 mm) to that of the Sesheke and Caprivi sites. Regeneration density in Kabompo site increased with non-recent fire and showed reduction with recent fire.

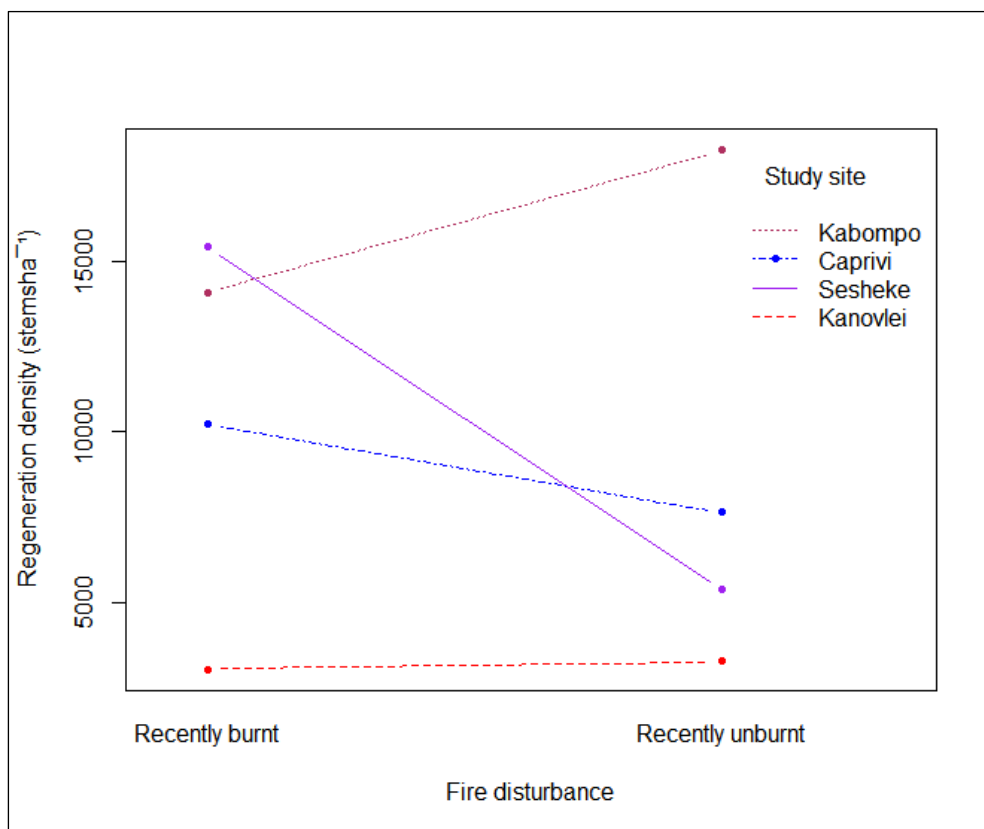


Figure 4.3: An interaction plot demonstrating the interaction effect of fire treatment with study site on natural regeneration density.

The natural regeneration density of all woody (tree and shrub) species across study sites was further revealed to be affected by the fire frequency on the plot. The interaction of fire frequency with site was statistically significant ($p= 0.027$, $df=4$). Figure 4.4 shows natural regeneration density function of fire frequency per site.

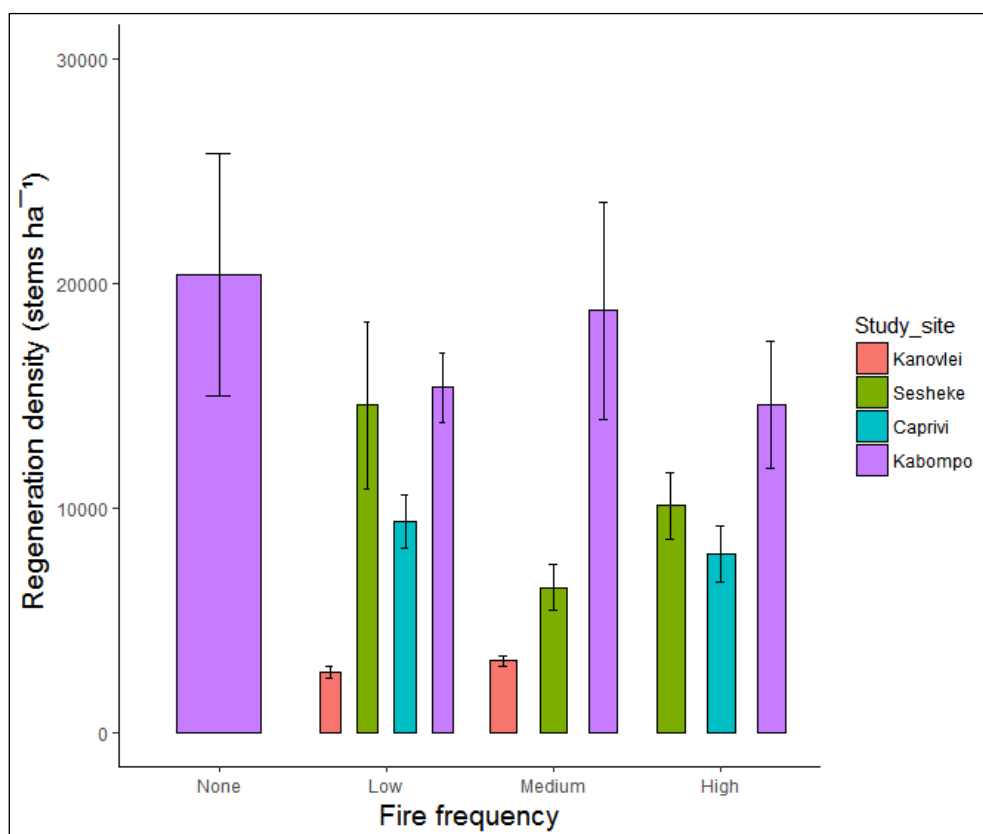


Figure 4.4: Natural regeneration density of all tree and shrub species due to interactive effect of fire frequency on the plot with site. The fire frequencies correspond to 0 burn (none), 1 – 3 burns (low), 4 – 6 burns (medium) and > 6 burns (high) during the period 2001 - 2015.

4.3.2. Regeneration density for selected species

Mean densities per hectare for each of the selected species were calculated in each of the study sites in order to get an idea of natural regeneration response to fire treatment.

Regeneration of *B. plurijuga* was only present at the Sesheke and Caprivi sites (Figure 4.5). Test of statistical significance showed that there was no significant interaction effect between fire treatment and site ($p=0.77$, $df=3$). Thus, each factor was assessed individually in the absence of the interaction effect. Regeneration density of *B. plurijuga* differed significantly ($p < 0.0001$) between sites and among RB and RU ($p= 0.003$). Mean regeneration density for Sesheke site was 289 (± 289) individuals per hectare in RB and 78 (± 67) individuals per hectare in RU whereas at Caprivi was 333 (± 112) individuals per hectare in RB and 29 (± 29) individuals per hectare in RU.

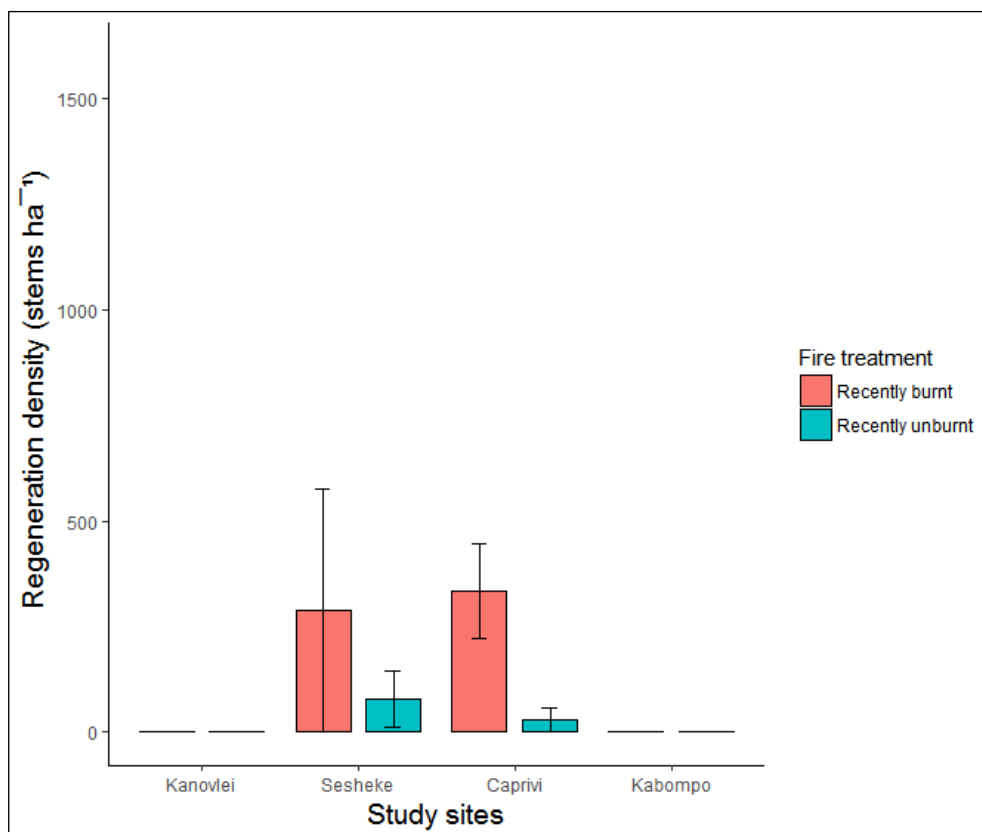


Figure 4.5: The fire treatment effect on natural regeneration density of *Baikiaea plurijuga* at Sesheke and Caprivi state forests.

Regeneration density of *B. africana* was present at three of the four sites (Figure 4.6). The mean densities at the Kanovlei site was 80 (± 37) individuals per hectare in RB and 122 (± 54) individuals per hectare in RU; Caprivi of 167 (± 109) individuals per hectare in RB and 14 (± 14) in RU; and Kabompo site of 311 (± 108) individuals per hectare in RB and 422 (± 144) individuals per hectare in RU. The interaction of fire with site had no statistically significant effect ($p = 0.128$, $df=3$) on regeneration densities of *B. africana*. Regeneration density at the Kabompo site differed significantly from that of the Caprivi site (baseline) and Kanovlei site ($p < 0.0001$), while fire treatment had a significant effect on regeneration density of the Caprivi state forest ($p=0.032$) and not at the other two sites.

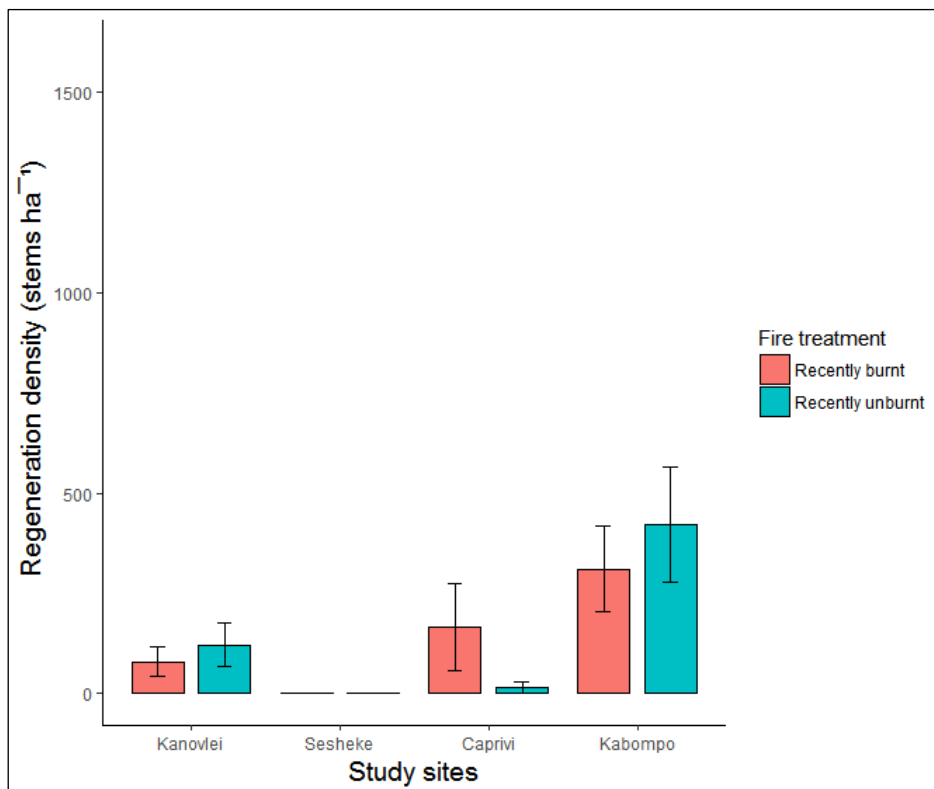


Figure 4.6: The fire treatment effect on natural regeneration density of *Burkea africana* at Kanovlei, Caprivi and Kabompo state forests.

Regeneration and mother trees of *G. coleosperma* lacked in all study sites except at Kabompo state forest (Figure 4.7). The mean regeneration density was 0 in RB and 578 (± 578) individuals per hectare in RU. There was no evidence of statistically significant differences on regeneration density due to interaction effect, in RB and RU, and among study sites as all the p-values were > 0.05 .

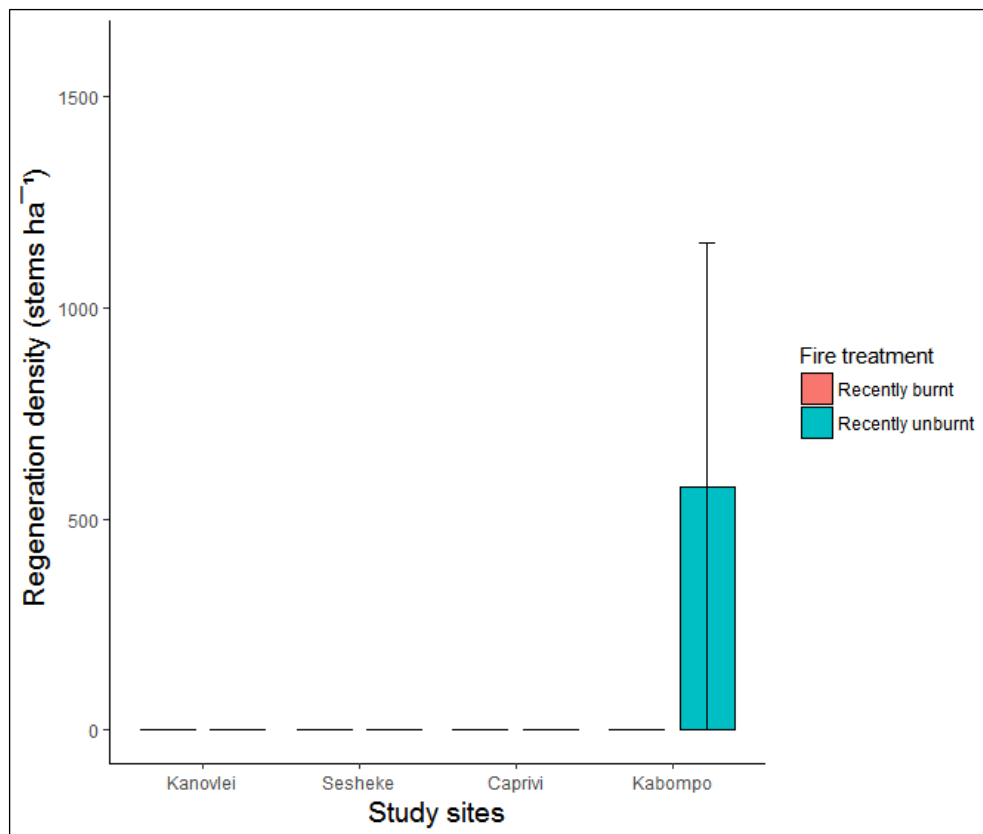


Figure 4.7: The fire treatment effect on natural regeneration density of *Guibourtia coleosperma* at Kabompo state forest.

Regeneration of *P. angolensis* was only present at the Caprivi and Kabompo sites. Figure 4.8 shows regeneration densities which were 0 in RB and 14 (± 14) individuals per hectare in RU for Caprivi whereas in Kabompo they were at 422 (± 224) individuals per hectare in RB and 244 (± 86) individuals per hectare in RU. The interaction effect of fire with site on regeneration density was not statistically different ($p = 0.854$) across study sites. Hence, individual factors were assessed for effects on regeneration density. Regeneration densities of the Kabompo and Caprivi sites were significantly different ($p < 0.0001$), while the effect of fire treatment was not significantly different ($p = 0.477$) in either of the study site.

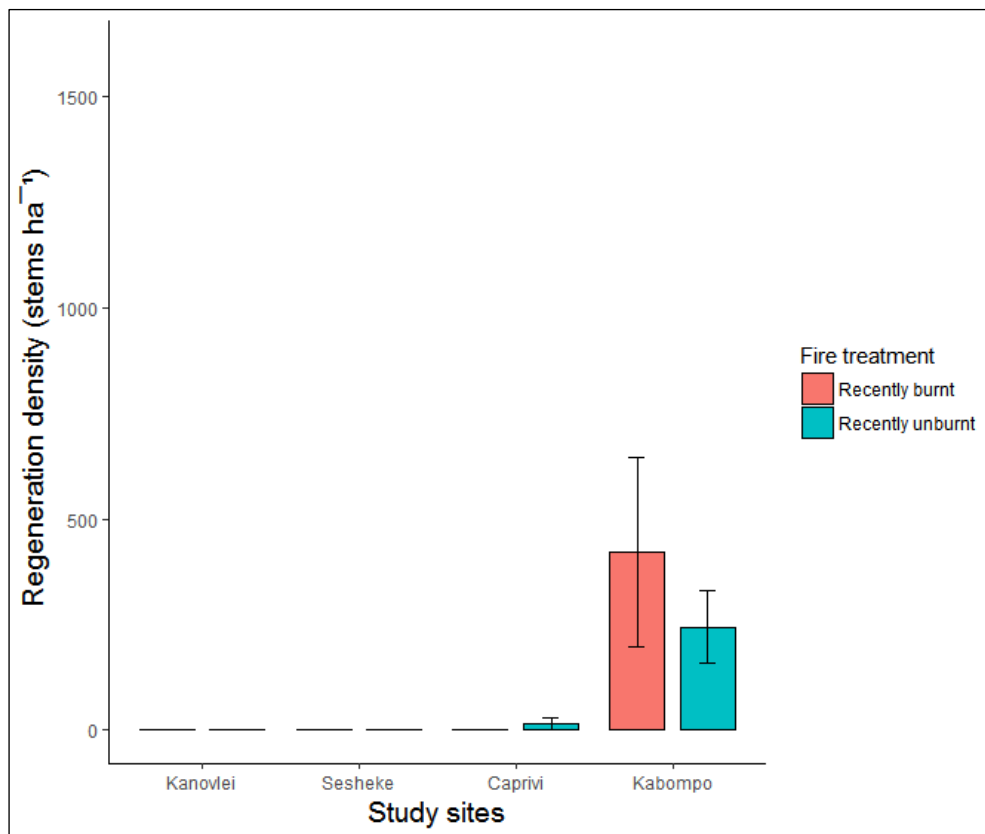


Figure 4.8: The fire treatment effect on natural regeneration density of *Pterocarpus angolensis* at Caprivi and Kabompo state forests.

Mean regeneration density of *S. rautanenii* ranged from 0 to 400 individuals per hectare across study sites. The species recorded presence (Figure 4.9) only in RB in all study sites. The Kanovlei site had 10 (\pm 10) individuals per hectare, the Sesheke site recorded 400 (\pm 311) individuals per hectare, the Caprivi site recorded 33 (\pm 33) individuals per hectare and Kabompo recorded 211 (\pm 85) individuals per hectare. There was no statistically significant effect due to interaction of fire with site ($p > 0.05$) on regeneration density. Regeneration densities were significantly different ($p < 0.001$) for the Kanovlei, Sesheke and Kabompo sites compared to that of the Caprivi site, which was the baseline. The fire treatment had no significant effect ($p = 0.254$) within study sites.

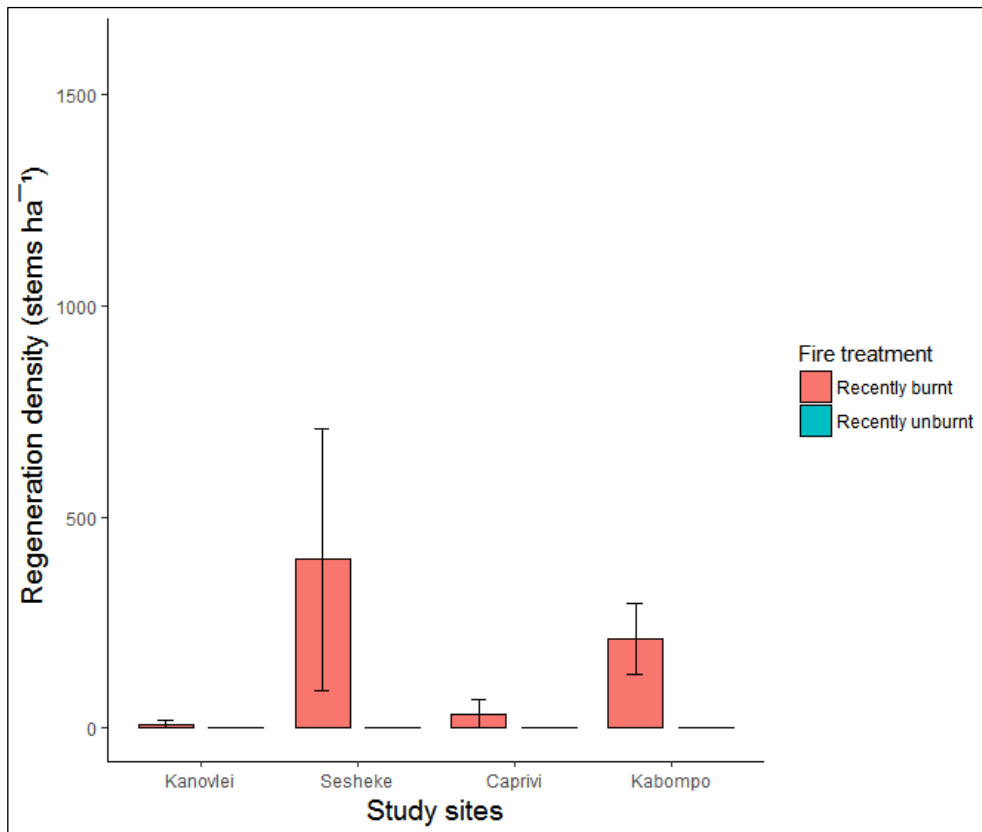


Figure 4.9: The fire treatment effect on natural regeneration density of *Schinziophyton rautanenii* across the study sites.

Regeneration of *T. sericea* was recorded at both the Kanovlei and Caprivi sites and absent at Sesheke and Kabompo sites (Figure 4.10). The mean regeneration density at the Kanovlei site was 360 (± 91) individuals per hectare in RB and 411 (± 89) individuals per hectare in RU while at the Caprivi site was 500 (± 198) individuals per hectare in RB and 1 143 (± 388) individuals per hectare in RU. The interaction effect of fire with site was not statistically significant ($p = 0.513$) on the mean densities. However, mean regeneration density at the Kanovlei site ($p < 0.0001$) was significantly different from the baseline at Caprivi. The effect of fire treatment on regeneration density at the Caprivi site was statistically different ($p = 0.013$) between RU and RB but not at the Kanovlei site.

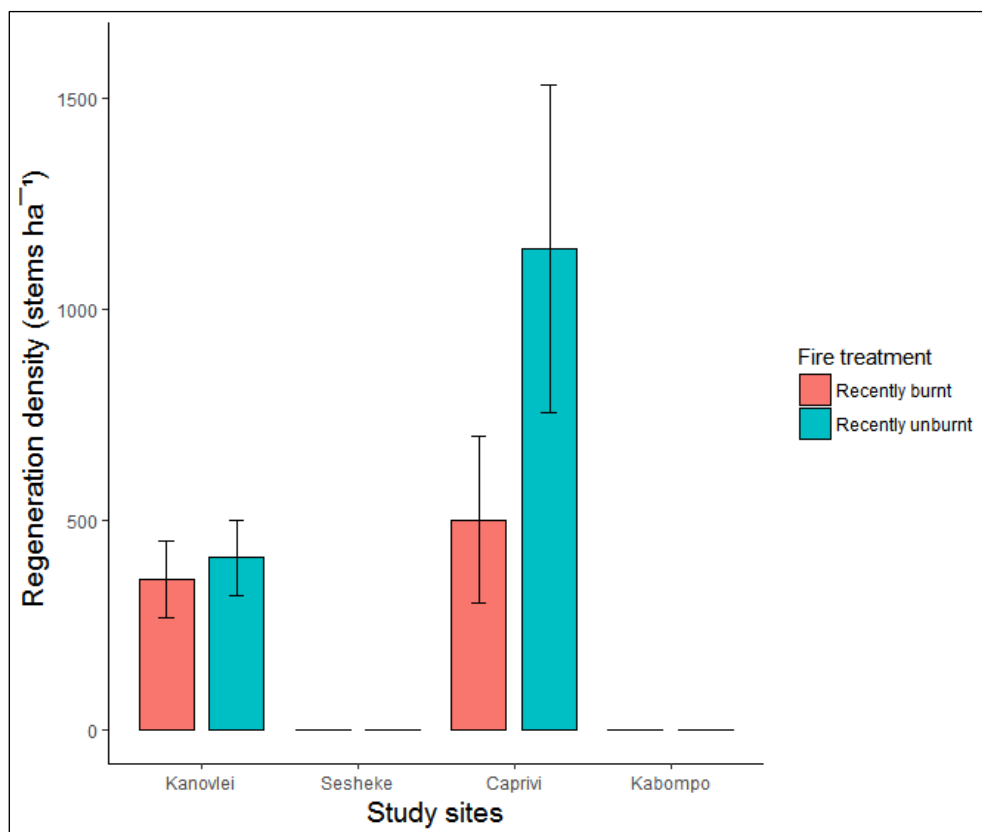


Figure 4.10: The fire treatment effect on natural regeneration density of *Terminalia sericea* at Kanovlei and Caprivi state forests.

Further probing of the effect of the interaction of the fire frequency with species revealed that the effect was statistically significant ($p < 0.0001$, $df = 15$) on the natural regeneration density of the selected species across the study sites. Figure 4.11 displays the natural regeneration densities of selected species, according to fire frequency categories.

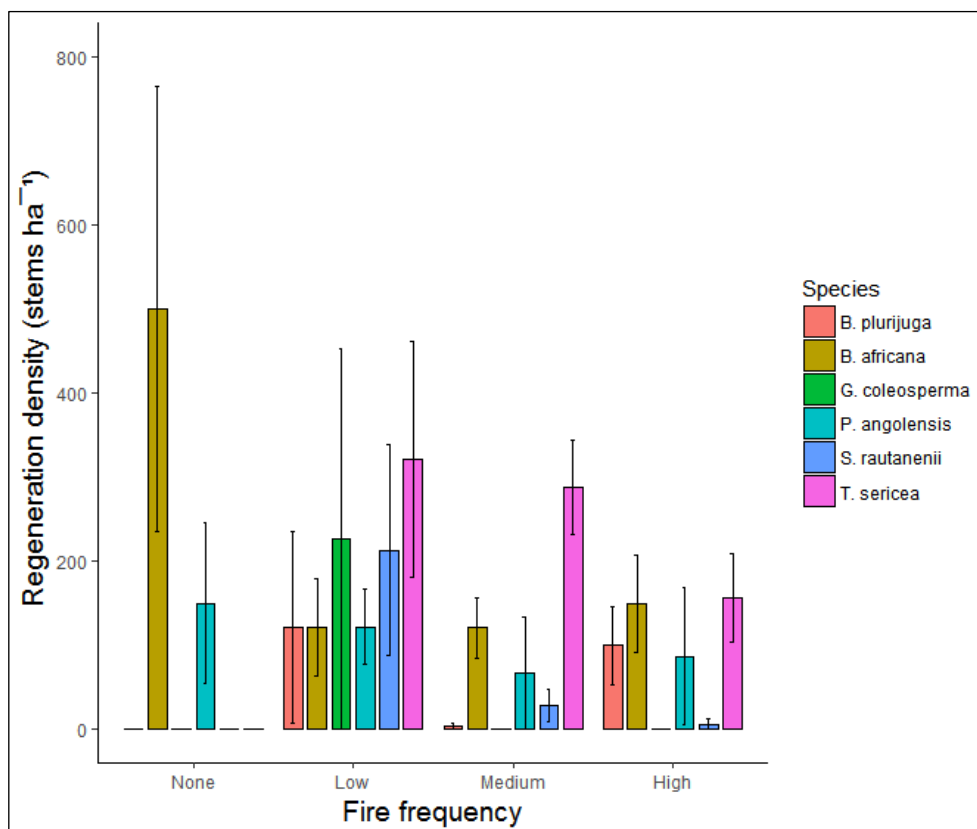


Figure 4.11: The effect of fire frequency on the natural regeneration density of selected species across study sites were highly significant according to fire frequency categories.

4.3.3. Recruitment pattern of selected species

Seedling to sapling ratio in Table 4.8 showed the selected species that were recruiting and those that had low recruitment in the community while sapling to adult ratio showed the selected species are surviving regeneration to juvenile and adult stages. Results of size class distributions (SCD) slopes for species recruitment, survival ability and stability of juvenile populations are given in Table 4.9. The results are further referred to under Section 4.4.4 and 4.4.5, with graphical illustrations on height and diameter distributions for the selected tree species.

Seedling to sapling ratios > 1 and negative height SCD slopes are indicative of species that were recruiting while < 1 and positive slopes indicated species that have hindered recruitment. In addition, species that have good survival ability and stability of juvenile populations were indicated by sapling to adult ratio greater > 1 and negative diameter SCD slopes, while ratios < 1 and positive slopes indicated low species survival to the next life stage and unstable juvenile populations (West *et al.*, 2000; Mwavu & Witkowski, 2009).

Table 4.8: Indicators of species recruitment (seedlings to sapling ratio) and species survival to the next life stage (sapling to adult ratio).

Species	Study site	Number of individuals/hectare			Recruitment ratios	
		Seedlings	Saplings	Adults	Seedling: Sapling ratio	Sapling: Adult ratio
<i>Baikiaea plurijuga</i>	Kanovlei	0	0	0	0	0
	Sesheke	366.7	0.7	44.8	523.9	0.02
	Caprivi	240	0.7	15	342.9	0.1
	Kabompo	0	0	0	0	0
<i>Burkea africana</i>	Kanovlei	200	2.9	15.8	69.0	0.2
	Sesheke	0	0	0	0	0
	Caprivi	120	0	0	0	0
	Kabompo	700	0.8	0.1	875	8
<i>Guibourtia coleosperma</i>	Kanovlei	0	0	0	0	0
	Sesheke	0	0	0	0	0
	Caprivi	0	0	0	0	0
	Kabompo	577.7	0.3	0.1	1 925.7	3
<i>Pterocarpus angolensis</i>	Kanovlei	0	0.6	15.8	0	0.04
	Sesheke	0	0	0	0	0
	Caprivi	20	1.9	6.5	10.5	0.3
	Kabompo	666.6	1.6	0.1	416.6	16
<i>Schinziophyton rautanenii</i>	Kanovlei	10.5	0.4	0.6	26.3	0.6
	Sesheke	399.9	0	0	0	0
	Caprivi	20	0	3.6	0	0
	Kabompo	211.1	0.2	0	1 055	0
<i>Terminalia sericea</i>	Kanovlei	757.9	0.2	0	3 789.5	0
	Sesheke	0	0	0	0	0
	Caprivi	1899	2.9	2.2	654.8	1.3
	Kabompo	0	0	0	0	0

Table 4.9: The different size class distribution ordinary least-squares regression slopes of selected species recruitment, survival ability to the next life stage and stability of juvenile populations across the study sites.

Species	Study site	Height SCD				Diameter SCD			
		Slope	SE	R ²	P-value	Slope	SE	R ²	P-value
<i>Baikiaea plurijuga</i>	Sesheke	0.121	1.232	0.026	0.659	0.382	0.516	0.204	0.141
	Caprivi	-0.146	0.829	0.078	0.435	0.051	0.169	0.040	0.557
<i>Burkea africana</i>	Kanovlei	-0.046	0.902	0.007	0.817	-0.734	0.612	0.402	0.027
	Caprivi	-0.128	0.356	0.260	0.133	n.a	n.a	n.a	n.a
	Kabompo	-0.330	0.481	0.562	0.013	-0.271	0.380	0.191	0.155
<i>Guibourtia coleosperma</i>	Kabompo	-0.120	0.809	0.057	0.507	0.064	0.280	0.024	0.631
<i>Pterocarpus angolensis</i>	Kanovlei	0.102	0.922	0.032	0.619	-0.325	0.456	0.191	0.155
	Caprivi	0.045	0.664	0.012	0.759	-0.644	0.192	0.841	<0.0001
	Kabompo	-0.321	0.388	0.651	0.005	-0.383	0.244	0.535	0.007
<i>Schinziophyton rautanenii</i>	Kanovlei	-0.003	0.232	0.000	0.955	n.a	n.a	n.a	n.a
	Sesheke	0.121	1.232	0.025	0.659	n.a	n.a	n.a	n.a
	Caprivi	0.053	0.358	0.056	0.511	0.129	0.317	0.072	0.399
	Kabompo	-0.139	0.574	0.137	0.292	n.a	n.a	n.a	n.a
<i>Terminalia sericea</i>	Kanovlei	-0.003	0.232	0.000	0.955	n.a	n.a	n.a	n.a
	Caprivi	-0.631	0.631	0.731	0.002	-0.611	0.296	0.666	0.001
	Kabompo	-0.139	0.574	0.137	0.292	n.a	n.a	n.a	n.a

4.3.4. Regeneration limitation

Table 4.10 compares overall regeneration limitation of selected species calculated from aggregated counts across study sites, which ranged from 63.9% to 99.2%. What stands out in the table is that there was more than 60% regeneration limitation of the selected species across the study region. *T. sericea* displayed the least regeneration limitation at 63.8% while *G. coleosperma* had the highest at 99.2%. In RB plots, *G. coleosperma* did not occur in any of them while *S. rautanenii* did not occur in RU plots across the study sites. The Mann-Whitney test showed significantly ($P=0.021$) different regeneration limitation of *S. rautanenii* among RB and RU across study sites. The regeneration limitation for rest of selected tree species were non-significantly different among RB and RU across the study sites.

Table 4.10: Distribution of regeneration limitation of selected species across study sites

Species	Regeneration limitation (%)		
	Recently burnt	Recently unburnt	Site overall
<i>Guibourtia coleosperma</i>	100	98.5	99.2
<i>Baikiaea plurijuga</i>	90.3	97.1	93.8
<i>Schinziophyton rautanenii</i>	83.9	100	92.3
<i>Pterocarpus angolensis</i>	91.9	86.8	89.2
<i>Burkea africana</i>	75.8	73.5	74.6
<i>Terminalia sericea</i>	66.1	61.8	63.8

Regeneration limitation of selected species within study sites ranged from 14.3% for *T. sericea* at Caprivi to 100% for some selected species at any of the study sites (see Appendix 4.2).

B. plurijuga had a regeneration limitation of 100% in RB, RU and overall plots, at the Kanovlei and Kabompo sites. The Caprivi site had the least limitation in RB plots (16.7%) and overall (70%) whereas the Sesheke site had a uniform limitation at 94.4% in RB, RU and overall plots.

B. africana showed maximum limitation (100%) at Sesheke site. The least limitation was in Kabompo with a mean of 50% for all plots, 55.6% in RB and 44.4% in RU. Kanovlei had a mean regeneration limitation of 68.4%, 75% in RB and 61.1% in RU while overall regeneration limitation at Caprivi was at 85%, 66.7% in RB and 92.9% in RU.

G. coleosperma recorded maximum regeneration limitation in all plots across the study region except in the RU plots of the Kabompo site at 94.4%.

Maximum regeneration limitation for *P. angolensis* was recorded at the Kanovlei and Sesheke sites on all plots whereas the Caprivi site recorded maximum limitation in RB plots. The least limitation was at the Kabompo site with 72.2% in RB, 55.6% in RU and 63.9% in all plots while Caprivi recorded 92.9% in RU and 95% in all plots.

Regeneration presence for *S. rautanenii* was recorded in RB plots while RU plots recorded maximum regeneration limitation in all four-study sites. Kanovlei registered a regeneration limitation of 95% in RB and 97.4% overall, Sesheke had 88.9% in RB plots and 94.4% overall, Caprivi had 83.3% in RB and 95% overall whereas in Kabompo 66.7% was in RB and 83.3% overall.

Regeneration limitation for *T. sericea* was least in Caprivi, followed by Kanovlei and then Kabompo. Caprivi recorded 33.3% in RB, 14.3% in RU and 20% overall while Kanovlei

recorded 20% in RB, 22.2% in RU and 21.1% overall. Sesheke recorded a maximum limitation in all plots whereas Kabompo had 94.4% in RB, 100% in RU and 97.2% overall.

Table 4.11 displays the results of the five tree species dominating natural regeneration composition in both RB and RU within respective study sites. The contribution of tree species to the total regeneration density ranged from 26 to 92.5% between the RB and RU across the study sites. At Kanovlei site, the contributions of the tree species were 34.3 % in RB, 45.4% in RB and 40.1% overall; at Sesheke site 26% in RB, 47.4% in RU and 32.2% overall; at Caprivi site 92.5% in RB, 50.3% in RU and 74.5% overall, while at Kabompo it was 52.2% in RB, 56.5% in RU and 54.6% overall.

Table 4.11: The five woody species dominating regeneration compositions across study sites

Kanovlei study site				Sesheke study site			Caprivi study site			Kabompo study site		
Recently burnt												
S/N	Species	#ha ⁻¹	RL (%)	Species	#ha ⁻¹	RL (%)	Species	#ha ⁻¹	RL (%)	Species	#ha ⁻¹	RL (%)
1	<i>Combretum species</i>	400	10	<i>Pseudolachnostylis maprouneifolia</i>	1 866	50	<i>Combretum species</i>	7 732	0	<i>Baphia bequaertii</i>	2 765	0
2	<i>Terminalia sericea</i>	360	20	<i>Schinziophyton rautanenii</i>	400	88.9	<i>Terminalia sericea</i>	500	33.3	<i>Diplorhynchus condylocarpon</i>	2 699	16.7
3	<i>Ochna pulchra</i>	200	45	<i>Pterocarpus lucens</i>	311	53.3	<i>Baikiaea plurijuga</i>	333	16.7	<i>Ochna pulchra</i>	1 410	44.4
4	<i>Burkea africana</i>	80	75	<i>Baikiaea plurijuga</i>	289	94.4	<i>Ochna pulchra</i>	200	33.3	<i>Combretum molle</i>	555	50
5	<i>Schinziophyton rautanenii</i>	10	95	<i>Schrebera trichoclada</i>	244	77.8	<i>Burkea africana</i>	167	66.7	<i>Julbernardia paniculata</i>	478	66.7
Recently unburnt												
1	<i>Combretum species</i>	644	27.8	<i>Pterocarpus lucens</i>	1 066	44.4	<i>Combretum species</i>	2 142	7.1	<i>Diplorhynchus condylocarpon</i>	2 877	33.3
2	<i>Terminalia sericea</i>	411	22.2	<i>Pseudolachnostylis maprouneifolia</i>	667	33.3	<i>Terminalia sericea</i>	1 142	14.3	<i>Baphia bequaertii</i>	1 544	11.1
3	<i>Ochna pulchra</i>	289	50	<i>Rhus tenuinervis</i>	369	77.8	<i>Vangueria infausta</i>	171	78.6	<i>Ochna pulchra</i>	1 089	5.6
4	<i>Burkea africana</i>	44	61.1	<i>Schrebera trichoclada</i>	300	83.3	<i>Ochna pulchra</i>	71	78.6	<i>Diospyros batocana</i>	644	55.6
5				<i>Baikiaea plurijuga</i>	78	94.4	<i>Burkea africana</i>	29	92.9	<i>Guibourtia coleosperma</i>	577	94.4

#ha⁻¹: Number of regeneration individuals per hectare; **RL (%)**: Regeneration limitation

4.4. Stand structure

4.4.1. Correlation of all species across study sites

The results from the linear model (Figure 4.12) showed that there was a positive association between diameter and height ($p < 0.0001$). The equation derived for the association of diameter and height showed that for every unit increase in diameter, height is expected to increase by 0.222 m everything being equal. The strength of the relationship was estimated at $R^2 = 0.474$, which explains 47.4% of the variance of errors in the model.

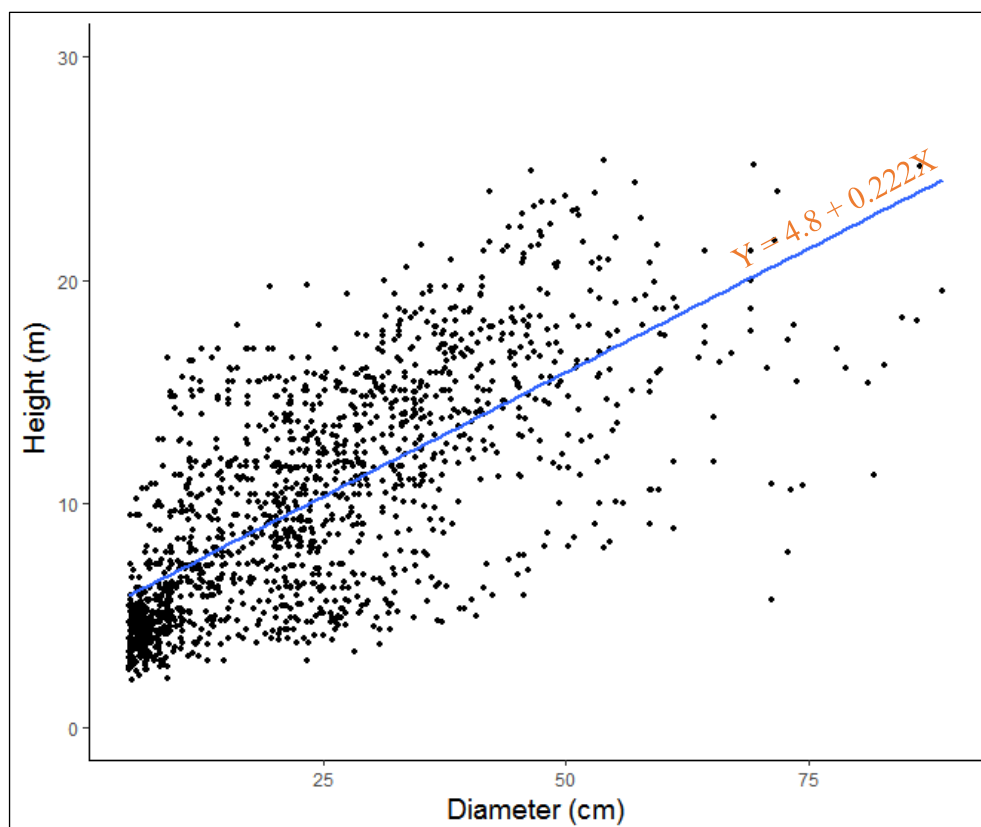


Figure 4.12: The plot of diameter and height correlation for all the woody species with dbh ≥ 5 cm across the study sites.

Comparing RB and RU, results of the correlational analysis indicated RB to have a better model than RU (Table 4.12). In both RB and RU, the association of diameter and height is positive. However, the model for RB is better and explains more the variance of error.

Table 4.12 Results of correlation analysis between tree diameter and height in RB and RU across study sites

Fire disturbance	Intercept	Slope	R ²	Adjusted- R ²	p-value
Recently burnt	3.50	0.25	0.636	0.636	< 0.0001
Recently unburnt	5.74	0.200	0.373	0.372	< 0.0001

4.4.2. Correlation of all species within study sites

Figure 4.11 shows the association of diameter and height in each of the study sites. Results indicate that all the study sites had positive slopes ($p < 0.0001$) which would result in corresponding increase in height when there is unit increase in diameter. The strength of relationship was weak in Kanovlei ($R^2 = 0.216$) followed by Caprivi ($R^2 = 0.274$) and Sesheke ($R^2 = 0.434$) while it was strong in Kabompo ($R^2 = 0.777$). Therefore, the variations in the model were better accounted for in Kabompo at 77.7%, followed by Sesheke at 43.4%, then Caprivi at 27.4% and lastly Kanovlei at 21.6%.

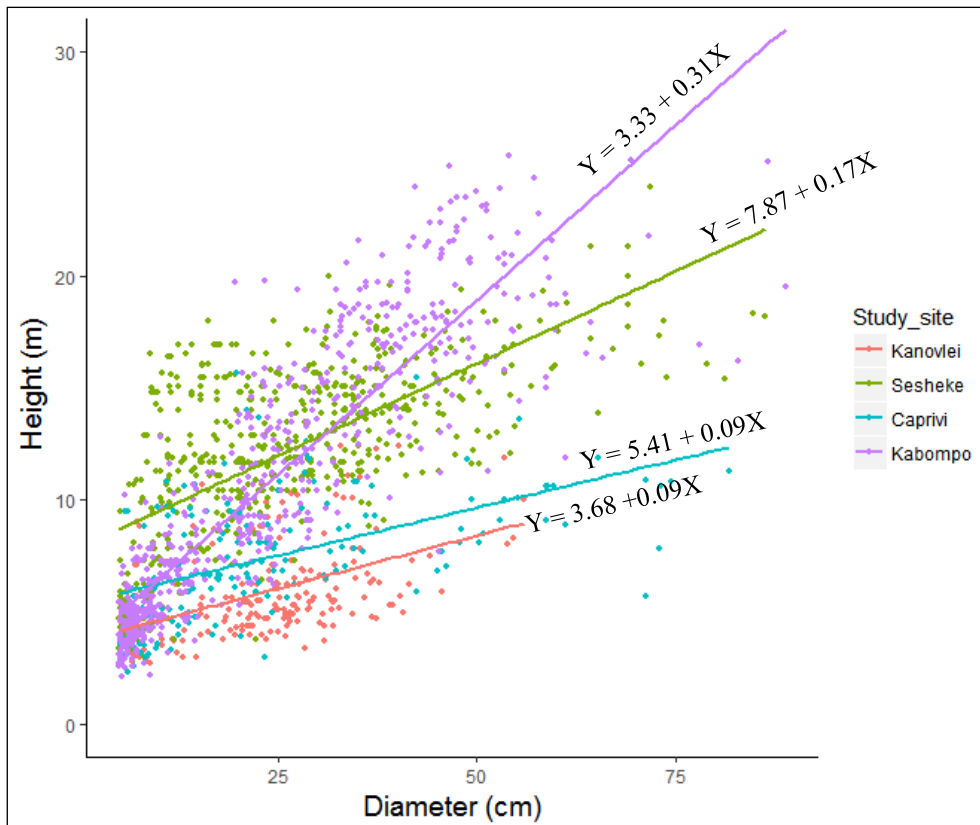


Figure 4.13 Correlation plot of diameter and height of all woody species per study site

4.4.3. Correlation of selected species

Table 4.13 compares the intercorrelations of diameter and height on selected species within the four study sites. Due to few stems in RB or RU for some of the selected species, regression results are not presented.

Table 4.13: Diameter and height correlation regression results of selected species in study sites where they occurred.

Species	Study site	Intercept	Slope	R ²	Adjusted- R ²	p-value
<i>Baikiaea plurijuga</i>	Sesheke	9.11	0.13	0.409	0.407	< 0.0001
	Caprivi	9.10	0.01	0.008	-0.015	0.552
<i>Burkea africana</i>	Kanovlei	4.17	0.07	0.097	0.088	0.002
	Kabompo	4.94	0.32	0.705	0.697	< 0.0001
<i>Guibourtia coleosperma</i>	Kabompo	8.90	0.12	0.646	0.630	< 0.0001
<i>Pterocarpus angolensis</i>	Kanovlei	4.57	0.07	0.120	0.109	0.001
	Caprivi	4.18	0.15	0.593	0.573	< 0.0001
	Kabompo	4.89	0.29	0.715	0.701	< 0.0001
<i>Schinziophyton rautanenii</i>	Caprivi	8.66	0.04	0.103	-0.047	0.439
	Kabompo	1.49	0.43	0.947	0.934	0.002
<i>Terminalia sericea</i>	Caprivi	3.59	0.05	0.068	-0.025	0.412

Results of diameter and height correlation for *B. plurijuga* at the Sesheke and Caprivi sites indicated positive but weak slopes and positive strength of the relationship for the two variables. The Caprivi site showed that the explanation towards the response variable was very low as indicated by a negative adjusted R² and non-significant explanatory variables ($p=0.552$). Therefore, the model cannot account for any variations whereas at Sesheke site only 40.9% of variations could be accounted for in the model.

The Kanovlei and Kabompo sites both indicated correlation of diameter and height in *B. africana*. Though the measure of relationship strength (R²) was weak for Kanovlei, it was nevertheless a positive one indicating a proportional response in height whenever there is a change in the unit of diameter. The explanations towards response variable were significant and better accounted for at Kabompo site with 70.5% unlike at Kanovlei site with 8.7%.

Correlation of diameter and height for *G. coleosperma* at the Kabompo site indicated a significant correlation ($p < 0.0001$). The relationship strength was positive where variances in the model were accounted for at 64.6%.

Correlation assessments for *P. angolensis* were done at Kanovlei, Caprivi and Kabompo sites. As shown in Table 4.6, association of diameter and height were confirmed to exist and were positive. The results for the Kanovlei site showed a weak relationship. This is confirmed by a very small slope and R^2 , compared with those at the other two sites. The relation strength of the two variables explained the variances at 12% at Kanovlei, 59.3% at Caprivi and 71.5% at Kabompo.

Results of correlation assessment between diameter and height for *S. rautanenii* in Caprivi was poor indicating that the response was very low or negligible ($p=0.439$) resulting in a negative R^2 . The Kabompo site showed a positive significant ($p=0.002$) and very strong relationship strength that is able to explain 94.7% of the variations in the model.

As shown in Table 4.13, results for *T. sericea* indicated very poor correlation between diameter and height. The model was poor and fit the data badly. Hence, a negative adjusted R^2 means that accounting for any variations in the model was very low or negligible.

4.4.4. Height size classes

Regeneration and adult trees were combined. They were then categorised in various size classes with a view to show population height dynamics and distribution of the selected woody species in the study sites. As earlier pointed out, SCD can indicate recruitment, survival ability and population stability of the species (Obiri et al., 2002). Table 4.9 provide results of ordinary least-squares regressions of midpoint size class and average number of individuals in size classes for the species at each site.

B. plurijuga's positive slope at the Sesheke site (0.121) indicated an inhibited and unstable recruitment. At the Caprivi site on the other hand, the negative slope (-0.146) was evidence of good recruitment. The shallow slopes meant that the population for the species at both the Sesheke and Caprivi sites are not stable resulting in few saplings. However, the species at the Caprivi site have some ability to survive to the next stage of life. Figure 4.14 shows the population dynamics for the species at the two sites. It is clearly shown that the species population is unstable due to very few individuals beyond the 1 m height class and the small amount of individuals in the crown classes.

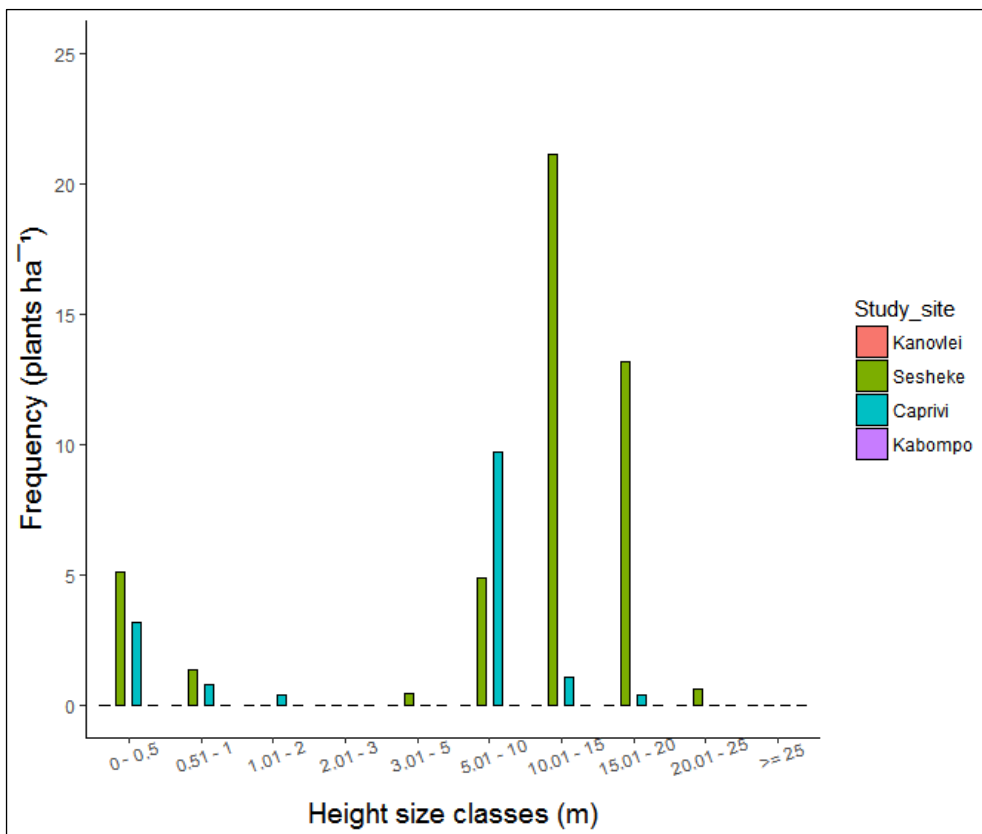


Figure 4.14: Height size class distribution of *Baikiaea plurijuga* at Sesheke and Caprivi study sites with absence of the species at Kanovlei and Kabompo study sites.

Statistical tests revealed that the interaction of fire with site was significant in the 0.51 – 1 m ($p=0.03$) and 10.1 – 15 m ($p=0.006$) classes. In the other size classes, fire treatment was non-significant ($p > 0.05$) at both sites. The site effect was significant ($p < 0.0001$) in the absence of interaction of the two treatments on SCD.

The presence of *B. africana* was registered at the sites of Kanovlei, Caprivi and Kabompo. The regression slopes were all negative though flat at the Kanovlei site and slightly shallow at the Caprivi and Kabompo sites. The negative slopes indicated recruitment of the species in the three sites. The slopes at the Kanovlei and Caprivi sites were not negatively significant while at the Kabompo site it was. Figure 4.15 gives a visual display of the population height dynamics of the species at the study sites. Height structure of the species at the Kanovlei site resembles a unimodal shape (Everard et al., 1995) with small number of individuals in the smaller size classes, which then picks up after the 3 m height and drops drastically after the 10 m height mark. At the Caprivi site, the species is shown to have very few individuals beyond the 1 m height mark only to appear after the 5 m height mark even though slope was indicative of a recruiting species. The Kabompo site has resemblance of a reverse J-shaped height structure, indicative of good recruitment, survival to the next stage and some stability of the species' population.

Statistical tests on SCD for the species in the three sites indicated that the interaction of fire with site was significant in the 0 – 0.5 m ($p=0.002$), 5.1 – 10 m ($p < 0.0001$) and 10.01 – 15 m ($p=0.029$) size classes. In the absence of interaction effect, fire treatment was significant in the 1.01 – 2 m size class ($p=0.043$). The site effect was significant (Appendix 4.4) in five other size classes in the absence of the interaction effect and effect of fire at each of the study sites.

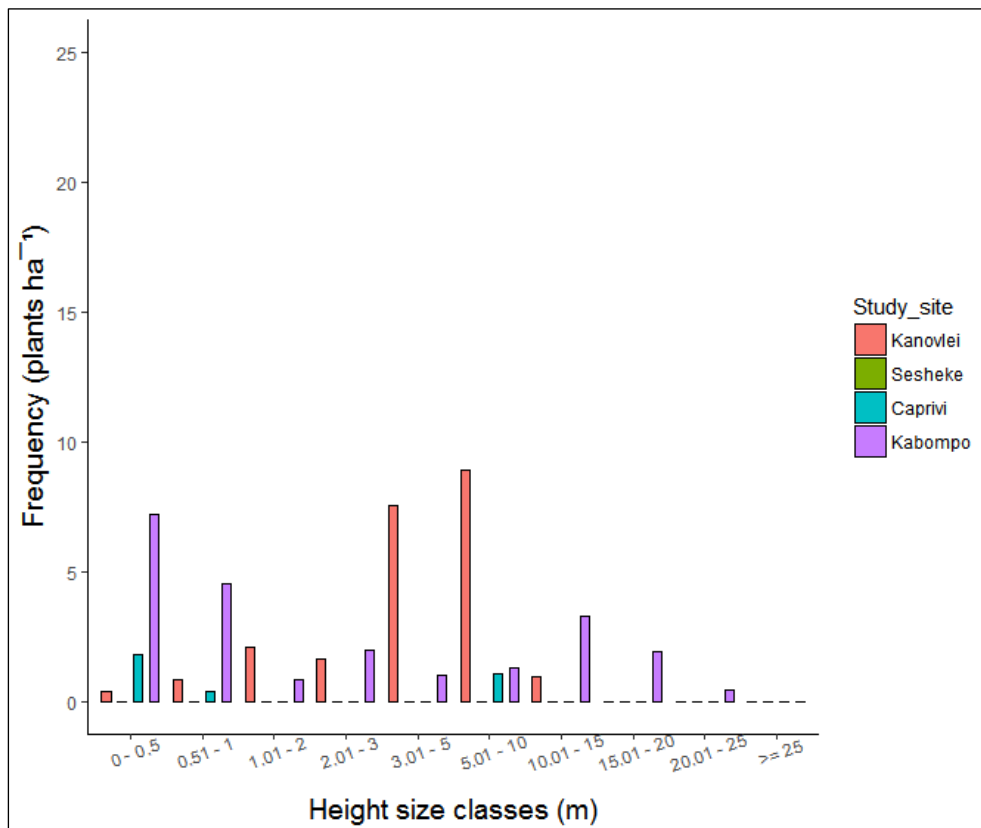


Figure 4.15: Height size class distribution of *Burkea africana* at Kanovlei, Caprivi and Kabompo study sites.

The SCD regression slope of *G. coleosperma* in Table 4.9 was negative (-0.120), an indicator of the species ability to recruit new individuals in the population with low densities. Lack of a steep slope is also an indicator of inhibited recruitment; hence, the negative slope was non-significant negative slope. Figure 4.16 clearly shows that there is less rejuvenation in 0 – 0.5 m class and the population of the species is declining as there are very few individuals surviving to the next stage of saplings and juveniles since individuals are missing in the 2.01 – 3 m and 3.01 – 5 m classes. This indicates that the recruitment is somewhat episodic.

The statistical tests on the effect of fire treatment showed that there were no significant ($p > 0.05$) differences in all the size classes for the species.

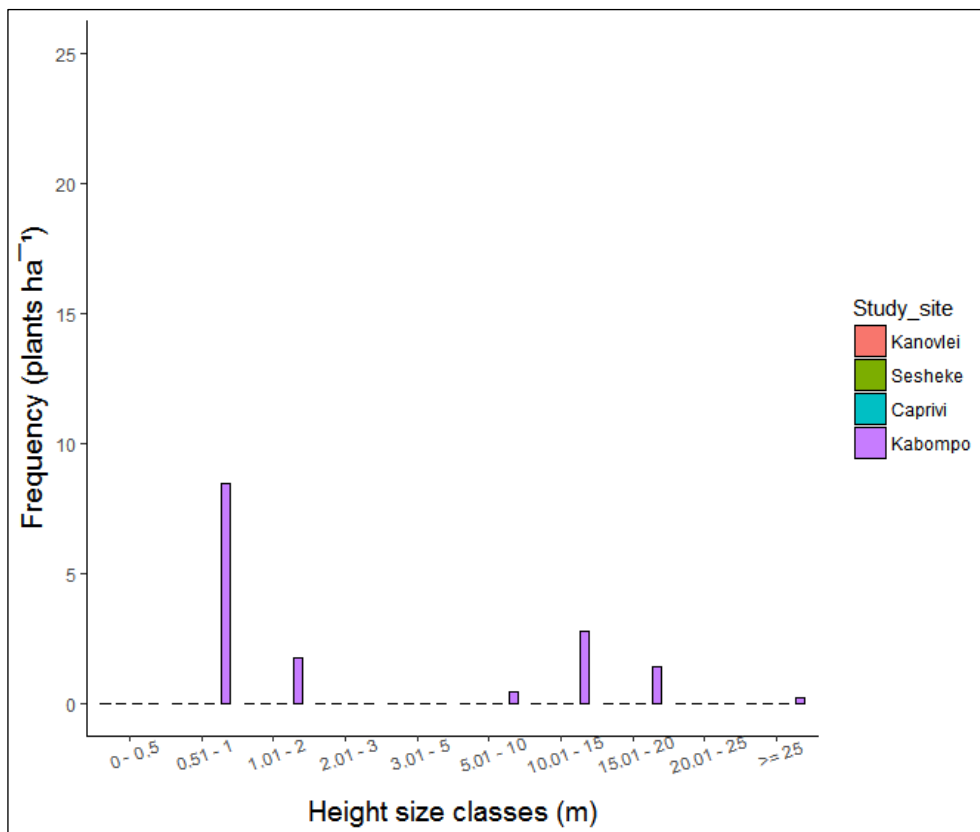


Figure 4.16: Height size class distribution of *Guibourtia coleosperma* at Kabompo study site.

The regression slopes on SCD of *P. angolensis* at the Kanovlei and Caprivi sites were positive slopes (Table 4.9). The flat slopes were caused by similar numbers of individuals in small and large size classes leading to low recruitment and diminishing populations of the species at the two sites. The Kabompo site on the other hand had a significant negative slope (-0.321 , $p=0.005$) indicative of abundant recruitment of individuals in the population. Figure 4.17 displays SCD at the three sites.

The statistical evidence on the interaction effect of fire with site revealed significant effects in the 5.01 -10 m ($p=0.010$) and 10.01 – 15 m ($p=0.004$) size classes. The effect of fire treatment was significant in 2.01 – 3 m ($p=0.041$) size class of Kanovlei site. The site effect was significant ($p < 0.0001$) in four size classes whereas the remaining four size classes were non-significant ($p > 0.05$) including on the interaction and fire treatment effects.

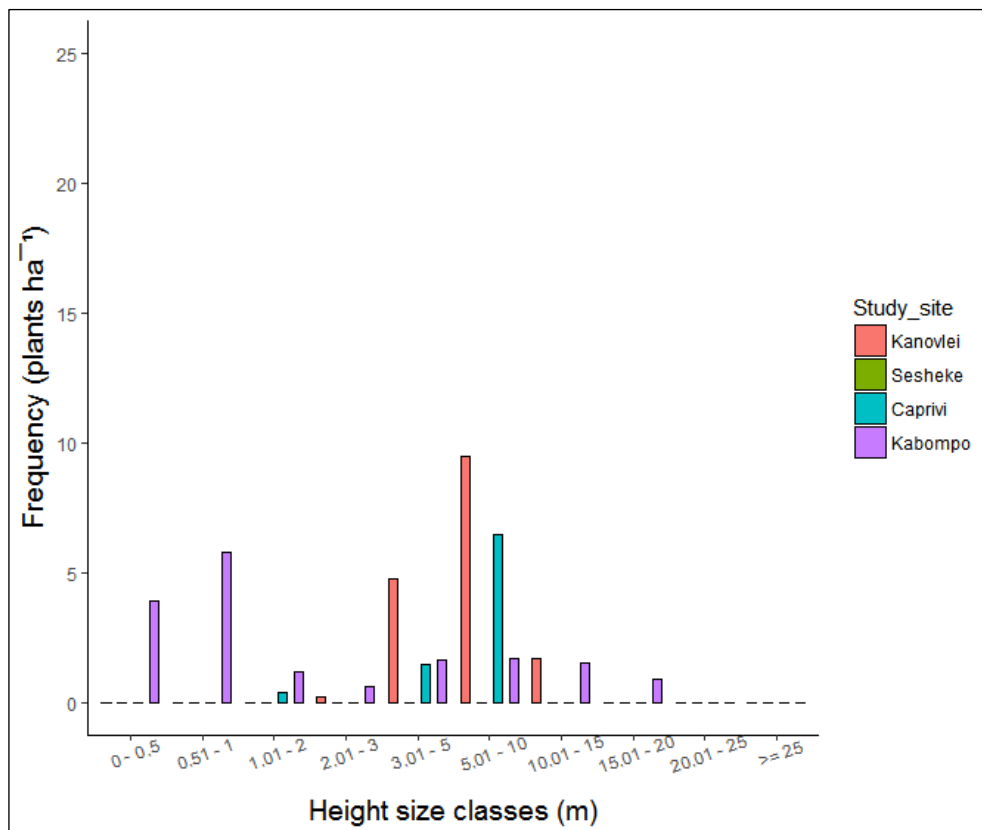


Figure 4.17: Height size class distribution of *Pterocarpus angolensis* at Kanovlei, Caprivi and Kabompo study sites.

The regression slopes on SCD for *S. rautanenii* in the study sites reviewed that the species had poor recruitment at the Sesheke (0.121) and Caprivi sites (0.056). The negative slopes at Kanovlei (-0.003) and Kabompo sites (-0.139) indicated slightly better recruitment and stability of the species in the next life stages. However, overall recruitment is inhibited resulting in unstable population in all the sites since all the slopes lacked steep negative slopes. Figure 4.18 shows SCD of the species.

The statistical tests on the species SCD showed that the effect of interaction between fire and site was non-significant ($p > 0.05$) in all the size classes where the species was present. The effect of fire treatment was also non-significant ($p > 0.05$) on the species at all the sites. However, the site effect was significant ($p < 0.05$) in six of the size classes with an exception of the 10.01 – 15 m size class. The last three size classes had no counts to make inferences on.

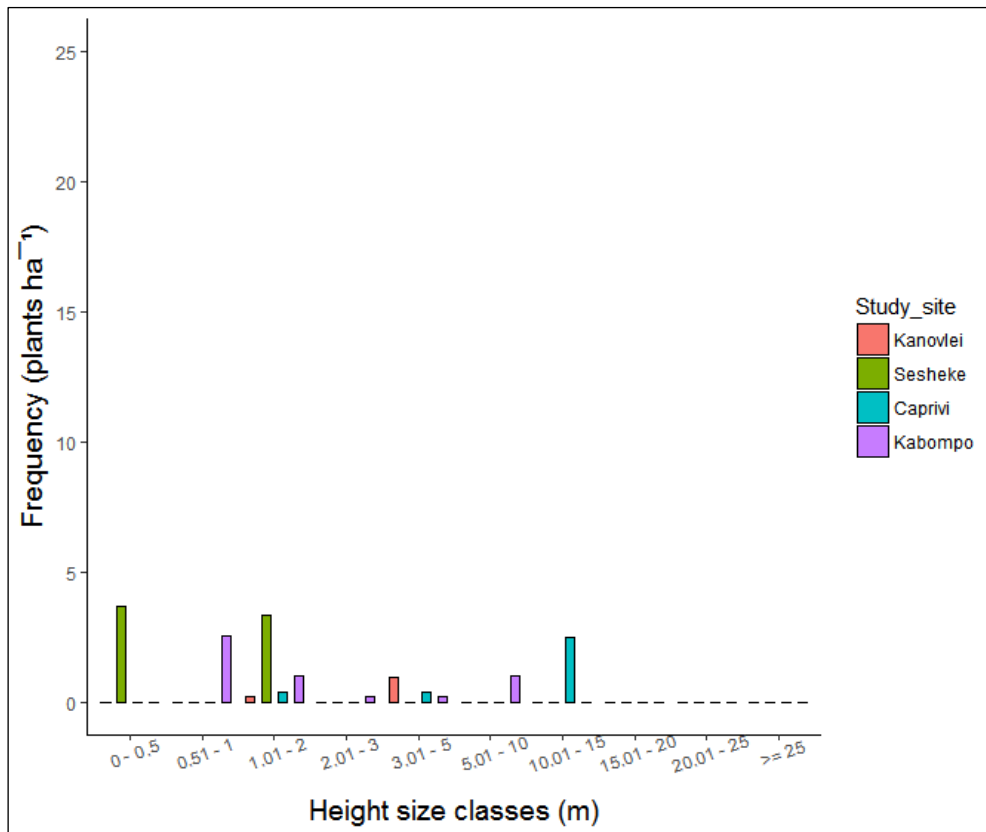


Figure 4.18: Height size class distribution of *Schinziophyton rautanenii* across the four study sites.

Table 4.9 indicate that regression slopes for *T. sericea* were negative at the Kanovlei (-0.003), Caprivi (-0.631) and Kabompo sites (-0.138). The flat slope at the Kanovlei site is an indicator of equal number of individuals in small size classes and large size classes. The slope at the Caprivi site indicated good recruitment and survival to sapling stage while that at Kabompo site indicated recruitment only. Figure 4.19 shows rejuvenation of the species in the small size classes and the absence of the species in the large size classes.

There was evidence of statistically significant differences in the first three size classes on the effect of interaction between fire treatment and site. The fire treatment was also significant in two size classes of Caprivi site in the absence of interaction effect while site effect was significant in four of the size classes in the absence interaction effect. P-values are shown in Appendix 4.5.

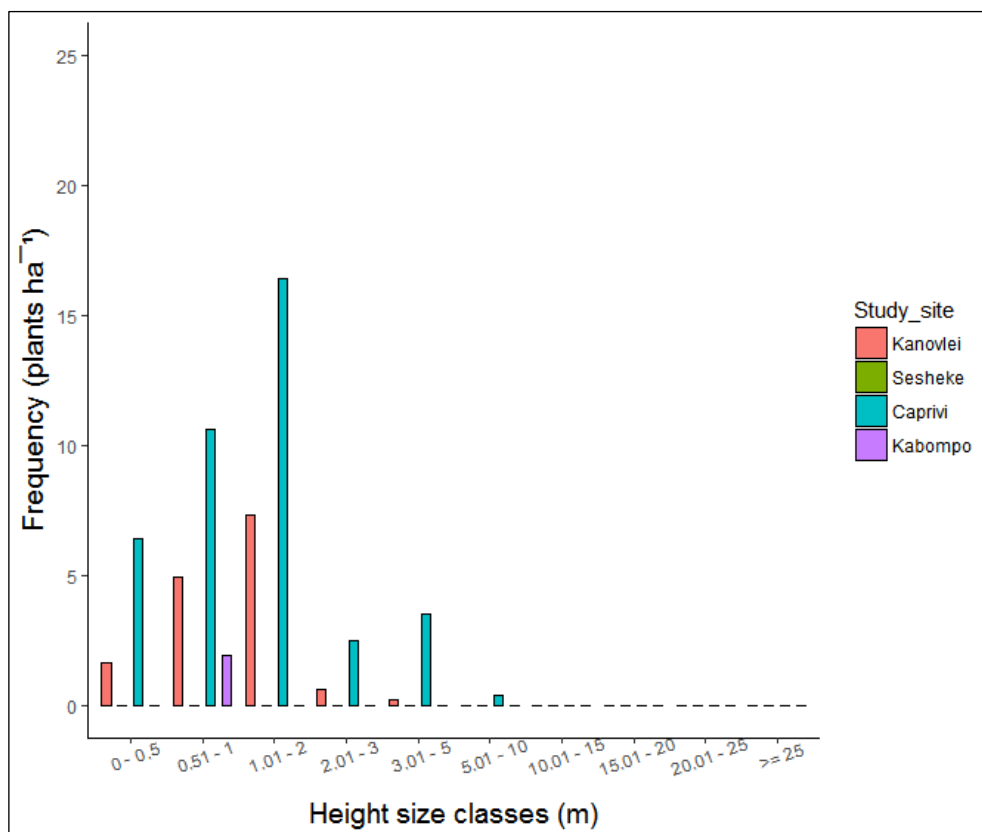


Figure 4.19: Height size class distribution of *Terminalia sericea* at Kanovlei, Caprivi and Kabompo study sites.

4.4.5. Diameter size classes

Results of OLS regression for selected species at study sites are given in Table 4.9. The OLS regression results give a good understanding of population structure and dynamics by way of making inference on the slopes (Sop *et al.*, 2011; Traoré *et al.*, 2012). The SCD slopes ranged from -0.734 for *B. africana* at Kanovlei site to 0.382 for *B. plurijuga* at Sesheke site.

B. plurijuga showed a poor and inhibited population structure at both the Sesheke and Caprivi sites since the slopes were positive and almost flat. This meant that there were slightly more individuals in the large size classes than the smaller size classes. Positive slopes in both sites implied an aging and declining population for the species.

Figure 4.20 shows the Sesheke site to have a bell shape distribution with a peak in the 35.1 – 40 cm size class. The bell shaped population structure at the Sesheke site is a clear indication of an unstable population for the species. The Caprivi site had a number of individuals per hectare varying from one size class to the next without any defined shape to describe it with and certain size classes having almost the same number of individuals resulting in very poor unstable population structure with less rejuvenation.

The effect of interaction of fire with site on diameter SCD was negatively significant only in the 40 – 45 cm class ($p=0.047$). The effect of fire treatment was negatively significant in the 50 – 55 cm ($p=0.003$) and > 60 cm ($p=0.005$) classes of Sesheke site. The effect of site was highly significant ($p < 0.0001$) in all size classes.

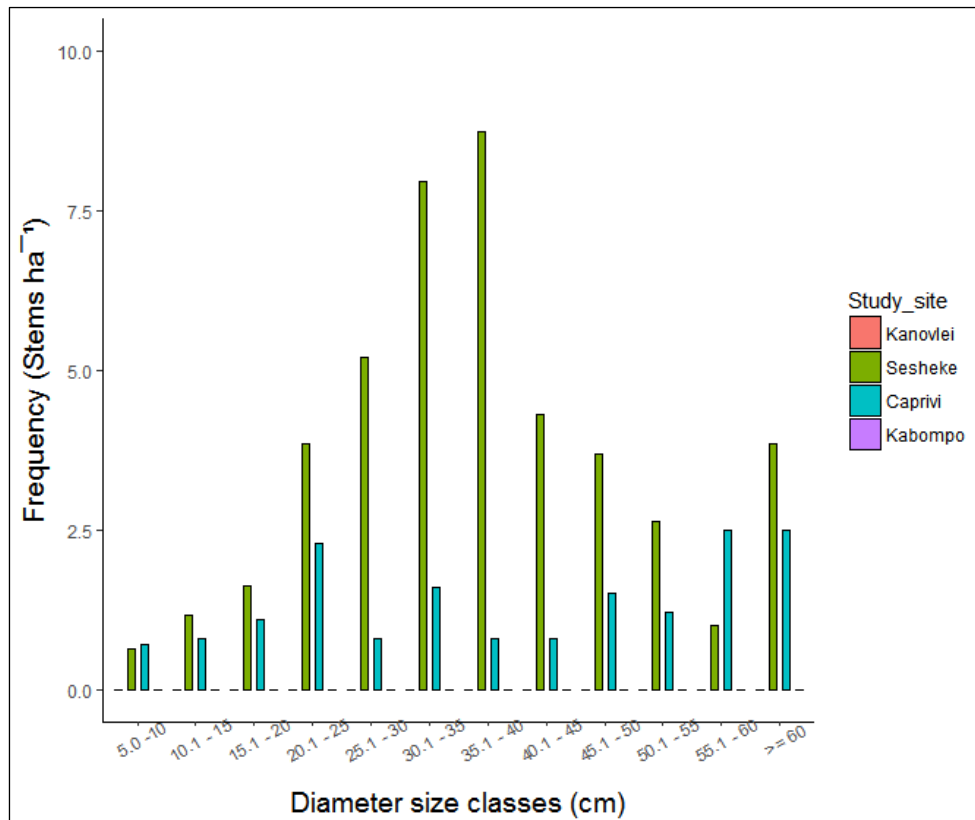


Figure 4.20: Diameter size class distribution of *Baikiaea plurijuga* at Sesheke and Caprivi study sites.

B. africana had negative slopes (Table 4.9) at both the Kanovlei and Kabompo sites, an indicator that the species was rejuvenating. The Kanovlei site displayed a significant negative slope ($p=0.027$) that indicated significant differences in diameter distributions across size classes while at the Kabompo site the negative slope did not indicate significant differences in diameter distribution for species.

Figure 4.21 shows diameter distribution of the species at then Kanovlei and Kabompo sites as well as indicating absence at the Sesheke and Caprivi sites. Both sites indicated variations in the population structure with the Kanovlei site having a sporadic one and that of the Kabompo site almost a uniform one before rising slightly and dropping in the large classes. The lack of reverse J-shaped population structures at both sites implies unstable populations even though there are indication of rejuvenation in small size classes.

Statistical tests showed that the interaction effect was positively significant ($p=0.025$) in the 20 – 25 cm class only. The fire treatment was non-significant ($p > 0.05$) in all the other size classes. However, site effect was significant ($p < 0.0001$) in 6 other size classes except in the last 5 large size classes ($p > 0.05$).

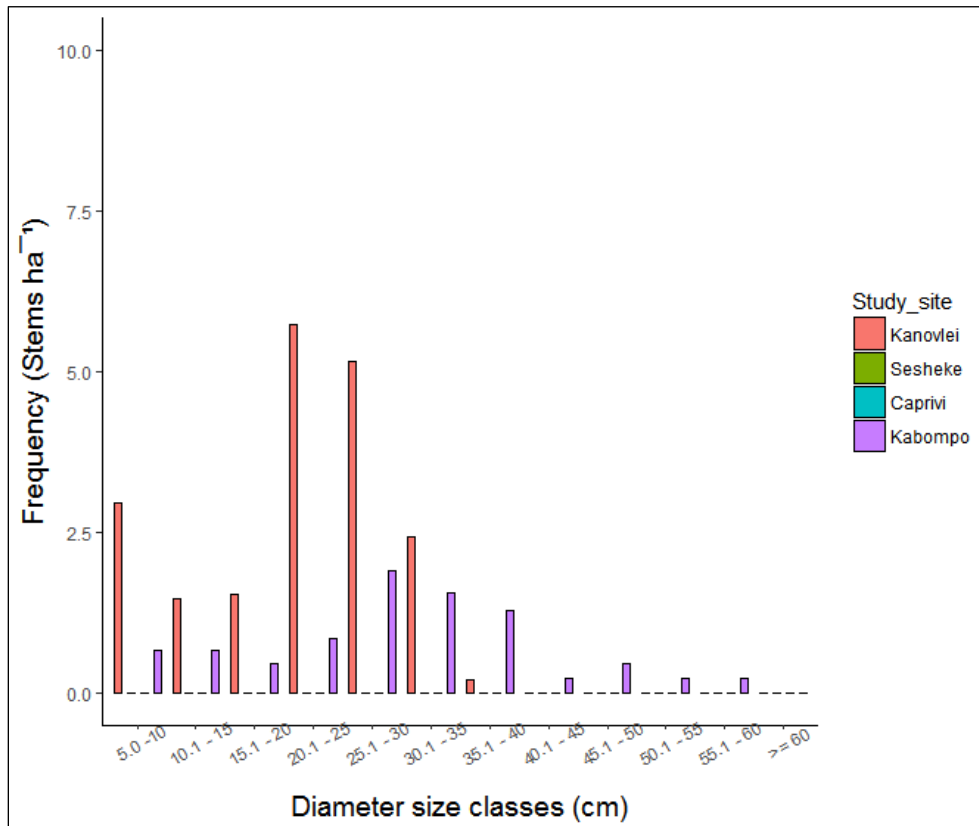


Figure 4.21: Diameter size class distribution of *Burkea africana* at Kanovlei and Kabompo study sites.

G. coleosperma was absent in three sites except for the Kabompo site. As indicated in Table 4.9, the regression slope for the species was flat and positive, an indication of declining population for the species. Figure 4.22 clearly shows the species absence in three sites, lack of rejuvenation and absence of individuals in certain large size classes at the Kabompo site.

Statistical significance test of fire treatment at the Kabompo site showed that the SCD was non-significant ($p > 0.05$). The number of individuals per hectare were less than 2 in size classes where the individuals were present.

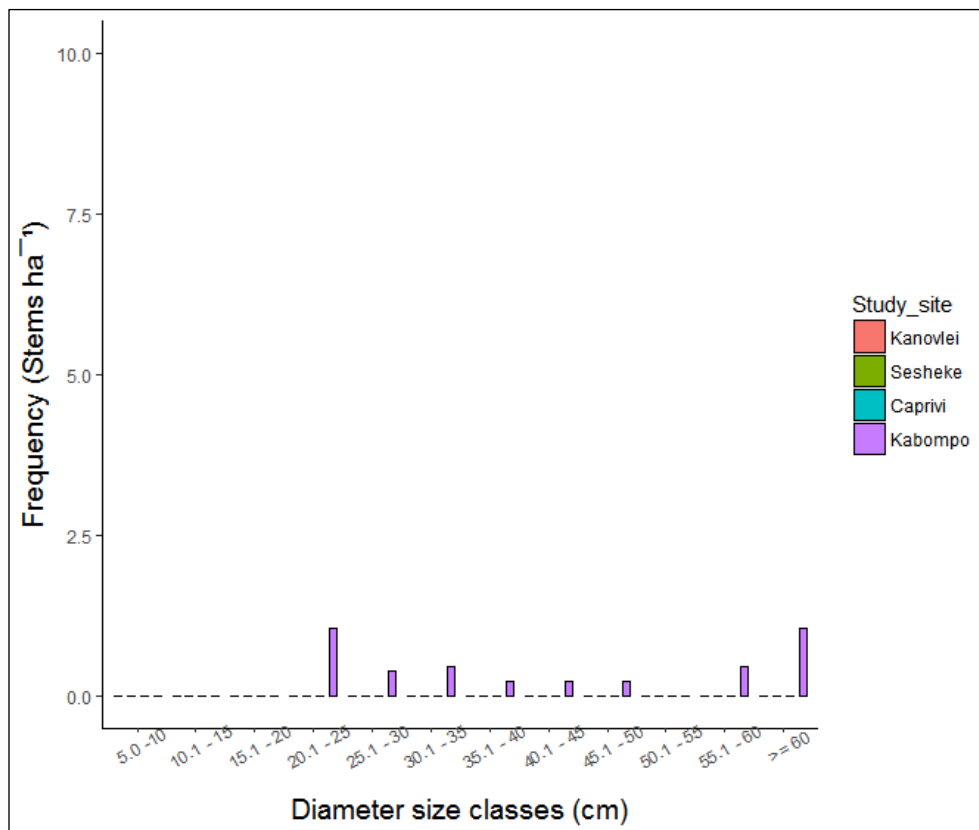


Figure 4.22: Diameter size class distribution of *Guibourtia coleosperma* at Kabompo study site.

Table 4.9 shows that the regression slopes on SCD for *P. angolensis* were negative at the Kanovlei (-0.325), Caprivi (-0.644) and Kabompo (-0.383) sites. The negative slopes were evidence of rejuvenation and presence of more individuals in small size classes than the large ones. The slope for the Kanovlei site was not negatively significant ($p = 0.155$) whereas Caprivi ($p < 0.001$) and Kabompo ($p = 0.007$) sites were negatively significant.

Figure 4.23 shows the size class distribution at the Kanovlei site to be a bell shaped whereas at the Caprivi and Kabompo sites the population structures resembles a reverse J-shape though not perfectly. Statistical test results for the species (Appendix 4.6) indicates that the interaction effect of fire with site was significant in six size classes. The remaining size classes were non-significant on the interaction effect. However, 10 – 15 cm size class at the Caprivi site showed that fire treatment had a significant effect on diameter distribution while all the size classes that showed non-significant differences on the interaction effect showed significant differences due to site effect

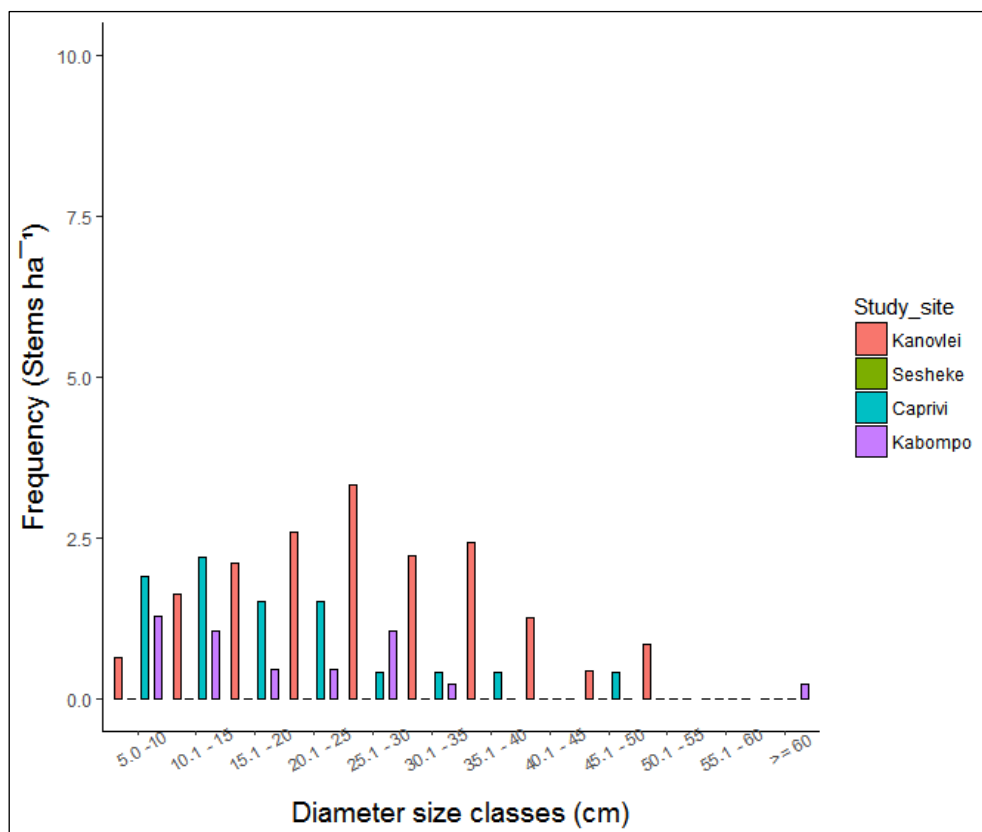


Figure 4.23: Diameter size class distributions of *Pterocarpus angolensis* at Kanovlei, Caprivi and Kabompo study sites.

The regression slope for *S. rautanenii* as shown in Table 4.9 is positive, which meant that there were more individuals in large size classes than in small size classes. The positive slope meant lack of rejuvenation as shown by the absence of individuals in the small size classes, an indicator of unstable and declining population for the species. The population cannot be said to have rapid growth in the small size classes or to have high survival rate since the SCD distribution is flat (Lykke, 1998). Due to the few number of individuals in Kanovlei and Kabompo sites, regression slopes could not be calculated for the two sites.

The average numbers of individuals at both the Kanovlei and Kabompo replicates were found to be less than one in the size classes where the species presence was recorded. At the Caprivi site, the species is barely surviving, as there are less than two individuals per hectare in the size classes where the species was present (Appendix 4.7).

Statistical test results showed that interaction effect of fire with site was significant only in the 15 – 20 cm ($p = 0.0001$). The fire treatment effect was significant ($p = 0.002$) at the Caprivi site while other size classes at all sites were non-significant ($p > 0.05$) where interaction effect was non-significant. However, site effect was significant ($p < 0.0001$) in these same size classes.

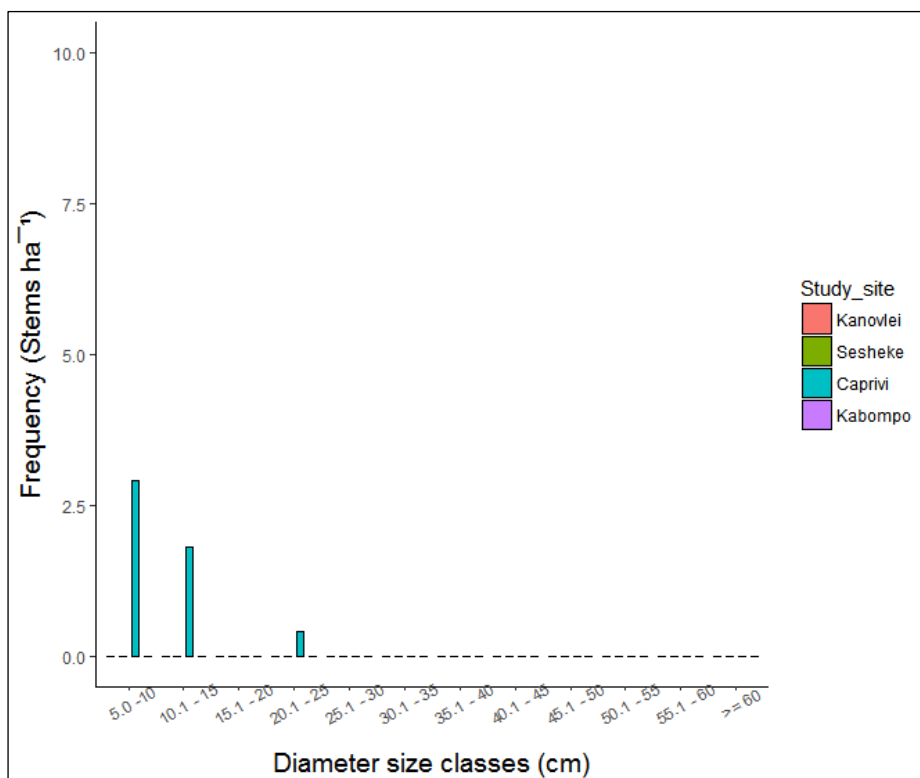


Figure 4.24: Diameter size class distribution of *Terminalia sericea* at Caprivi study site.

Adult *T. sericea* trees were only present at the Caprivi site. A negative slope for SCD indicated that the species was rejuvenating at the Caprivi site and there were more individuals in the small size classes than in the large size classes. The slope was also negatively significant ($p = 0.001$), which implied that there were significant differences in the number of individuals among the size classes.

Figure 4.24 display the diameter distribution in size classes at Caprivi state forest and its absence in the other sites. The reverse J-shaped population structure indicated good rejuvenation but individuals did not show much progress beyond juvenile stage.

Statistical test results revealed that fire treatment effect was significant ($p < 0.001$) in the size classes where individuals were present.

4.4.6. Mean dbh and mean height

Figure 4.25 shows the mean diameter and mean height for all species ($\text{dbh} \geq 5 \text{ cm}$) at three study sites of Kanovlei, Sesheke and Kabompo. The Caprivi study site was excluded on these parameters due to the heavy influence of anthropogenic effects. The Kruskal-Wallis test indicated that site had a significant effect ($p=0.002$, $df=2$) on mean diameter. The Holm corrected p-values showed the actual significant difference between Kanovlei and Sesheke sites ($p=0.002$), and also between Kanovlei and Kabompo sites ($p=0.02$). The mean diameter according to site were 26.0 (± 0.9) cm at the Kanovlei, 32.7 (± 1.96) cm at the Sesheke and 29.6 (± 0.7) cm at the Kabompo sites. The effect of fire treatment was significant ($p=0.008$) only significant at the Sesheke site ($p=0.0004$) and not at the other two sites ($p>0.05$).

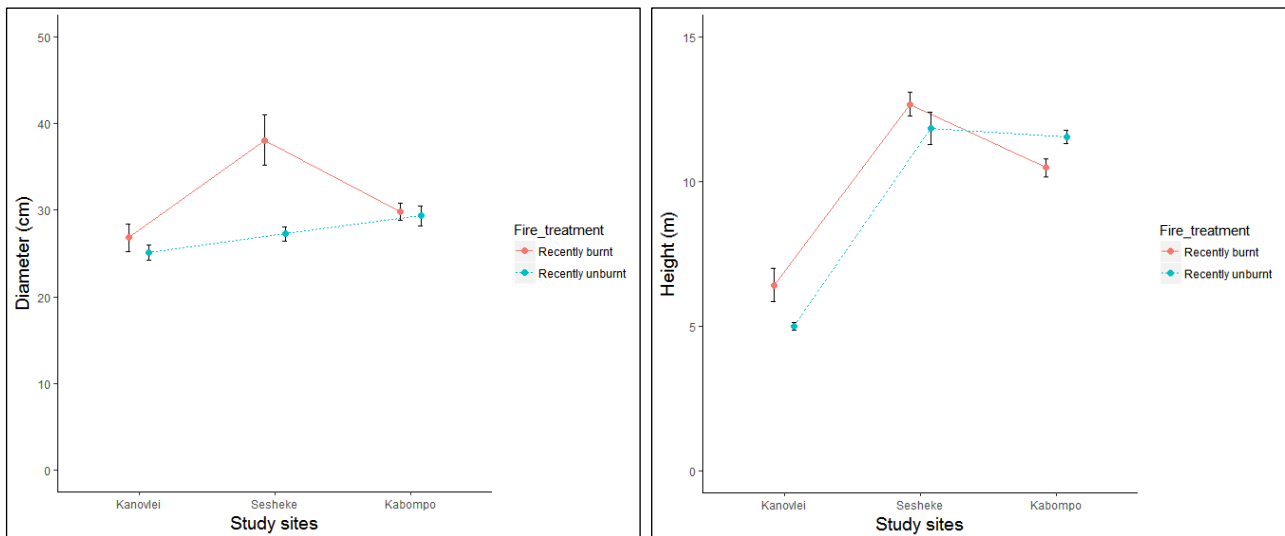


Figure 4.25: The distribution of mean diameter and mean height of all the woody species across study sites comparing the effect of RB and RU fire treatment. The means are plotted with standard errors. Caprivi site was excluded due to heavy influence of anthropogenic effects.

The interaction of fire treatment with site had a significant effect ($p=0.012$, $df=2$) on mean height across study sites. The overall site mean height, not shown in Figure 4.23, were 5.8 (± 0.3) m at Kanovlei, 12.3 (± 0.4) m at Sesheke and 11.0 (± 0.2) m at Kabompo.

Table 4.15 presents the mean diameter and mean height results of selected species across study sites. The difference of mean diameter of *B. plurijuga* was non-significant ($p=0.625$, $df=1$) between the Sesheke and Caprivi sites. The mean diameter of *B. africana* ($p=0.011$, $df=1$) and *P. angolensis* ($p=0.002$, $df=2$) were significantly different between the Kanovlei and Kabompo sites. The mean diameter of *S. rautanenii* was non-significant ($p=0.229$, $df=2$). The mean diameters for all the selected species were non-significant ($p > 0.05$) between RB and RU within sites.

The mean height of *B. plurijuga* was significantly different ($p = 0.024$, $df=1$) between the Sesheke and Caprivi sites. The mean height of *B. africana* was also significant ($p < 0.0001$, $df=1$) between the Kanovlei and Kabompo sites while the mean height of *S. rautanenii* was non-significant ($p=0.501$). All selected species showed non-significant ($p > 0.05$) differences between RB and RU within sites.

Table 4.14: Results of the mean diameter and mean height of selected species at sites where they occurred. Shown in brackets are standard errors of the mean.

Species	Study site	Mean diameter (cm)			Mean height (m)		
		RB	RU	SM	RB	RU	SM
<i>B. plurijuga</i>	Sesheke	41.8 (± 2.6)	42.2 (± 3.2)	42.0 (± 2.0)	14.1 (± 0.3)	14.2 (± 0.4)	14.2 (± 0.3)
	Caprivi	30.9 (± 5.2)	45.5 (± 5.5)	40.7 (± 4.6)	7.6 (± 0.3)	10.1 (± 0.5)	9.3 (± 0.5)
<i>B. africana</i>	Kanovlei	23.1 (± 1.6)	22.9 (± 0.9)	23.0 (± 0.9)	6.1 (± 0.6)	4.8 (± 0.1)	5.42 (± 0.3)
	Kabompo	34.6 (± 5.3)	26.9 (± 2.2)	30.8 (± 3.0)	15.2 (± 1.7)	13.4 (± 0.9)	14.3 (± 1.0)
<i>G. coleosperma</i>	Kabompo	55.7 (± 9.3)	49.4 (± 9.9)	52.3 (± 6.6)	14.0 (± 1.0)	16.3 (± 2.0)	15.3 (± 1.2)
<i>P. angolensis</i>	Kanovlei	30.0 (± 3.3)	31.9 (± 1.9)	30.9 (± 1.9)	7.1 (± 0.9)	5.8 (± 0.4)	6.5 (± 0.6)
	Caprivi	19.7 (± 1.1)	23.3 (± 11.6)	21.5 (± 4.9)	7.1 (± 0.8)	6.8 (± 2.0)	6.9 (± 0.9)
	Kabompo	18.6 (± 7.7)	18.4 (± 3.3)	18.5 (± 3.8)	8.91 (± 2.2)	10.8 (± 1.0)	9.9 (± 1.1)
<i>S. rautanenii</i>	Kanovlei	-	14.7	14.7	-	3.8	3.8
	Caprivi	16.8	38.62	27.7 (± 10.9)	4.6	10.8	7.4 (± 3.4)
	Kabompo	13.3	12.69 (± 3.1)	12.8 (± 2.4)	7.2	7.0 (± 1.4)	7.0 (± 1.1)
<i>T. sericea</i>	Caprivi	10.6 (± 1.5)	7.3	9.5 (± 1.4)	4.0 (± 0.2)	4.1	4.0 (± 0.1)

RB: Recently burnt; RU: Recently unburnt; SM: overall mean for the site

Mixed results were revealed from the assessment of the effect fire frequency on mean diameter and mean height across study sites and on selected species.

The Kruskal-Wallis test revealed that fire frequency had no significant ($p=0.096$) effect on mean diameter of the woody species across the study sites. Figure 4.26 shows the mean diameter according to fire frequency on the plots.

The Kruskal-Wallis test results revealed that fire frequency had no significant ($p=0.4001$) effect on the mean diameters of the selected species. However, the mean diameter of the selected species differed significantly ($p<0.0001$, $df=5$) among study sites. The post-hoc Dunn test (Holm corrected) results for the species that had significantly different mean diameters are shown in Appendix 4.8.

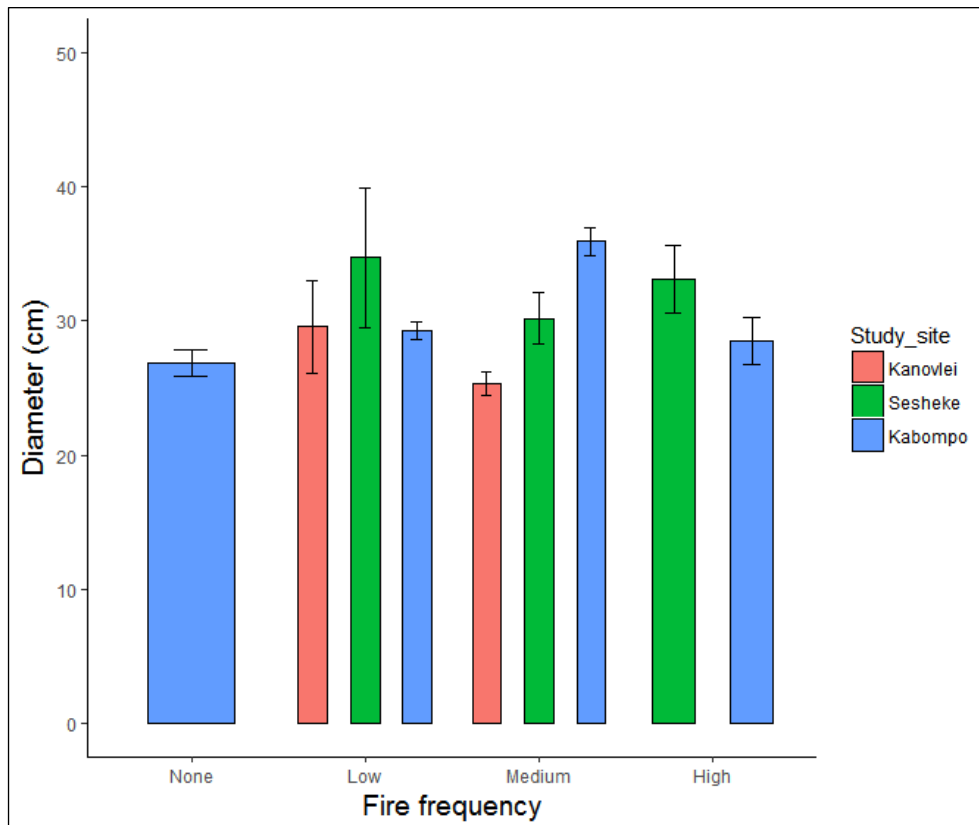


Figure 4.26: The mean diameter at breast height (dbh) of all woody species with minimum dbh of 5 cm at Kanovlei, Sesheke and Kabompo sites.

Figure 4.27 shows the mean diameter for all the selected species according to fire frequency across study sites.

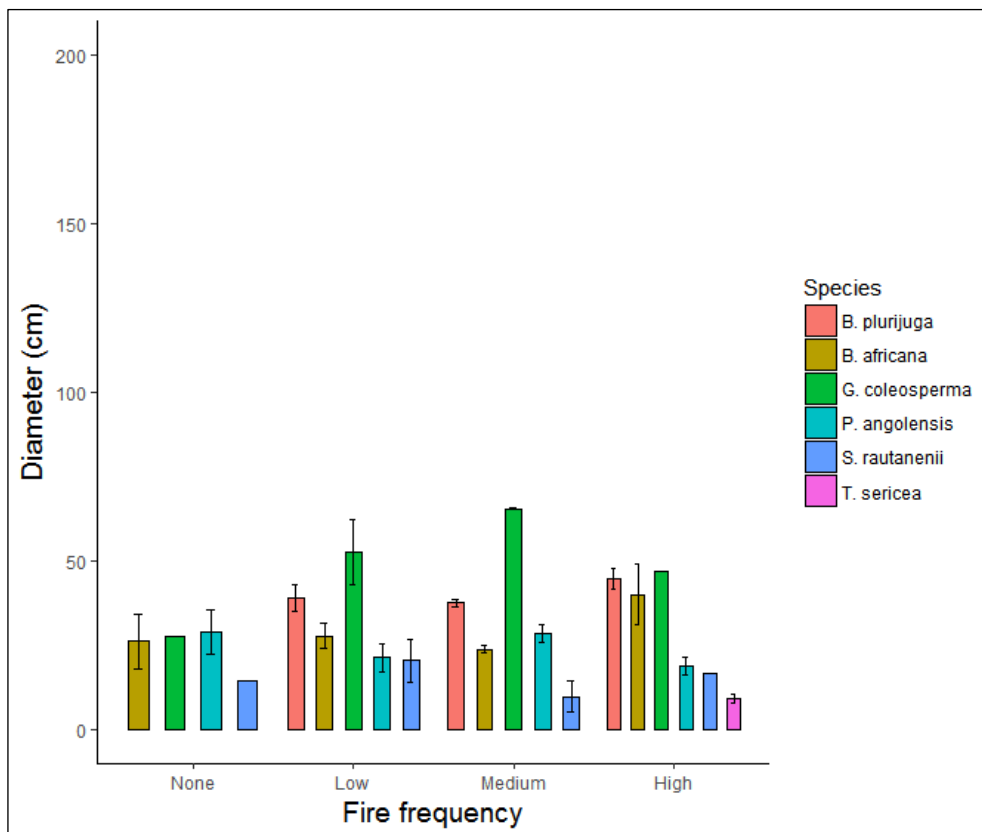


Figure 4.27: The mean diameter of the selected species based on the fire frequency at the plot across the study sites.

The analysis of variance (ANOVA) results for the interaction of fire frequency with site revealed a significant effect ($p=0.012$) on mean height of all woody species across at Kanovlei, Sesheke and Kabompo study sites. Figure 4.28 shows the distribution of mean height for all the woody species according to fire frequency. The ANOVA table is shown in Appendix 4.9.

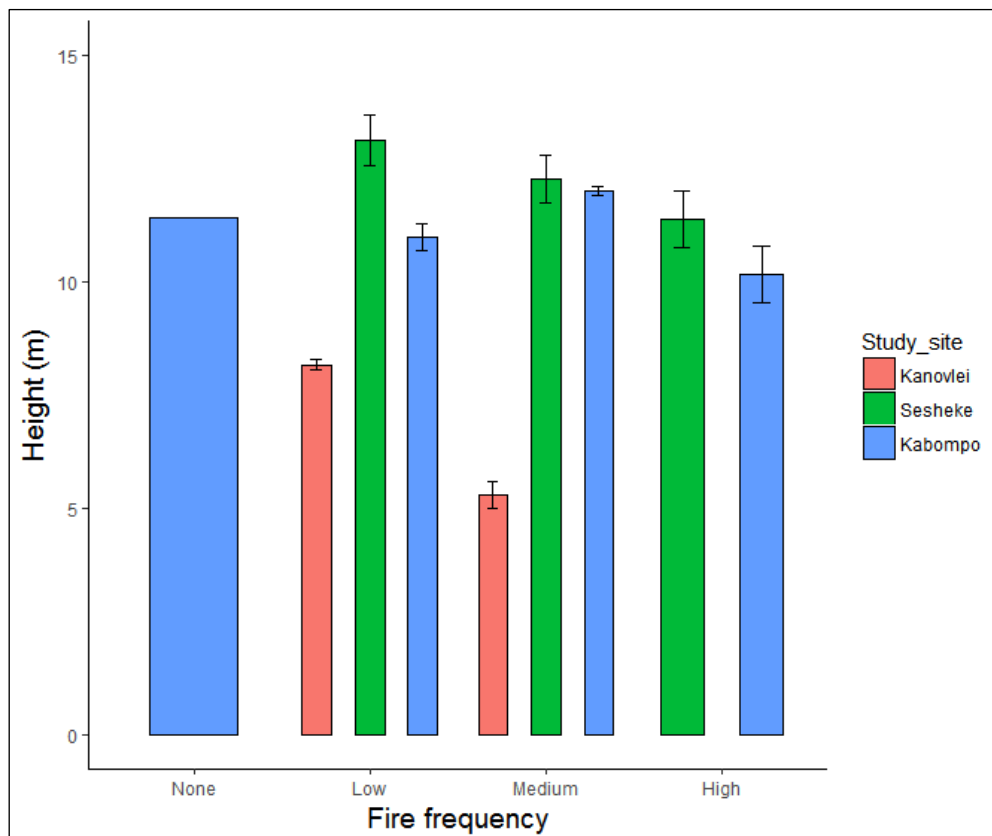


Figure 4.28: The mean height distribution of all woody species as affected by fire frequency at Kanovlei, Sesheke and Kabompo study sites.

The mean height of the selected species is shown in Figure 4.29. The variations were more in the medium and high fire frequency bands compared to the other two bands. The interaction of fire frequency with the species had a significant effect on the mean height of the selected species. The Anova results are shown in Appendix 4.10.

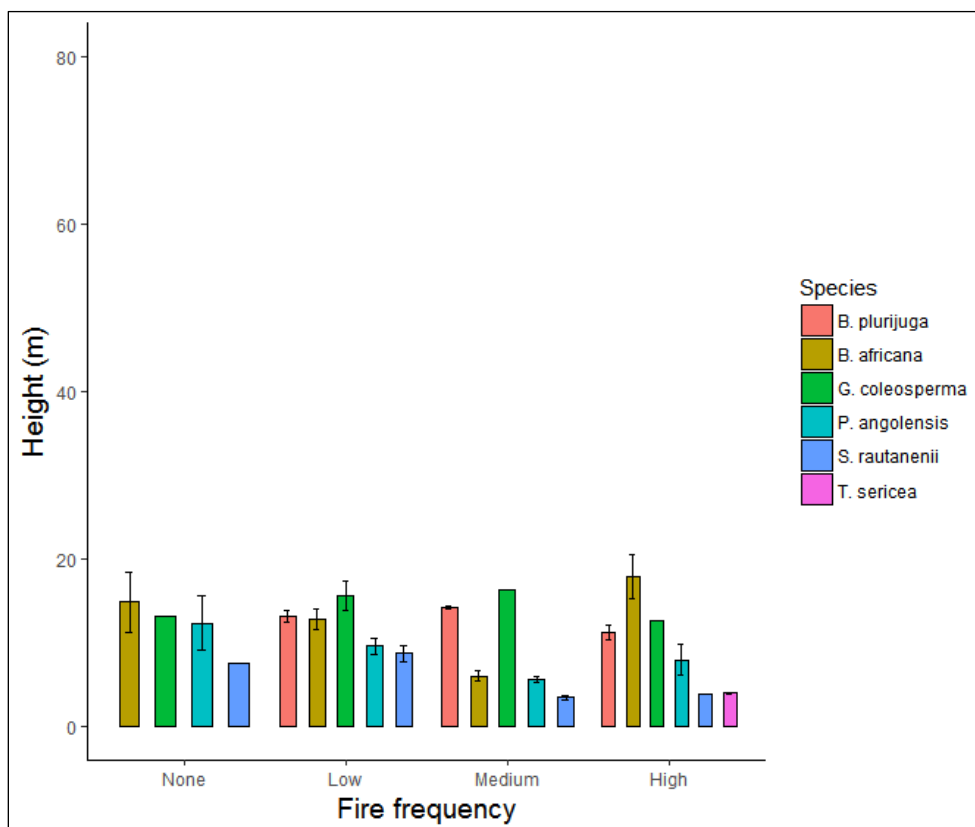


Figure 4.29: The mean height distribution of the selected species following the fire frequency at Kanovlei, Sesheke and Kabompo study sites.

4.4.7. Basal area for all species across study region

Figure 4.30 shows basal area for all species in RB and RU across study sites. The smallest basal area was at the Kanovlei site with $1.2 (\pm 0.2) \text{ m}^2 \text{ ha}^{-1}$, followed by the Caprivi site with $1.7 (\pm 0.3) \text{ m}^2 \text{ ha}^{-1}$ and the Kabompo site with $3.2 (\pm 0.5) \text{ m}^2 \text{ ha}^{-1}$ while the Sesheke site had the highest with $8.2 (\pm 0.9) \text{ m}^2 \text{ ha}^{-1}$ irrespective of the fire treatment.

The Mann-Whitney test revealed that the effect of fire treatment on basal area within sites was not statistically significant ($p > 0.05$) across study sites except at the Sesheke study site ($p=0.04$). The Kruskal-Wallis test on the effect of site was highly significant ($p < 0.0001$, $df = 3$) on basal area. The post-hoc Dunn test (holm corrected) showed significant differences ($p < 0.0001$) between the Sesheke and the other three sites of Kanovlei, Caprivi and Kabompo.

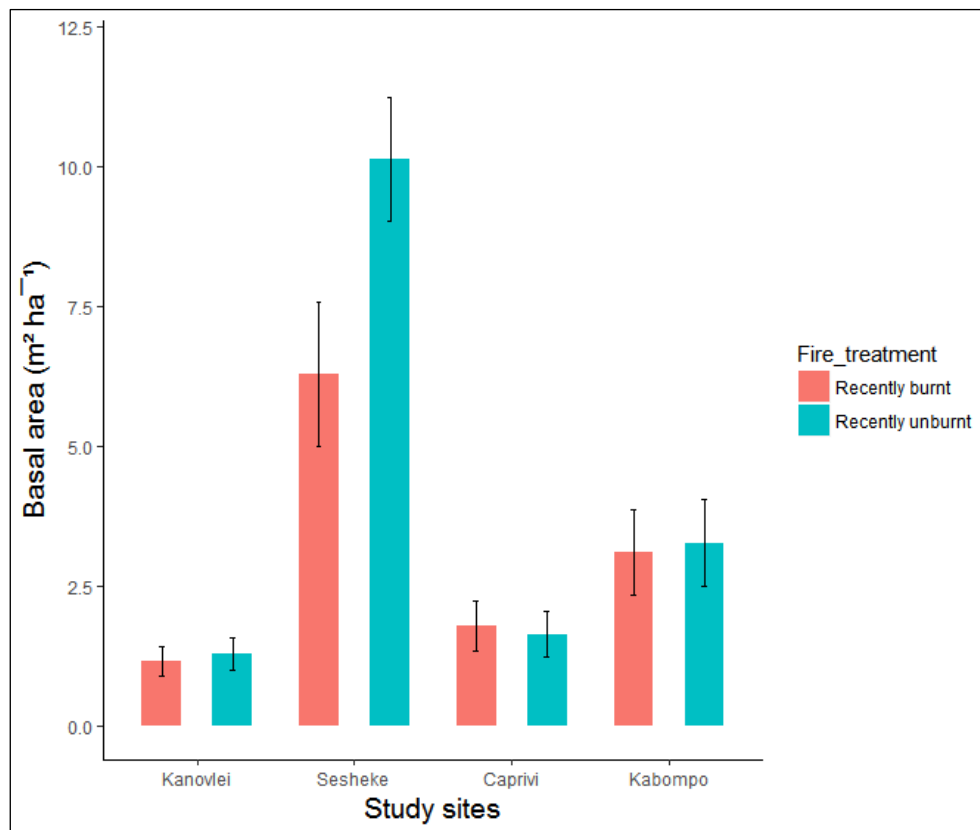


Figure 4.30: The basal area of all woody species function of the fire treatment across the study sites

The basal area was further assessed against fire frequency. The Kruskal-Wallis test results shows that the differences were non-significant ($p=0.901$, $df=3$) across the study sites. Figure 4.31 shows the distribution of basal area according to the fire frequency across the study sites.

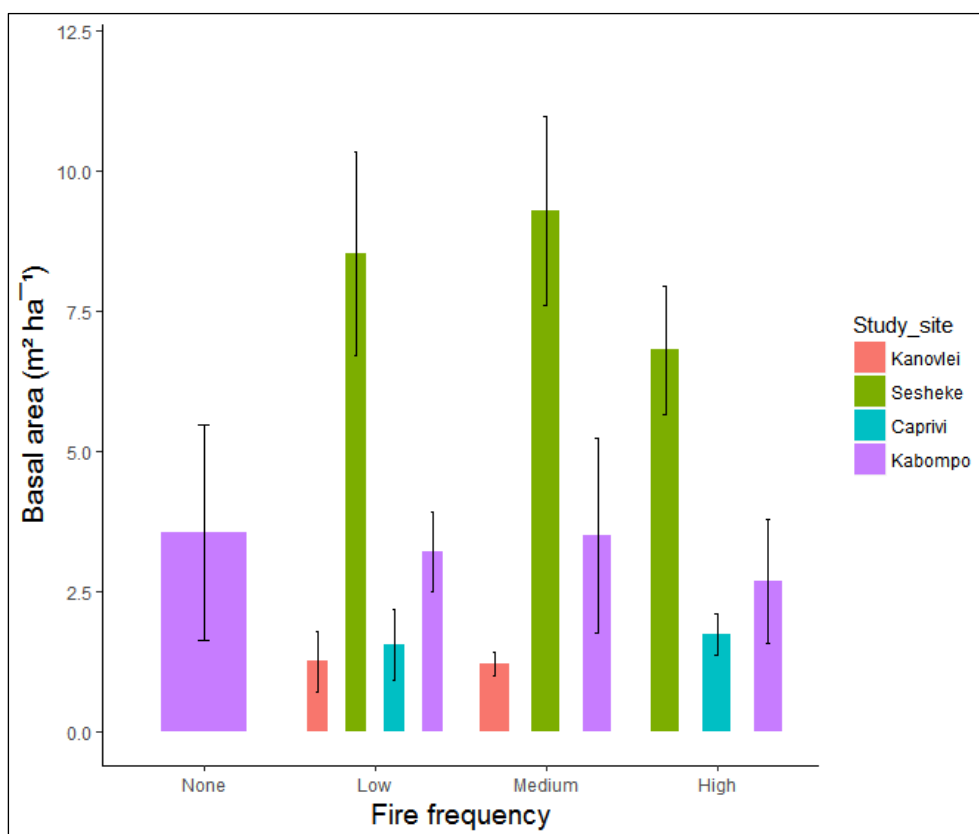


Figure 4.31: The distribution of basal area according the fire frequency across the study sites.

4.4.8. Basal area of selected species

Basal area for each of the selected species across study sites is shown in Figure 4.32. Results show that none of the selected species was recorded at all the four sites.

The basal area of *B. plurijuga* at the Sesheke and Caprivi sites showed significant differences with the Kruskal-Wallis test ($p=0.000$, $df=1$) while the effect of fire treatment was significant ($p=0.042$) at the Sesheke site and non-significant ($p=0.109$) at the Caprivi site. The basal area at the Sesheke site were at $10.9 (\pm 1.1) \text{ m}^2 \text{ ha}^{-1}$ in RB, and $7.4 (\pm 0.9) \text{ m}^2 \text{ ha}^{-1}$ in RU. The basal area at the Caprivi site was $1.0 (\pm 0.2) \text{ m}^2 \text{ ha}^{-1}$ in RB and $3.9 (\pm 2.7) \text{ m}^2 \text{ ha}^{-1}$ in RU.

The difference in basal area for *B. africana* between the Kanovlei and Kabompo sites was non-significant for the site effect ($p=0.085$) and fire treatment effect ($p>0.05$) within the sites. The basal area at Kanovlei was $1.8 (0.5) \text{ m}^2 \text{ ha}^{-1}$ in RB and $2.2 (0.5) \text{ m}^2 \text{ ha}^{-1}$ in RU while at Kabompo site it was at $1.1 (\pm 0.3) \text{ m}^2 \text{ ha}^{-1}$ in RB and $1.1 (\pm 0.5) \text{ m}^2 \text{ ha}^{-1}$ in RU.

The effect of fire treatment on the basal area of *G. coleosperma* species was non-significant ($p=0.649$). There was no test of the site effect since the species was only recorded at the Kabompo site. The mean basal area was $0.9 (\pm 0.4) \text{ m}^2 \text{ ha}^{-1}$ in RB and $1.2 (\pm 0.4) \text{ m}^2 \text{ ha}^{-1}$ in RU.

The Mann-Whitney test on the effect of fire treatment on the basal area of *P. angolensis* was non-significant ($p > 0.05$) at the Kanovlei, Caprivi and Kabompo study sites. The Kruskal-Wallis test on the site effect was significant ($p = 0.000$, $df = 2$). The Holm corrected p-values showed the differences were between the Kanovlei and the other two sites of Caprivi ($p = 0.004$) and Kabompo ($p = 0.023$). The Kanovlei site had mean basal area of $2.7 (\pm 0.5) \text{ m}^2 \text{ ha}^{-1}$ in RB and $2.6 (\pm 0.7) \text{ m}^2 \text{ ha}^{-1}$ in RU. At Caprivi, basal area was $2.8 (\pm 1.4) \text{ m}^2 \text{ ha}^{-1}$ in RB and $0.2 (\pm 0.1) \text{ m}^2 \text{ ha}^{-1}$ in RU and at Kabompo the mean basal area was $0.5 (\pm 0.2) \text{ m}^2 \text{ ha}^{-1}$ in RB and $0.5 (\pm 0.2) \text{ m}^2 \text{ ha}^{-1}$ in RU.

The mean basal area of *S. rautanenii* at Kanovlei site was $0 \text{ m}^2 \text{ ha}^{-1}$ in RB and $0.2 (\pm 0.2) \text{ m}^2 \text{ ha}^{-1}$ in RU. At the Caprivi site was $0.2 (\pm 0.2) \text{ m}^2 \text{ ha}^{-1}$ in RB and $0.6 (\pm 0.6) \text{ m}^2 \text{ ha}^{-1}$ in RU, and at the Kabompo site was $0 \text{ m}^2 \text{ ha}^{-1}$ in RB and $0.3 (\pm 0.1) \text{ m}^2 \text{ ha}^{-1}$ in RU. The basal area was non-significant on the site effect ($p = 0.239$, $df = 2$) and on the effect of fire treatment ($p > 0.05$) within the sites.

The effect of fire treatment on the basal area of *T. sericea* was non-significant ($p = 0.092$) while there was no test on the effect of site since stems with $\text{dbh} \geq 5 \text{ cm}$ were only sampled at the Caprivi site. The mean basal area were $1.1 (\pm 0.7) \text{ m}^2 \text{ ha}^{-1}$ in RB and $< 0 \text{ m}^2 \text{ ha}^{-1}$ in RU.

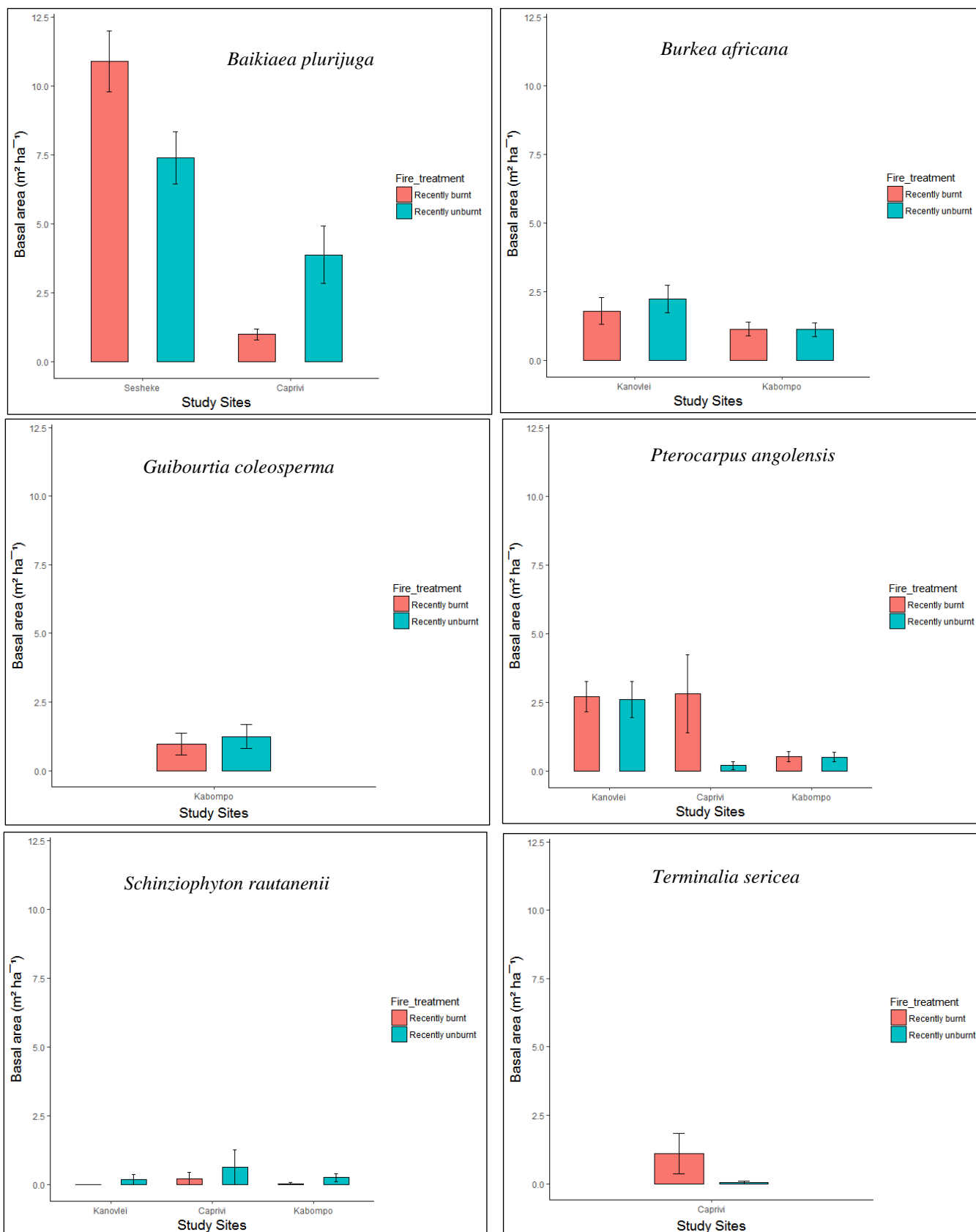


Figure 4.32: The Mean basal area for each of the selected species across the study sites. None of selected species had adult trees at all the study sites.

The Kruskal-Wallis test results showed that fire frequency had no significant (0.901) effect on the basal area of the selected species across the study sites. However, the basal area of the selected species were significantly ($p < 0.0001$) different across the study sites. The exact differences are listed in Appendix 4.11 with their respective Holm corrected p-values. The distribution of basal area for the selected species according to fire frequency is shown in Figure 4.33.

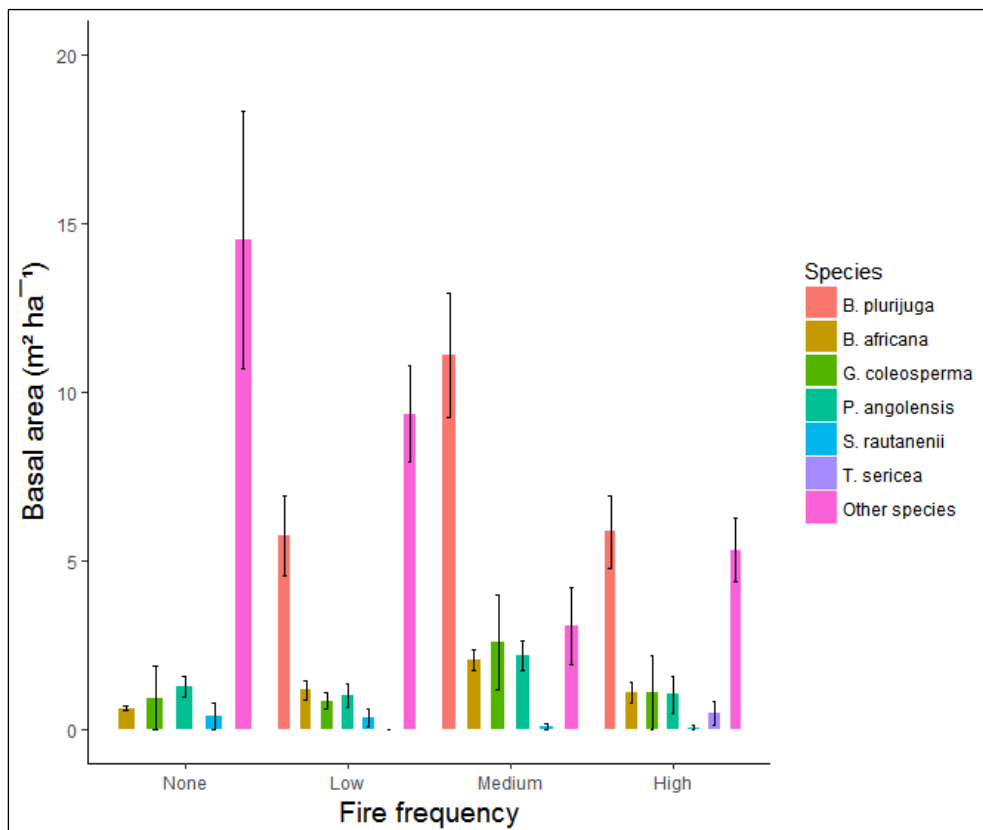


Figure 4.33: The basal area distribution for the selected species following the fire frequency across the study sites.

4.5. Vegetation cover

4.5.1. Canopy cover

Results on assessment of canopy cover with a Bitterlich gauge revealed that crown cover ranged from 20 % at the Kanovlei site to 75.5 % at the Kabompo site. Figure 4.34 shows the mean percentage cover in RB and RU across study sites. The results clearly indicates that the drier site has less canopy cover in relation to the wetter site disregarding Caprivi site which has too much anthropogenic interference.

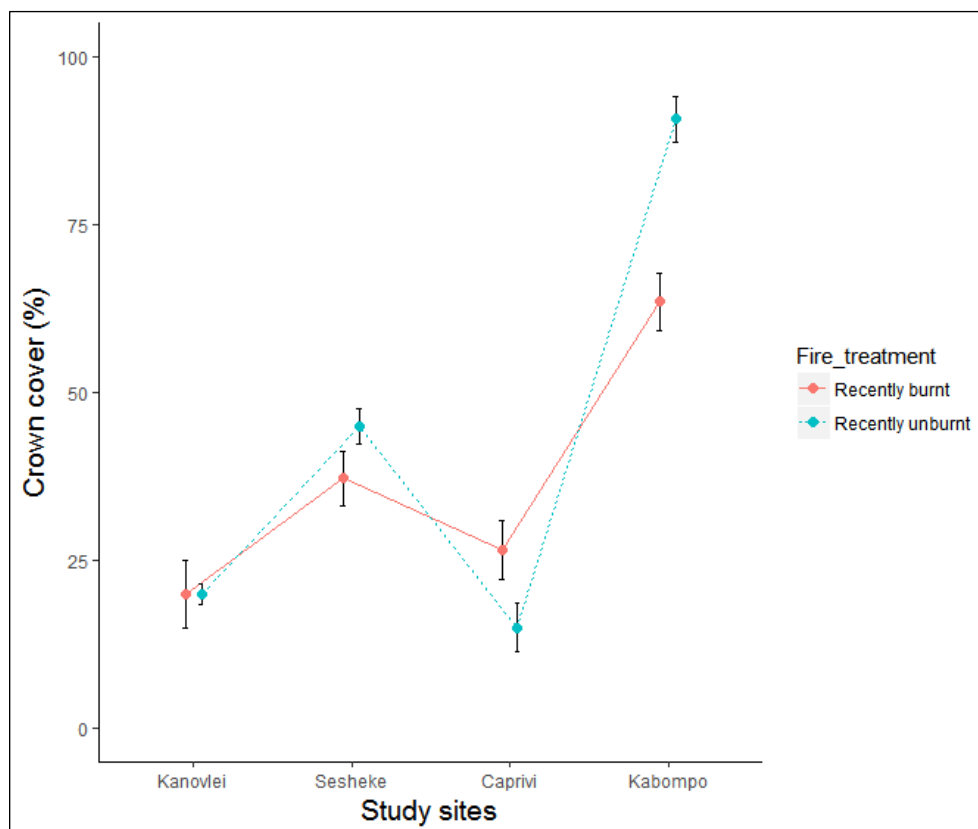


Figure 4.34: The percentage cover of tree crowns on the forest floor in the recently burnt and recently unburnt plots across the study sites.

The interaction effect between fire treatment and site was highly significant ($p=0.000$, $df=3$) on mean percentage cover of tree crowns.

The effect of fire frequency on canopy cover at each of the study site was non-significant ($p=0.064$) whereas the canopy cover at study sites was significantly ($p<0.0001$) different. The post-hoc results on the significantly different canopy cover at the study sites are shown in Appendix 4.12. Figure 4.35 shows the crown cover of woody species on the forest floor function of the fire frequency on the plots across the study sites.

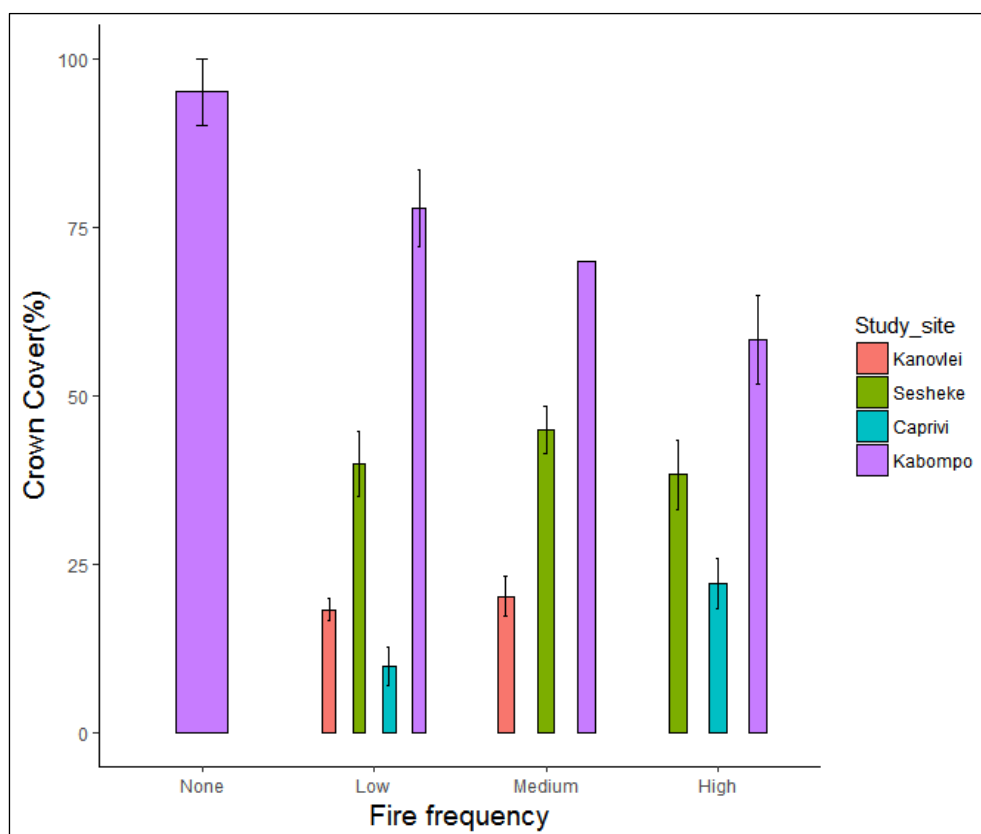


Figure 4.35: The crown cover of woody species on the forest floor following the fire frequency at the plots across the study sites

4.5.2. Substrate cover

Results of the line intercept method are presented in Figure 4.36 for grass, herbs, and shrubs and bare ground cover. Generally, the bigger patches in the forests were covered by bare ground, followed by shrubs, grasses and lastly herbs.

Grass cover ranged from 2.6% at Kabompo site to 19.5% at Caprivi site for RB while in RU the least was at Kanovlei site with 6.5% and the highest was at Kabompo site with 14.6%. The interaction effect of fire treatment with site was highly significant ($p < 0.0001$, $df=3$) on grass cover of forest floor.

Herbs cover accounted for under 10% in both RB and RU across study sites. Cover percentage in RB ranged between 2.7% at Sesheke site and 7.7% at Kanovlei site whereas in RU, coverage ranged from 1.8 % at Caprivi site to 6.8% at both Kanovlei and Kabompo sites. The interaction effect of fire treatment with site was non-significant ($p=0.392$, $df=3$) on herbs cover of the forest floor. The individual effects of fire treatment ($p=0.267$, $df=1$) and site ($p=0.126$, $df=3$) were also non-significant.

The shrubs coverage in RB ranged between 4.6% and 14.5% whereas in RU the range was from 4.4% to 25.1% across study sites. The interaction effect of fire treatment with site was significant ($p=0.009$, $df=3$) on shrubs cover of the forest floor.

As earlier pointed out in Section 3.8.3, any stretch of forest floor that was not covered by live grass, herbs, or shrubs was considered to be bare ground even if it was covered by dead matter such as twigs, leaves, shrubs or herbs. Thus, large patches of the forest floor were recorded as bare ground in all the study sites. The coverage in RB ranged from 51.4% to 88.6% while in RU the range was from 56.3% to 85.7% across study sites. The interaction effect of fire treatment with site was significant ($p=0.024$, $df=3$) on the forest floor that was either complete bare or covered by dead matter.

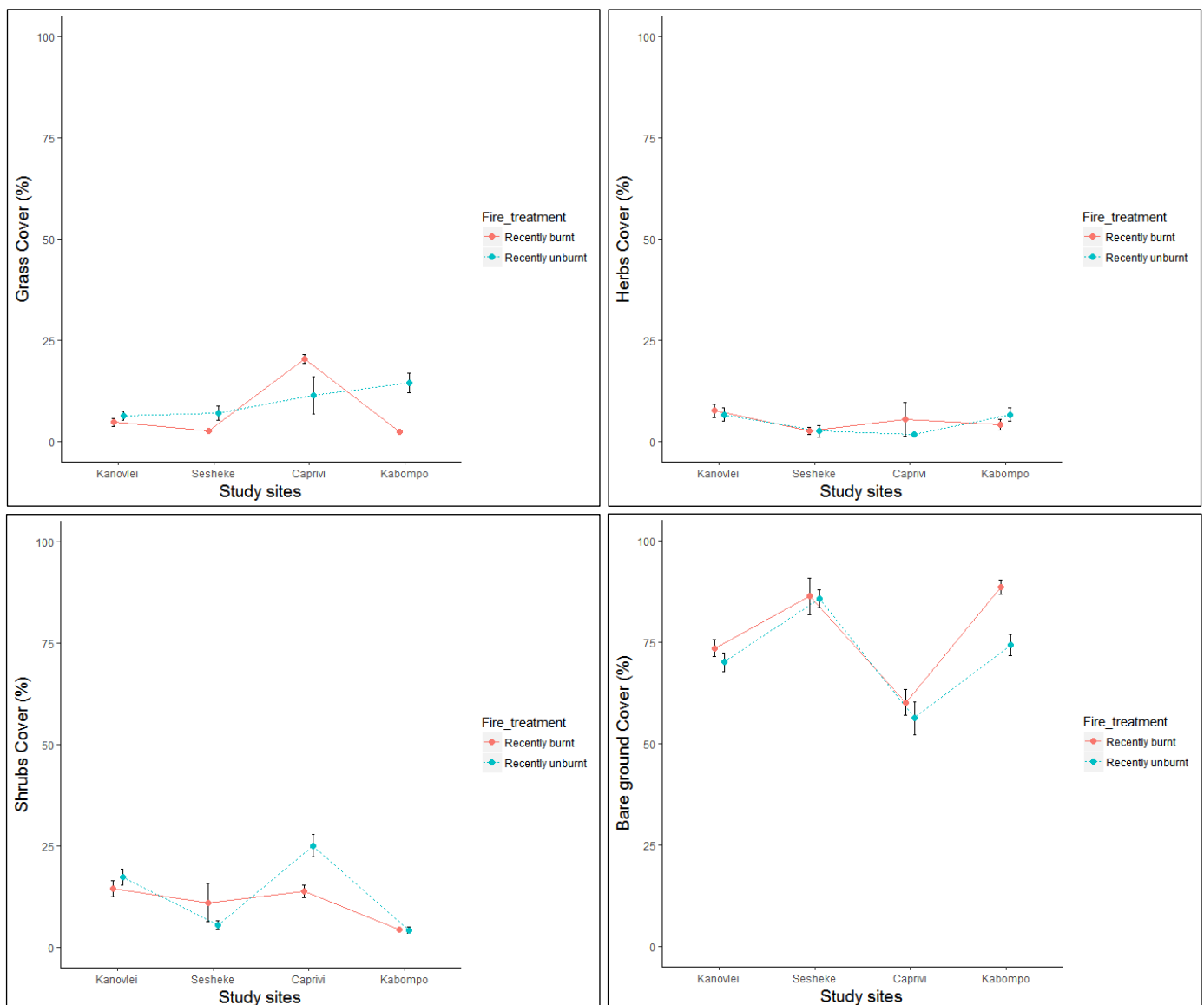


Figure 4.36: The substrate cover on the forest floor in the RB and RU plots across the study sites.

The substrate cover on forest floor was also assessed in light of the fire frequency on the plots. The interaction of fire frequency with substrate was highly significant ($p < 0.0001$) across the study sites. Figure 4.37 shows the substrate cover following the fire frequency.

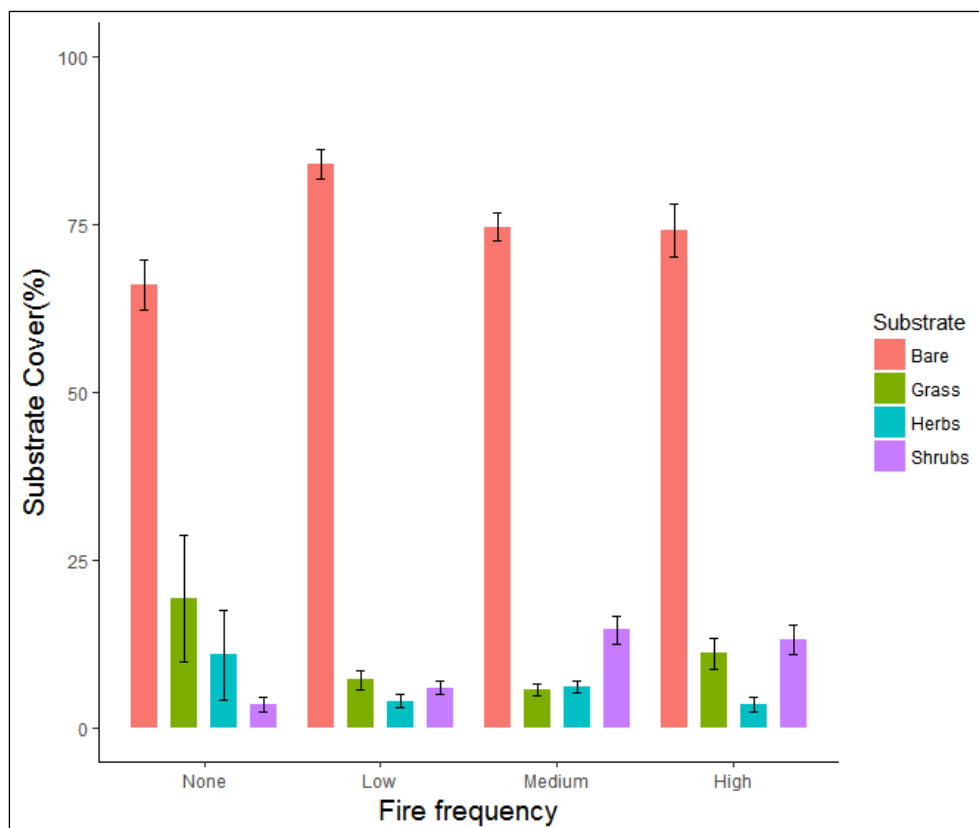


Figure 4.37: The substrate cover on forest floor following the fire frequency at the plots across the study sites.

4.6. Chapter summary

Chapter four has presented the results of the study on the parameters investigated. Many parameters investigated revealed significant differences due to interaction effects, fire treatment or site effects in cases where interaction effect was non-significant. The results showed high species richness of 68 vascular plants represented in 26 families and 54 genera across the study sites. The Shannon diversity indices, ranging between 1.5 and 3.34, revealed a high species diversity across the study sites and at each of the study sites. Regeneration density was significantly different across study sites and also selected tree species have shown good regeneration although regeneration limitation was high. The results have also shown the recruitment ability of the selected woody species indicated by negative OLS regression slopes even though the regeneration limitation was high. The contribution of the selected woody species to community parameters varied from one species to the other and from one study site to the next. *B. plurijuga* stood out among the selected species as the most ecologically

important species with a high importance value index. Generally, basal area was low ranging from $1.2 \text{ m}^2 \text{ ha}^{-1}$ at Kanovlei to $8.2 \text{ m}^2 \text{ ha}^{-1}$ at Sesheke for all woody species.

The effects of fire frequency revealed significant differences on regeneration, mean diameter, mean height, substrate cover with an exception of canopy cover, which was non-significant.

Chapter 5

DISCUSSION

5.1. Introduction

This chapter discusses the results presented in chapter 4. The discussion comprises two parts with Section 5.2 – 5.5 discussing the general results across the study sites, whereas Section 5.6 discusses the selected woody species in the context of natural regeneration, stand structure and their contribution to species community parameters.

5.2. Floristic composition

5.2.1. Species richness and diversity

Species richness, a measure of alpha diversity, is expressed as a number of species in a sample (Whittaker, 1972), was high across the study sites as indicated by the Shannon diversity indices. As shown in Table 4.3, the woody species richness did not differ much between sites and between fire treatments. This was confirmed by the high indices of Shannon, Simpson evenness and equitability. There was, however, a show of equal dominance and evenness by the Combretum species in RB at the Caprivi study site. However, it cannot be said that Kabompo, the site with a highest number of species and higher Shannon index was more diverse than the other sites. The high species diversity can be attributed to diverse habitats that are found in the miombo woodlands (Campbell, 1996). The Shannon diversity index values in this study were similar to those that have been reported by earlier studies within the miombo woodlands in countries such as Angola, Namibia, Tanzania and Zambia. The indices found in these studies ranged between 1.89 and 3.44 (Chidumayo, 1987; Chakanga *et al.*, 1999; Angombe *et al.*, 2001; Maliondo *et al.*, 2005; Kalaba *et al.*, 2013; Mwakalukwa *et al.*, 2014; Jew *et al.*, 2016; Gonçalves *et al.*, 2017). As stated by Magurran (2004), a high diversity Shannon index range between 1.5 and 3.5, although other authors postulated that a Shannon diversity index of greater than 2 is an indicator of medium to high diversity (Barbour *et al.*, 1987). Nevertheless, the upper limit rarely exceeds 5 (Margalef, 1972; Whittaker, 1972). The results of non-significant difference between fire treatment conforms to earlier findings of O'Brien (1993) where species richness did not vary with fire effect but varied with MAP and the length of rainy season. Similarly, Revermann *et al.* (2016), analysed the alpha diversity along the Okavango basin in their study and found high species diversity which decreased southwards, from 65 to 15 species per 10^3 m^2 (i.e. the number of vascular plant species per vegetation plot). Low species richness is associated with intensely disturbed areas compared to mild or moderate disturbed (Gautam *et al.*, 2016). Thus, low species richness could mean incomplete vegetation change (succession) as some species are yet to

return or may never return after a disturbance (Gotosa *et al.*, 2013). In this study, high species richness among RB and RU areas of the study sites did not show pattern of vegetation change as suggested by Gotosa *et al.* (2013).

5.2.2. Species similarity

The multiple-site similarity coefficient of 0.51 showed that the study sites had very few species in common. Hence, the high Shannon diversity index (3.34) across the study sites. As shown in Appendix 4.1, *S. rautanenii* was the only woody species to have been shared by all the four study sites whereas other species were either shared by three or two study sites. The Sørensen coefficient similarity indices also revealed low similarity of species between any two-study sites. The least similarity was between the Sesheke and Kabompo study sites while highest similarity was between the Kanovlei and Caprivi study sites. The relative similarity between Kanovlei and Caprivi study sites was unexpected given the fact that Sesheke study site is geographically closer to Caprivi study site and the MAP at the two sites is non-significantly different, hence it was expected Sesheke and Caprivi study sites to have a high Sørensen coefficient similarity index. This low coefficient similarity index could not be explained by the study but can be assumed that other environmental and non-environmental variables not investigated by this study are highly controlling species occurrence at the study sites. For instance, other studies found certain factors to control the occurrence and distribution of species. Among these factors were altitude and slope in Mensah *et al.* (2016), land use history in Gonçalves *et al.* (2017), climate and soil type in Burke (2006) or water surplus and water deficit in Thornthwaite, (1948). The low Sørensen coefficient similarity indices between the Kabompo and the other three state forests were expected and was in line with other studies on woody cover and species richness (*sensu* O'Brien, 1993; Sankaran *et al.*, 2005; Staver *et al.*, 2011) along a rainfall gradient. Bioclimatic and abiotic factors that influences species distribution at both regional and local scales are complex (Burke, 2006; Ceccon *et al.*, 2006). The Sørensen coefficient similarity indices between fire treatments within the study sites were also high and ranged between 0.67 and 0.87. Similarity coefficients decrease with increase of species observed in one or few sites while it increases with an increase of the number of species observed in several sites (Diserud & Ødegaard, 2007). Hence, the observed lower similarity coefficient index at Sesheke, which had more species that only occurred in RB. It is indicated from the results that fire treatment had minimal influence on species composition and richness between the RB and RU at each study sites. These results collaborates the findings of Nepolo & Mapaure, (2012) and Oyedeji *et al.*, (2016) on the short-term effect of fire on species richness and diversity. However, exposure of vegetation to long-term biennial and dry-season fires reduces woody vegetation cover and species richness (Smit *et al.*, 2010).

5.2.3. Plant community parameters

The species community parameters revealed high variations across the study sites for the tree species that this study focuses on. The results in Table 4.4 to 4.8 shows high variation in species abundance, relative frequency, relative density, relative dominance and importance value index (IVI) for the selected tree species across and within each of the study sites. IVI, a profiling index for the structural role of a species in a stand (Roberts-Pichette & Gillespie, 1999), was used in assessing the importance of the selected species at the study site. IVI has been widely used in the miombo woodlands to describe the composition and structure of the woodlands (Gonçalves *et al.*, 2017; Jew *et al.*, 2016; Kalaba *et al.*, 2013; Maliondo *et al.*, 2005; Mwakalukwa *et al.*, 2014). The top ranked species according to IVI across the study sites are *B. plurijuga*, followed by *P. angolensis*, *B. africana*, *T. sericea*, *G. coleosperma* and lastly *S. rautanenii*. *B. plurijuga* dominated other selected species at both the Sesheke and Caprivi study sites. The domination of the species at the two study sites can be attributed to the location of the two study sites in the *Baikiaea* woodlands where the species is dominant or co-dominant with *P. angolensis*, hence giving the woodlands the general name (Chakanga *et al.*, 1998). At the Kanovlei and Kabompo study sites, the dominant selected tree species was *B. africana*. A possible explanation might be that the dominance at the Kanovlei study site is due to the vegetation type where *B. africana* and *P. angolensis* are the dominant species (Muhoko & Kamwi, 2014). On the other hand *B. africana*'s dominance at the Kabompo study site may be attributed to the species' ability to grow in various woodland types and over a wide distribution range (Coates-Palgrave, 2002; Wilson & Witkowski, 2003) even though the vegetation type at the Kabompo study site is typical of the wet miombo woodland (Chidumayo, 1987a). Although the species studied only presented 10 to 20 % of the stems (except at Kanovlei), they did contribute much more to the basal area. Therefore, this is a confirmation of their importance as potential timber trees. The Kabompo site being in the wet miombo has other species of the *Brachystegia-Julbernardia-Isobertia* genera dominating (Chidumayo, 1987a). In this study *B. spiciformis* with a basal area of $0.82 \text{ m}^2 \text{ ha}^{-1}$, *B. longifolia* ($0.74 \text{ m}^2 \text{ ha}^{-1}$) and *J. paniculata* ($0.74 \text{ m}^2 \text{ ha}^{-1}$) had a density of 21 stems ha^{-1} , 17 stems ha^{-1} and 21.4 stems ha^{-1} , respectively. However, considering the high anthropogenic disturbance at Caprivi and the low mean diameter at Kabompo, it can be confirmed that both sites are dominated by early succession species where *T. sericea*, a typical pioneer species (De Cauwer *et al.*, 2016), was also expected to dominate but this was not the case. According to Lamprecht (1989), equal or similar IVI in the stand is an indication of same or at least similar composition and structure of the stand, including site requirements and comparable dynamics among species. In this study, IVI for selected species varied highly at the study sites. This indicates a diverse species composition with each species having a different important role at the study site (Wale *et al.*, 2012).

5.3. Natural regeneration

The assessment of natural regeneration across study sites also included many shrubs as natural regeneration. Natural regeneration was defined by a dbh less than 5 cm. As many shrubs do not get much bigger than 5 cm dbh, for some shrub species this could have included not only the regeneration but the whole population. The results in Figure 4.2 showed that fire treatment influenced the density and composition of natural regeneration along the rainfall gradient. The interaction plot (Figure 4.3) clearly shows the interaction pattern within sites. What stood out from the figure is that fire treatment had no effect on natural regeneration at the Kanovlei site, which has the lowest MAP value. The results conform to findings of Sankaran *et al.* (2005) where maximum realised woody cover increased with MAP in semi-arid areas ($< 650 \pm 134$ mm) but shows little relationship with fire return interval. A variety of species dominated the density of regeneration at each of the study sites in RB and RU, as shown in Table 4.11. The variation across the study sites was no surprise because plant species richness is expected to vary in Southern Africa based on climate, and also from west to east and decrease from north to south based on MAP (O'Brien, 1993). Hence, a clearer pattern from a dry site to wetter site could have come out in Figure 4.2 had it not been for Caprivi site's exposure to high anthropogenic disturbances.

The Sesheke and Caprivi sites showed higher density of regeneration in RB than RU. The observed higher regeneration densities in RB could be attributed to fire's positive impact on stimulating reproductive buds and dormant seeds in the soils (Bond & Van Wilgen, 1996; Oyediji *et al.*, 2016). The differences can also be attributed to the open patches created by the disturbance and death of mature trees (Borghetti & Giannini, 2005) which are then taken up by pioneering species (Boaler & Sciwale, 1966; Gotosa *et al.*, 2013). Whereas the low densities in RU could be as a result of the competition seedlings face from grasses and other substrates for resources (Seifert *et al.*, 2014). The higher seedling density in RU at Kabompo can be attributed to the influence of canopy cover. The Kabompo site presents typical miombo woodland with species that require canopy gaps to establish and recruit individuals (Syampungani *et al.*, 2016). However, in the absence of canopy gaps, species that are shade tolerant and have the ability to establish in undergrowth are better suited to establish in wet miombo (Chidumayo, 1997). This could mean that the canopy cover at the Kabompo site is already closed enough to favour the later succession species. The tree species contributed less 40 % to the regeneration density at Kanovlei and Sesheke sites while at the Caprivi and Kabompo sites they contributed more than 50 % compared to the shrub species. This shows that shrub species reacts more positively to fire treatment and the opener conditions fire created at Kanovlei and Sesheke sites. Overall, fire frequency significantly affected the regeneration density (Figure 4.4), associating higher

regeneration density with low to medium frequency across the study region. In other studies similar patterns were found indicating high regeneration density in low to moderate burned plots and also low regeneration density in high-intensely burnt plots (Kennard *et al.*, 2002; Gautam *et al.*, 2016) and plots with no recent fires (Chidumayo, 1988). The possible explanations for these trends, firstly could be that low to medium fire frequency enhances regeneration by overcoming seed dormancy, improving light penetration, moisture availability and seed intensity. Secondly, the disturbance to existing seedlings is not severe to damage or hinder regeneration (Wiafe, 2014). Thus, higher regeneration will result from either seed banks in the soil, through coppicing (suffrutices) or seed that is dispersed after the fire (Kennard *et al.*, 2002).

The density and composition of natural regeneration were dominated by the species that are associated with fire disturbance across study sites in both RB and RU, similar to findings of Ribeiro *et al.* (2008). Among others were the shrub species of *B. massaiensis*, *B. petersiana*, and the pioneering tree species which are fire tolerant such as *F. obovata*, *B. bequaertii*, *D. condylocarpon*, *Combretum* species, and *X. odoratissima*. A few tree species only occurred in the RU sites, among others were *Albizia adianthifolia*, *Enlerophytum magalismontanum*, *C. herereonse*, *Croton gratissimus*, *Dialium engleranum*, *Ficus sycomorus*, *Maprounea africana*, *Rhus tenuinervis* and *Uriastrum hexaloboides*. Geldenhuys (1977), listed *D. engleranum* among the tree species that regenerated well under complete protection from fire. The tree species that only occurred in RU for this study may be fire sensitive probably excluding *M. africana*, which was classified as fire tolerant by Trapnell (1959).

The species that dominated regeneration density also had the least regeneration limitations within the study sites (Table 4.11). The regeneration limitation percentage ranged from 5% to 95% at the respective study sites. This showed that the top three tree species had the presence of regeneration in at least 30% of the plots sampled. The tree species were *O. pulchra*, *Combretum* species and *T. sericea*. However, the regeneration limitation varied for each species among the RB and RU. It also varied from site to site such that one species may have low limitation in one site but high limitation in the next site. Generally, species are strongly regeneration-limited due to few adult individuals and limited dispersal mechanisms. Exceptions can be observed among the pioneer species that produce very high number of seeds and are able to disperse them over long distances (Muller-landau *et al.*, 2002). In this study, there were only a few woody species observed that have low regeneration limitation. These included among others *Combretum* species, *O. pulchra*, *S. rautanenii* and *D. condylocarpon* that were able to regenerate over a wide range of environmental conditions ranging from semi-arid to wetter conditions. Environmental factors such as soil type and dynamic factors such

fire, and species traits generally influence species composition at any site (Backéus *et al.*, 2006). Additionally, these species are associated with being pioneer or early succession species due to their nature of either being fire-tolerant and/or shade-intolerant (Stromgaard, 1986).

5.4. Stand structure

5.4.1. Correlation of diameter and height of all woody species

Diameter at breast height (dbh) and height relationships are often used in forest stand characterisation (Mugasha *et al.*, 2013; Sumida *et al.*, 2013). In this study, correlation was used to determine the dbh-height relationship, the strength of relationship and the predictor equation. As stated under Section 3.9.5, of the three methods used to derive the best model for the correlation of dbh and height, the linear model performed better than polynomial and exponential models on this data set. The method in this study was different to methods used in similar studies who used non-linear model (Mugasha *et al.*, 2013) and linear mixed model (Sumida *et al.*, 2013).

The diameter and height correlation was significant on the pooled data, between RB and RU across the study sites (Table 4.12), as well as at each of the study sites (Figure 4.13). All the derived equations showed positive slopes implying that height was expected to increase with a corresponding unit increase in dbh. The strength of the relationship (R^2) showed high variation at site level, ranging from 0.216 at Kanovlei study site to 0.777 at Kabompo study site, and also among the RB and RU across the study sites. The strength of the relationship for the pooled data across the study sites was average, neither strong nor weak. Mugasha *et al.* (2013) conducted a countrywide study of Tanzania's four major forest types that included miombo woodland. In their study, they included trees with dbh starting at 1 cm and their models produced R^2 between 0.52 and 0.92. Some of the strong R^2 they found could be attributed to the methods they used where species were grouped in six groups based on tree allometry that reduced the variations in the model unlike running relationship model based on forest type. They concluded that species diversity contributes to low performance of forest based models while grouping species highly improves the model. The results from this study can be explained with the work of Sumida *et al.* (2013) who concluded that trees growing vigorously display almost linear increase in both dbh and height whereas short trees with declining dbh growth show height to be asymptotic. Thus, since in this study there is high species diversity across the study sites, it contributed to the strength of the relationship not being strong. The strength of the relationship at the Kabompo study site could partly be explained by the concept of trees growing vigorously to reach the canopy layer due to the nature of the closed forest. Whereas the weakness of the relationship at the Kanovlei study site could be due to the open type of the canopy where trees invest more resources into the stems than growing tall as there is less competition for light. However, in all cases, dbh

increase only contributes partially to explaining the increase in height as there are other factors unexplained in the variations, which play a part in the height growth of the trees. These could be soils, water deficit, stand density, competition for nutrients and other factors that were not investigated by this study.

5.4.2. Mean diameter and mean height of all woody species

The Caprivi study site was deliberately left out of the assessment of mean dbh and mean height. This was in order to observe the trends of these parameters from a semi-arid region to a wetter region with minimal anthropogenic disturbances than those encountered at the Caprivi state forest. The Caprivi state forest has been encroached in the recent years compared to the period when the inventory was conducted by Chakanga *et al.*, (1998). The three common anthropogenic factors influencing negatively on the vegetation of the state forest were illegal harvest of mature *P. angolensis* and *B. plurijuga*, clearing vegetation for agricultural fields, and intense grazing of livestock.

Mean diameter differed significantly among the three study site of Kanovlei, Sesheke and Kabompo. The largest differences were between the Kanovlei (26.0 ± 0.9 cm) and Sesheke (32.7 ± 1.9 cm), as well as the Kanovlei and Kabompo (29.5 ± 0.7 cm) study sites, while there was no statistical difference between that of the Sesheke and the Kabompo study sites. The assessment of mean diameter against fire treatment only showed a significant difference in mean diameter at the Sesheke study site ($p=0.004$), which recorded 38.1 cm in RB and 27.3 cm in RU, as shown in Figure 4.26 (a). This difference could be attributed to the composition of the forest stand after a fire. In the RB, most individuals that were left standing were from the larger diameter size classes especially for *B. plurijuga* (pers. obs.) The mean diameter differences in the RB and RU were non-significant ($p>0.05$) at the other two study sites. Further assessment of mean diameter against fire frequency at the study sites showed non-significant differences. This could mean the trees are able to withstand the subsequent fires after surviving the first fire.

Mean height differed significantly with fire treatment and with the interaction effects between fire treatment (Figure 4.25 (b)), or fire frequency (Figure 4.28) and site. The mean heights were 5.8 m at Kanovlei, 12.3 m at Sesheke and 11 m at Kabompo study site. The significant interaction effect could result from the fire causing increased soil nutrient availability to surviving trees in the short term (Trapnell *et al.*, 1976; Fisher & Binkley, 2000; Singh *et al.*, 2017) and reducing edaphic and light competition (Seifert *et al.*, 2014) by burning the grass, herbs, shrubs and small trees at the Kanovlei and Sesheke sites. Thus, the surviving trees grow vigorously and tall to escape canopy fires before the next fire occurs. The results at Kabompo site could be attributed to the canopy dominant species, the fire sensitive typical miombo species (Trapnell, 1959), which grow fast to escape fire damage.

The mean height recorded at Kanovlei study site for this study was lower than that of Muhoko and Kamwi (2014) who reported a mean height of 9.2 m. The difference in the mean height reported for this study and that of the inventory report could be the difference in the sampling intensity for the two studies.

5.4.3. Basal area of all woody species

Basal area was sensitive to site effect and showed significant ($p < 0.0001$) differences among the study sites. As shown in Figure 4.31, the basal area at the Sesheke study site was significantly higher from those at the other three study sites. The basal area at the Kanovlei, Caprivi and Kabompo did not show significant differences among these three study sites. In other studies, basal area was found to increase from arid to mesic region along a rainfall gradient (Shackleton & Scholes, 2011) and climatic zones (Mensah *et al.*, 2014). However, this was not the case for this study as Caprivi and Kabompo recorded a low basal area compared to that of Sesheke. Perhaps the most unexpected finding is the low basal area recorded for this study compared to the results from other studies in the region, including the miombo woodlands. The basal area reported in other studies focused on areas under some kind of protection such as forest reserve, game controlled area, nature reserve and research facility (Luoga *et al.*, 2002; Banda *et al.*, 2006; Shackleton & Scholes, 2011; Schelstraete, 2016) and was high, ranging between 7.4 and 24 m² ha⁻¹. A possible explanation to this disparity in basal area between this study and the cited studies could be that the state forests in this study have not recovered fully from previous disturbances. Hence, they have less mature trees in the larger size classes since the presence of these individuals contribute to high basal areas (Banda *et al.*, 2006a). The fire treatment (Figure 4.30) nor fire frequency (Figure 4.31) did, however, not have an effect on basal area at Kanovlei, Caprivi and Kabompo but a significant ($p = 0.04$) difference was observed at Sesheke study site based on fire treatment. Schrelstraete (2016) also found non-significant differences in basal area at a Namibian state forest based on fire frequency.

Luoga *et al.* (2002) found a similar scenario in the public lands which were heavily harvested to have a low basal area (3.1 m² ha⁻¹) compared to the forest reserve (9.8 m² ha⁻¹). In the study of Schelstraete (2016), a large difference in basal area between state forest and community forest was observed but could not be explained by the recent fire history.

5.5. Vegetation cover

5.5.1. Crown cover

Estimating crown cover of mature trees and shrubs can provide a way of making inferences on broad vegetation dynamics (Friedel & Chewings, 1988). Crown cover assessment is important in forest

stratification and prediction of forage on rangeland (Isango, 2007). In this study, crown cover was found to range from 20% to 80% at the dry site and the wetter site respectively. This indicated clearly that the trees were less dense at Kanovlei study site and progressively became denser at Kabompo study site. The interaction of the fire treatment with site was significant. Results in Figure 4.34 shows that crown cover was higher in RU than RB. This could imply that fire affected much of the crowns of larger shrubs and small trees in the under story of the RB while the smaller trees in RU have escaped the fire effect on crown by quickly establishing themselves beyond fire reach of taller than two metres (Bond & Keeley, 2005; Mapaire, 2013). Although the crown cover at Caprivi site shows to be lower than Sesheke study site, it is a result of excessive illegal harvest of mature *B. plurijuga* and *P. angolensis* trees at Caprivi study site. It is evident from the results that crown cover is expected to increase gradually from the dry site to the wetter site. Fire frequency was non-significant ($p=0.064$) and had minimal effect on crown cover at the study sites. This result suggest that fire damages the crown but that the damage is temporarily where the understory recover after a few years. Generally, it has been agreed that exposing vegetation to long-term biennial fire frequencies reduces woody cover (Smit *et al.*, 2010); the non-significant effect of fire frequency on crown cover could result from the fact that trees will have outgrown the range where subsequent fires are able to burn the canopies when the fire return. Another possibility could be that the variations caused by anthropogenic disturbances and wood harvesting has increased the variability, making it more difficult to show statistically significant differences (Ben du Toit, pers. comm.).

5.5.2. Substrate cover

The herbs had the lowest cover on the forest floor with a coverage ranging from 1.8 to 7.7 %, while grasses (2.6 – 14.6%) and shrubs (4.4 – 25.1 %) had higher values across the study sites. Most of the ground was bare ranging between 51.4 % and 88.6 % across study sites. The substrates cover on the forest floor in function of fire treatment (Figure 4.37) or fire frequency (Figure 4.38) was highly significant for the grasses, shrubs and bare ground but not for the herbs. In an assessment of short-term effect of fire on herbaceous composition, diversity and grass biomass production, Nepolo & Mapaire (2013) found that fire significantly influenced herbaceous and grass cover. This could explain the low cover of grasses and herbs in this study. The results also contradicts the findings of others who found fire to promote grass growth by removal of accumulated moribund grass biomass (Lohmann *et al.*, 2014). In the short-term, there was more regeneration of woody and shrub species at Sesheke site (Figure 4.2) taking advantage of the space created by the reduced herbs and grasses, which are unable to mount competition for available resources. Thus, Sesheke exhibited a high regeneration density in areas that recorded low herbs and grass coverage. The significant effect of fire frequency on the substrate cover on the forest floor is evident of high reduction of herbs, grasses

and shrubs on the forest floor, for the first two years, leaving it mostly covered with dead material or completely bare (Figure 4.36).

5.6. Selected species

Results for selected species in this study have shown different trends for regeneration, population structure, and the species contribution to community parameters. Some of the results are compared to inventory reports of Kanovlei State Forest and Caprivi State Forest.

5.6.1. *Baikiaea plurijuga*

The species showed significantly higher density at Sesheke than Caprivi study sites and also between the RB and RU of the two study sites (Figure 4.5). The differences in density between the study sites could be attributed to the species being sensitive to fire and water availability (Burke, 2006; Ngoma *et al.*, 2015) including other local factors such as soil nutrients. The unanticipated finding was that the regeneration density was higher in RB than RU at both sites. The species is fire sensitive and it is expected to perform better in fire exclusion area (Geldenhuys, 1977). The species regeneration is highly influenced by fire, competition, frost, browsing and rodents (DFSC, 2001). Hence, the difference between RB and RU could have resulted from fire having taken out competition in the RB, as the species is easily out competed by shrubs and grasses, while in RU the species may have been browsed on since livestock and wildlife prefer them and also been eaten by rodents (DFSC, 2001). The regeneration density in the RB at the two study sites support the previous figures recorded for the forest inventory of the Caprivi state forest which was at 290 individuals per hectare (Chakanga *et al.*, 1998). The regeneration density of the species also showed that it was significantly affected by the fire frequency. However, it was difficult to interpret the results of the function of fire frequency the regeneration density of the species.

B. plurijuga displayed a good ability to recruit new individuals in the population at Sesheke and Caprivi study sites. The recruitment ratio of seedlings to saplings were > 1 for both study site (Table 4.8), an indicator of good recruitment (West *et al.*, 2000; Mwavu & Witkowski, 2009). The recruitment of the species, as indicated in Table 4.9, was also confirmed from height SCD ordinary least-square (OLS) regression, which showed a negative slope for the Caprivi study site, which meant good recruitment and ability to survive to the next life stage. The positive slope at the Sesheke study site is an indicator of disturbed and unstable recruitment (Obiri *et al.*, 2002). The height SCD in Figure 4.14 shows the species to have a population with episodic regeneration. The population of the species has been known to be regulated by episodic recruitment because successful recruitment in the next

size classes does not occur every year. It may take 1 to 2 years or 2 to 3 years for the species to move from seedling to sapling or sapling to juvenile stage (Childes, 1984).

The species displayed a high regeneration limitation at Sesheke and Caprivi study sites where mother trees were present. Regeneration was absent in 94.4% of the 36 plots at Sesheke and in 70% of the 20 plots at Caprivi. This rather surprising result may be related to the species sensitivity to fire and water availability as earlier alluded to. It is also possible that these results merely reflect a sample site selection effect.

The results on the population structure of the species revealed different scenarios at Sesheke and Caprivi study sites. The correlation of dbh and height at both study sites is positive but with a weak strength of the relation (Figure 4.12). At Sesheke study site, only 40.9% of the variations are accounted for whereas at Caprivi study site an insignificant 0.8% of variations are accounted for. The results have indicated that height growth is much faster at Sesheke and the maximum height is much higher. Hence, the differences in height growth at Caprivi are small and are more difficult to explain. There were variations in the dbh SCD (Figure 4.20) for species at the study sites. The SCD showed a bell shape with more individuals in the large size classes than in the smaller size classes at Sesheke, whereas at the Caprivi study site was more of a bimodal J-shaped population structure. The population structure have resulted from unstable regeneration and lack of rejuvenation in the next size classes over time. The bell-shaped population structure at Sesheke coupled with positive OLS regression slope is an indication of disturbance caused by anthropogenic and climatic factors (Traoré *et al.*, 2012). The lack of reverse J-shaped population structures of the species at the two sites shows that the population lacks stability. Theoretically, the population can be said to be unhealthy or lacking ability to expand (Sop *et al.*, 2011).

The mean dbh of the species (Table 4.13) was non-significantly different whereas the mean height was significantly different at the two study sites. Chakanga *et al.* (1998) reported a mean height of 10.7 m for the species in the Caprivi state inventory, whereas in this study at Caprivi mean height was 9.3 ± 0.5 m and at Sesheke was 14.2 ± 0.3 m. The observed significant differences in basal area (Figure 4.33) between the study sites, and in the RB and RU at Sesheke could be attributed to the anthropogenic and the fire effects affecting the species at the study sites. Thus, most individuals are left in larger size classes after a fire especially at the Sesheke study site while mature individuals have been legally or illegally harvested at Caprivi study site (per. obs.). Large basal area indicates presence of mature trees in larger size classes (Banda *et al.*, 2006a) while small basal area could indicate heavy harvesting of species (Luoga *et al.*, 2002; Schelstraete, 2016).

The structural analysis of the selected woody species are represented by the relative values of frequency, density, dominance and important value index (IVI) in Tables 4.4 – 4.7. The structural role of *B. plurijuga* in the stand shows that the species dominated other selected species with an IVI value of 44.8% across the study sites. The species also dominated at Sesheke and Caprivi study sites where it was ranked first. This is the reason vegetation at Sesheke and Caprivi state forests are referred to as *Baikiaea* woodlands (Chakanga *et al.*, 1998; Fanshawe, 2010).

5.6.2. *Burkea africana*

The results on the regeneration density of the *B. africana* showed that the species was regenerating at Kanovlei, Caprivi and Kabompo study sites (Figure 4.6). The regeneration density at the Caprivi study site was significantly different on fire treatment in RB and RU. The significant differences in the RB and RU were not expected since the species is fire resistant and occur on a wider range of altitudes and habitats (Mendelsohn & el Obeid, 2005; Wilson & Witkowski, 2003). However, the higher density in RB could be attributed to the species' ability to take up space created after a fire. *B. africana* displays characteristics of early succession species (De Cauwer *et al.*, 2016). The regeneration density of the species in this study ranged from 14 individuals per hectare in RU of Caprivi to 422 individuals per hectare in RU of Kabompo. The results differed highly with the findings of Muhoko & Kamwi (2014) who recorded over 6 000 individuals per hectare at Kanovlei state forest. Nevertheless, the study results corroborates the inventory results of Chakanga *et al.* (1998) and Angombe *et al.* (2001) on regeneration density for the species at Caprivi state forest and M'kata community forest near Kanovlei state forest, respectively. The huge difference between the study results and that of Muhoko & Kamwi could be attributed to the difference in the sample size and intensity.

The species showed good recruitment as illustrated by the high seedling to sapling ratios (Table 4.8) and negative OLS regression slopes (Table 4.9) at Kanovlei and Kabompo study site. At Caprivi, recruitment is hindered since the recruitment ratio was indicated to be less than 1 even though the OLS regression slope was negative for the species. The height SCD (Figure 4.16) for species varied and were significantly different at the study sites. The SCD at Kanovlei resembled the classical J-shape indicating less survival and stability of juvenile individuals in the next size classes (Obiri *et al.*, 2002). The SCD at Caprivi was flat and gapped. The population at this site could be said to be lacking rejuvenation and declining in recruitment (Condit *et al.*, 1998; Lykke, 1998). The fewer individuals in the smaller size classes at Kanovlei and Caprivi could also been contributed by the effect of fire (Rutherford, 1981) and the damage by livestock and wildlife (DFSC, 2001). Thus, the species is able to regenerate but is unable to recruit successfully to the next size classes. The SCD at Kabompo was

a reverse J-shaped indicating a good recruitment, health and stability of juvenile population in the next size classes.

Regeneration limitation for the species was moderately high across the study sites at 74.6% of the 130 sample plots. Generally, the regeneration limitation was higher in the RB than RU at Kanovlei and Kabompo sites while at Caprivi it was higher in RU. This could mean that regeneration occurrence was negatively affected by recent fire at Kanovlei and Kabompo even though adult trees have high fire tolerance (Burke, 2006). The results at Caprivi could be due to the effect of livestock and wildlife browsing and tramping on the seedlings. However, it is possible that these results have been confounded by the species passive seed dispersal mechanism (Wilson & Witkowski, 2003).

The species showed a positive correlation between dbh and height at Kanovlei and Kabompo study sites (Table 4.13). There were no adult trees recorded at Caprivi, hence no results for the population structure for the species at this site. De Cauwer *et al.* (2016), explained that trees with dbh over 20 cm are relatively old which lead to natural self-thinning of the population and possibly explains the absence of mother trees at Caprivi site. The relationship strength at Kanovlei was weak implying that height growth is much slower at the site since the forest type is open. At Kabompo, 70% of variations in the correlations were accounted for. The density of the trees at these sites could be the contributing factor to the correlation results. The trees in the open forest such as that of Kanovlei do not invest many resources in the height compared to that of Kabompo where they require to compete for light and break into the canopy level as fast as possible. The results of the diameter SCD (Figure 4.21) showed that the species had more individuals in the smaller and medium size classes than the larger size classes although they lacked the reverse J-shape. The OLS regression slopes were negative and in the case of Kanovlei it was significantly negative (Table 4.13). The results indicated a healthy population and stable rejuvenation of the species in the next size classes. Thus, it shows the species the ability of population structure to expand and good recruitment potential for the species (Sop *et al.*, 2011). The significant differences in mean diameter and mean height of the species at Kabompo (higher) and Kanovlei (lower) study sites are due to the difference in forest types behavioural characteristics of the forest stands as explained earlier on dbh-height relationships. The fire frequency significantly affected the mean diameter and the mean height of the species. The species shows positive response in diameter and height with high fire frequency when seemingly there is no difference without fire or low fire frequency. This could be because fire eliminates the fire sensitive competitors hence providing the species with more resources for growth. The mean height at Kanovlei in this study (5.4 ± 0.3 m) differed with that of Chakanga *et al.* (1999) and Muhoko & Kamwi (2014), who reported 9.7 m and 9.0, respectively. A possible explanation for this difference

may be due to the lack of mature and tall individuals recorded in this study. The basal area (Figure 4.33) for the species was non-significant and quite small at both sites. Several factors could explain this observation. Firstly, it could mean that the species was heavily harvested since it is good for commercial timber (Barnes *et al.*, 2010). Secondly, it could be that the species is yet to recover fully from the disturbance, hence the lack of mature trees in the larger size classes (Banda *et al.*, 2006a), and thirdly, the species may be displaying the true characteristics of an early succession tree (De Cauwer *et al.*, 2016).

The species contribution to community parameters showed that it was second in abundance among the selected species across the study sites, as shown in Table 4.4. *B. africana* was the most abundant at Kanovlei and Kabompo study sites. The structural analysis of the selected woody species ranked *B. africana* as the third most important selected tree species across the study sites with an IVI of 14.9%. The species was ranked second at Kanovlei and first at Kabompo among the selected species. Other authors have shown importance of the species by its contribution to community parameters in the areas where it occurs (De Cauwer *et al.*, 2016; Strohbach & Petersen, 2007).

5.6.3. *Guibourtia coleosperma*

The regeneration distribution of *G. coleosperma* was limited across the study sites. As shown in Figure 4.7, the species was only recorded in the RU at Kabompo with 578 individuals per hectare. This observation was contrary to expectation since the species is known to grow on deep Kalahari sands and in association with *B. africana* and *P. angolensis* (Mendelsohn & el Obeid, 2005). The regeneration density was found to be affected by the fire frequency at the Kabompo site. The species high sensitivity to water deficit and fire (Burke, 2006; Geldenhuys, 1977) possibly explains the findings of this study. The fire exclusion highly favours regeneration while presence of fire is detrimental to the species (Geldenhuys, 1977). In addition, localised factors such as soil type (arenosols) and dispersal mechanism (Burke, 2006) could be limiting the distribution and regeneration of the species or the species is threatened by the fire frequencies in combination with droughts in the drier parts of its distribution range (Vera De Cauwer, pers. comm.). The regeneration density for the species in the inventory of Caprivi state forest was also low at four individuals per hectare. Hence, the results of this study are consistent with literature on the patchy distribution of the species on the Kalahari sands.

A seedling to sapling recruitment ratio of greater than one is an indicator of a recruiting species (Mwavu & Witkowski, 2009; West *et al.*, 2000). Therefore, *G. coleosperma* showed good recruitment at Kabompo study site in Table 4.8. The negative OLS regression slope in Table 4.9 also confirmed the species recruitment at the site. Figure 4.16 shows the height SCD with missing gaps in certain

size classes. The absence of individuals in the some of the size classes could be due to the impact of fire on the species as earlier mentioned. This shows the species inability to stabilise the population structure especially in the juvenile classes. Thus, the species is able to recruit new individuals in the population but struggle to survive to the next stage of life. Geldenhuys (1977) pointed out that fire exclusion boosts the species regeneration, which in turn may improve the species recruitment potential. However, this result for the species at Kabompo point to a declining population.

The maximum regeneration limitation shown in all plots except for the RU at Kabompo is a clear indication of the species' patchy distribution across the study sites. The species is highly influenced by local and regional factors (see Burke, 2006). The fires could have exacerbated the observed regeneration limitation over the years and species has never been able to survive them. Another possible explanation could be due to the absence of mature trees to propagate seeds for new regenerations especially at the study sites where maximum regeneration limitation was recorded.

The correlation of dbh and height was significantly positive (Table 4.13). The strength of the relationship was strong accounting for 64.6% of the variations towards the height. This relationship may partly be explained by the fact that *G. coleosperma* is a co-dominant species in vegetation where it occurs (Scholes *et al.*, 2002; Sheuyange *et al.*, 2005). Diameter SCD in Figure 4.22 showed that the species had no individuals in the smaller size classes and there were few individuals in the medium and larger size classes. The flat and positive OLS regression slope (Table 4.9) meant that the species is lacking rejuvenation in the sapling and juvenile size classes (Condit *et al.*, 1998; Lykke, 1998). Persistence of the situation could lead to population decline. Both the mean dbh and mean height, in Table 4.14, for the species were non-significant between the RB and RU. The overall mean dbh (52.3 ± 6.6 cm) at Kabompo was a clear indication of an aging population. The mean height of 15.3 ± 1.2 m was higher than 11.9 m reported by Chakanga *et al.* (1998) at Caprivi. This result was expected considering the difference in MAP at the Caprivi and Kabompo study sites. However, even though the species recorded a large mean dbh, the basal area was small (Figure 4.33). This inconsistency may be due to the species over exploitation for its high commercial value and high value for local use in form of poles. The species is currently exploited commercially in Zambia (IRDNC, 2015; Mackenzie, 2014) and it is also exploited for commercial purposes in Namibia (Barnes *et al.*, 2010). The commercial exploitation could also be a possible explanation to the patchy distribution of the species across the study sites in addition to what has been outlined earlier. Additional threats to existence of the species at Kabompo is the converting of the mature trees into dugout canoes, as shown in Figure 5.1. The canoe business has become lucrative for the local people who pay a small licence fee to the Forest Department to construct a dugout canoe in the state forest. The finished

dugout canoe is then exchanged with one or two oxen with communities living along the banks of Zambezi River in western part of Zambia. Due to lack of effective law enforcement and monitoring, the same permit could be used to convey more than one dugout canoes before the licence expires (Shadreck Bwalya, pers. comm.).

The species contribution to community parameters showed the species as second least abundant among the six selected species across the study sites (Figure 4.1). The structural analysis of the selected woody species ranked the species in the fifth position with an IVI of 6.8% as shown in Table 4.7. The species was ranked second to *B. africana* at Kabompo study site (IVI, 17.3%) while at the other three study sites it recorded an IVI of 0%. These results shows that the species play an important role at Kabompo while not so much as at the other study sites. However, it is possible that these results underestimate the true important roles the species play at the other study sites where its IVI were 0%.



Figure 5.1: A dugout canoe under construction at Kabompo state forest - an additional threat to the existence and survival of the species. Four of such canoe construction activities were observed within a distance of 5 km. Photo: Paul Mwansa, 2016.

5.6.4. *Pterocarpus angolensis*

The regeneration density for *P. angolensis* was only recorded at Caprivi and Kabompo study sites (Figure 4.8). The regeneration density was significantly different between the two sites with Kabompo site showing higher density than Caprivi site. The regeneration density recorded in this study support densities reported in Chakanga et al. (1998) and Muhoko & Kamwi (2014). In their

respective reports, they reported density of 15 and 182 individuals per hectare, respectively. Surprisingly, there was no regeneration recorded at Kanovlei for this study even when the study was conducted in the mature stand with presence of potential mother trees. Additionally, Kanovlei site has an open forest with low canopy cover which could enhance the regeneration of the species since it grows better in open forest with less than 20 % canopy cover (De Cauwer *et al.*, 2017). Generally, the species has poor regeneration under natural conditions resulting into a slow recruitment process for the population (Vermeulen, 1990; Schwartzet *et al.*, 2002; Caro *et al.*, 2005). The low regeneration density recorded in this study and in others could be attributed to a number of factors that affect the regeneration of the species. Some authors attribute the low regeneration of the species to the effect of large herbivores (Schwartz *et al.*, 2002), fire damage to seed and seedlings (Vermeulen, 1990; Banda *et al.*, 2006b) and low competitive ability for resources (Caro *et al.*, 2005; De Cauwer *et al.*, 2017). In addition, good regeneration is hindered by failure to disperse seeds very far, seeds becoming mouldy when physically damaged, low rainfall and long years of the species to attain maturity after suffrutex stage (Vermeulen, 1990).

The species exhibited poor recruitment at the Kanovlei study site but good recruitment at the Caprivi and Kabompo study sites (Table 4.8). Greater than one recruitment ratio of seedling to sapling were recorded at Caprivi and Kabompo, an indicator of good recruitment (Mwavu & Witkowski, 2009). Kanovlei and Caprivi had positive height SCD OLS regression slopes (Table 4.9), indicating a disturbed and unstable recruitment (Obiri *et al.*, 2002). At Kabompo, the slope was significantly negative indicating good recruitment and survival of individuals in the next size classes (Lykke, 1998; Mwavu & Witkowski, 2009). When the species shows poor recruitment such as at Kanovlei in this study, it could be that the regenerations are undergoing dieback commonly known as suffrutex stage (Shackleton, 2002; Van Holsbeeck *et al.*, 2016), which may lead to the species taking many years to recruit new individuals in the population (Banda *et al.*, 2006b). The height SCD curves (Figure 4.17) showed evidence of episodic regeneration that may be caused by changes in summer rainfall and temperature seasonality (De Cauwer *et al.*, 2017).

The results of the regeneration limitation for the species shows that the species had a high limitation across the study sites. It recorded maximum limitation at Kanovlei site while at Caprivi and Kabompo sites the species was absent in more than 50% of the subplots. In a related study, similar low recruitment success of the species were found in the Kavango region (Kabajani, 2016). These results concur the conclusion of Caro *et al.* (2005) stating that the population of the species in protected forests such as state forests may be declining despite the protection from harvest.

P. angolensis has been widely researched (sensu Geldenhuys, 1977; Vermeulen, 1990; Schwartz *et al.*, 2002; Shackleton, 2002; Caro *et al.*, 2005; Banda *et al.*, 2006b; Burke, 2006; Therrell *et al.*, 2007; De Cauwer *et al.*, 2014; Van Holsbeeck *et al.*, 2016) and yet there is so much more to understand about the species. The correlation results in Table 4.13 showed that the dbh-height relationship was significantly positive at all the three study sites. The strength of the relationship at Kanovlei was weak, whereas at Caprivi and Kabompo were strong. The weak relationship strength could be attributed to the open nature of the forest where the species is not hurried into breaking into the canopy level since there is not much competition for light resources (Seifert *et al.*, 2014). Another possible explanation for the weak relationship strength could be due to localised site conditions affecting the species (Burke, 2006). As shown in Figure 4.23, the Kanovlei site showed a bell shaped diameter SCD while at Caprivi and Kabompo the diameter SCDs were reverse J-shaped although not too perfect. The significant negative OLS regression slopes shown in Table 4.9 are indicators of good rejuvenation, stability in the next size classes and expanding population of the species (Sop *et al.*, 2011; Traoré *et al.*, 2012). The mean diameter of the species recorded at the three sites shows that most of the individuals have many more years to reach the minimum harvestable dbh of 35 cm. It has been documented that the species take 85 - 100 years to reach 35 - 40 cm dbh (Therrell *et al.*, 2007) or 95 - 100 years for 45 cm dbh (Van Holsbeeck *et al.*, 2016) after the suffrutex stage. The species is also known to show distinct and variations in growth rate at an individual and stand level (Therrell *et al.*, 2007) that could be influenced by rainfall, minimum temperature in winter and temperature seasonality (De Cauwer *et al.*, 2017). Therefore, the mean dbh (Table 4.14) of the species in this study is an indicator of the species having been heavily harvested in the past (Barnes *et al.*, 2010) and illegal harvesting which is still evident to date (Kabajani, 2013). The mean height of 6.5 ± 0.6 m at Kanovlei is different from 9.5 m reported in Muhoko and Kamwi (2014), while the mean height of 6.9 ± 0.9 m at the Caprivi site differed to the 12.6 m reported in Chakanga *et al.* (1998). The difference at Kanovlei is hard to account for but probably the sample size and intensity may have a role. As of the Caprivi difference, this could be attributed to the illegal harvest of the species that has taken place between 1998 and 2016. Figure 5.2 shows the example of illegal harvest of the species encountered at the Caprivi study site. The basal area, which is total cross sectional area occupied the stems of the species, was significantly different among the study sites. The highest basal area was at Kanovlei with 2.7 ± 0.4 m² ha⁻¹ and the lowest was at Kabompo with 0.5 ± 0.1 m² ha⁻¹. The possible explanation of lower basal area could be the illegal harvest of the species at Caprivi and Kabompo that are located closer to main highways. De Cauwer *et al.* (2017) found that the species performed better in forests located more than 10 km from the main roads in the study across Southern Africa. In their study, they recorded basal area ranging from 0.2 m² ha⁻¹ to 22.4 m² ha⁻¹ with a mean of 6.9 ± 3.9 m² ha⁻¹. The

results of this study on basal area can only point to the heavy harvest of the species in the past and present, since the species is highly sought for wood carvings and furniture (Graz, 2004a; Shackleton, 2002).

The species contribution to community parameters among the selected species was high. The species was recorded as the third-most abundant across the study sites. The IVI ranked (Table 4.7) the species as ecologically the second most important species among the selected tree species across the study sites. The ecological importance ranking at Kanovlei place the species first and third at Caprivi and Kabompo study sites. The illegal harvesting of the species could be compromising the species important contribution to stand structure at Caprivi and Kabompo. Thus, the results for this study may have presented a somewhat limited role for the species at the study sites, which could be an artefact of the illegal harvesting trade.



Figure 5.2: An Example of illegal harvest of *Pterocarpus angolensis* that has contributed to rapid decline of the population at Caprivi State Forest. Photo: Miya Kabajani, 2016.

5.6.5. *Schinziophyton rautanenii*

The regeneration density of *S. rautanenii* was among the lowest among the selected species across the study sites. The regeneration of the species occurred only in RB across the study site (Figure 4.9). This rather intriguing result contradicts the findings of Geldenhuys (1977) who observed that the regeneration of species increased with total protection from fire. However, the results of the study concur with Kabajani (2016) who found the seedling density to increase with fire effect. Chakanga *et al.* (1998) also reported low regeneration density of 12 individuals per hectare. Probably the low regeneration density can be understood in light of the conclusions from the few studies on the species.

From the study of Graz (2002), it is uncertain if regeneration is a continuous process or if takes place in cohorts (episodic) but the species rely mostly on animal or human dispersal mechanism to spread seeds away from parent trees. Additional of uncertainty arises from Chidumayo (2016) who stated that the regeneration of the species occurs in small mono-specific groups that could partly be affected by the soil clay content. Other authors also found the species to prefer soils that are slightly better than the average poor Kalahari sand soils (De Cauwer *et al.*, 2016). The results on the species should be interpreted with caution, since there is little information on the ecology of the species (Graz, 2002).

The species showed recruitment at Kanovlei and Kabompo indicated by recruitment ratios (Table 4.8) greater than one and negative OLS regression slopes (Table 4.9). The OLS regression slopes at Sesheke and Caprivi were positive and their recruitment ratios could not be determined because of the absence of saplings in the SCD. Hence, these results show that the species is recruiting and the individuals are surviving to the next size classes at Kanovlei and Kabompo sites, whereas at Sesheke and Caprivi it might not be the case. The positive slopes are a clear indication of disturbed and unstable recruitment (Obiri *et al.*, 2002) with individuals failing to survive in the next size classes.

S. raoutanenii was the only species among the selected species recorded at all the four study sites. In spite of the species presence at all the study sites, it displayed high regeneration limitation. It registered maximum limitation in RU at all the study sites; 97.4% at Kanovlei, 94.4 % at Sesheke, 95% at Caprivi and 83.3% at Kabompo. The regeneration limitation across the study sites was 92.3%. These results could point to the regeneration being an episodic event in small mono-specific groupings as suggested by other authors.

The species registered positive correlation of dbh and height at Caprivi and Kabompo sites (Table 4.13). The strength of the relationship at Caprivi was weak and non-significant failing to account for the variations towards the height. Since the species is very susceptible to fire damage (Chidumayo, 2016; Graz, 2003), the weak and non-significant correlation could be attributed to behaviour of the species after a fire. Graz (2002, citing Chimbela, 1983) observed trees on disturbed land showing characteristics of a short bole with many branches. It can also be possible that the species grows very quickly during very first years because of the existing root systems (Vera De Cauwer, pers. comm.). However, at Kabompo the species showed a strong R^2 indicating that the height of species was positively responding to the increase in diameter. The results at Kabompo resonate with the findings of Sumida *et al.* (2013) that vigorously growing trees have almost a linear increase in dbh and height, which could be the case for the species as it competes for light and other resources. The diameter SCD in Appendix 4.7 showed very few individuals in the sapling and juvenile size classes at Kanovlei and Kabompo sites while at Caprivi the species had individuals in the juvenile and some of the larger

size classes. Individuals were complete absent in all diameter size classes at Sesheke site. Population structure and dynamics for species are hard to interpret from this study. This is due to very few individuals encountered at Kanovlei and Kabompo such that there were no OLS regression slopes produced to use for inference. The OLS regression slope at Caprivi was positive and non-significant that clearly means that the species has unstable population and lacking good rejuvenation (Sop *et al.*, 2011). The mean dbh and mean height results (Table 4.14) for the species in this study might give a misleading conclusion because of few individuals encountered in the study especially at Kanovlei and Kabompo. However, the Caprivi results are reliable and almost similar to those reported by Chakanga *et al.* (1998). The authors reported a mean height of 8.6 m while in this study it was 7.4 m. The basal area was very small at the three study sites and did not change significantly with fire treatment (Figure 4.32). The small basal area could be attributed to the absence of individuals in larger size classes.

The species contribution to community parameters showed low percentages across the study sites and at each of the study sites. The species was ranked last among the selected species as the sixth abundant species across study sites. It ranked fourth in abundance at Kanovlei, Caprivi and Kabompo while it ranked second at Sesheke since it was the only other selected species present at the site besides *B. plurijuga*. The structural role of *S. rautanenii* in the stand placed the species at the list-end of the selected species across the study sites with 4.5% IVI (Table 4.7). The species was ranked fourth at Kanovlei, Caprivi and Kabompo while it was ranked second at Sesheke. The patchy distribution of the species could account for the species' poor contribution to community parameters.

5.6.6. Terminalia sericea

Regeneration of *T. sericea* was only present at Kanovlei and Caprivi sites and absent at Sesheke and Kabompo sites (Figure 4.10). The regeneration density were non-significant on the interaction effect but showed significant difference with the two sites. The density was significantly different with RU recording higher density than RB at Caprivi. This result was unexpected since the species is considered fire tolerant (Rutherford, 1981) and an early succession species (De Cauwer *et al.*, 2016) colonising the open space created after a fire (Boaler & Sciwale, 1966). The results for this study are similar to those in the inventory report at Caprivi but quite different from those reported for the Kanovlei state forest (Chakanga *et al.*, 1998; Muhoko & Kamwi, 2014). The results in the inventories reported 2 517 individuals per hectare for Kanovlei state forest and 347 individuals per hectare for Caprivi state forest. *T. sericea* showed higher regeneration density in this study among the selected species, as presented in the results under Section 4.3.2. The reason for this difference with other selected species may have to do with nature of the species itself. Generally, *T. sericea* is fire tolerant

although individuals less than 1.5 m are affected by fire (Rutherford, 1981). However, the species may endure the fire through several diebacks but remain latent and important for regeneration. (Trapnell, 1959). It is also a pioneer species with tendency of growing fast after a disturbance (Boaler & Sciwale, 1966) with an ability to reach heights of more than two meters within a short period to escape smaller browsers and fire (Moyo *et al.*, 2015).

The species displayed an ability to recruit new individuals in the population. There were high recruitment ratios at Kanovlei and Caprivi sites (Table 4.8), a show of good recruitment for the species (Mwavu & Witkowski, 2009; West *et al.*, 2000). The negative OLS regression slopes in Table 4.9 are also an indicator of recruitment and ability of the new individual to survive to the next life stage (Obiri *et al.*, 2002). However, the height SCD in Figure 4.19 reveals a J-shaped structure in the first three size classes, which then drops drastically. The J-shape structure could be as a result of fire effect (Rutherford, 1981) and browsing by small wildlife (Moyo *et al.*, 2015). The drastic drop in the number of individuals just after the two metre height class may not be explained by this study, however the species is known to have numbers fall-off rapidly as the stand increase (Boaler & Sciwale, 1966). De Cauwer *et al.* (2016), in their study found the species not to get very big and experience high mortality.

The species showed the least regeneration limitation among the selected species across the study sites (Appendix 4.2). This shows the species ability to grow in different ecological environments, and the opportunistic nature to take up space created by other species after a disturbance (Boaler & Sciwale, 1966). The regeneration limitation was highest at Sesheke with the maximum limitation while it was lowest at Caprivi. The species favourite habitat is open areas (Mendelsohn & el Obeid, 2005; Orwa *et al.*, 2009). This could be a reason for it to have the least regeneration limitation at Caprivi and Kanovlei where the forests have a lower canopy while where the forests are more closed the regeneration limitation was high.

T. sericea was one of the selected species with least information on population structure from the results of this study. Although there was positive correlation of diameter and height, the slope and relationship strength was very poor. As shown, in Table 4.13 the species had only individuals with $dbh > 5$ cm at Caprivi study site. The correlation was non-significant and accounting of the variations in the model were negligible. This could mean that the growth of diameter has not much bearing on the growth of the height for the species at Caprivi. The diameter SCD (Figure 4.24) showed significant difference among the sapling, juvenile and adult classes. The significant negative OLS regression slopes (Table 4.9) is a clear indication of the species recruiting new individuals in the sampling stage that go to survive to juvenile stage. This trend is a show of healthy and stability in the first few size

classes (Sop *et al.*, 2011). The population also reveal gaps in the structure and absence of individuals in the larger size classes as pointed out by Boaler & Sciwale (1966). The mean dbh and mean height (Table 4.14) point to the fact that the species has a young population at Caprivi. The mean height in this study is within the same range as those found in the inventories of Kanovlei (5.0 m) and Caprivi (4.8 m) state forests. The basal area was very small (Figure 4.32), a further testimony of a young population for the species at the site since the species is utilitarian and widely used (Coates-Palgrave, 2002; Orwa *et al.*, 2009; van Wyk & van Wyk, 1997). It is also possible that the absence of individuals in the larger size classes and the small basal area come from illegal harvest of the species for construction and fencing poles.

The structural analysis of the selected woody species showed the species being ranked differently in abundance, relative frequency, relative density, relative dominance and important value index (IVI) at the study sites in Table 4.7. The species was ranked fourth most abundant ahead of *G. coleosperma* and *S. rautanenii* across the study sites. It was the most abundant at Caprivi, third at Kanovlei and fifth at Kabompo. The species was absent at Sesheke and recorded nil abundance. The structural role ranking put the species on fourth among the selected woody species across sites. It was second at Caprivi, third at Kanovlei and fifth at Kabompo. Despite the small basal area and show of a young population, the species is valued at most of the study sites with good IVI ranking among the selected woody species.

5.7. Chapter summary

This chapter has discussed the results in chapter four with focus on the 78 shrub and woody species encountered that predominantly belonged to the Caesalpinioideae subfamily. The high species richness, diversity and the similarity indices formed the basis of discussion on floristic composition across the study sites. The discussion also looked into the trends of natural regeneration of the shrub and tree species, stand structure and vegetation cover across the study sites. The second part of the discussion zoomed onto the individual selected tree species on their natural regeneration, recruitment patterns, population structure and the contribution to community parameters across the study sites and at each of the study sites.

Chapter 6

CONCLUSIONS AND RECOMMENDATIONS

6.1. Introduction

This chapter presents the conclusions derived from the findings of this study and the recommendations for future research. The focus is on the degree to which the aim and the specific objectives of the study have been addressed.

6.2. Conclusions and implications for management

The study has shown that the species richness was high across the study sites and at each of the study sites individually as indicated by the Shannon diversity indices. It has been shown in this study that fire treatment did not have significant effect on species richness but the richness increased with MAP from a dry site at Kanovlei state forest to a wetter site at Kabompo state forest along the rainfall gradient. Species richness in this study was supported by the high species diversity, evenness and equitability indices, which showed non-significant variations after a fire disturbance. The conclusion derived is that fire history did not cause low species richness and species are able to re-establish themselves whenever the fire disturbance occurs. The study has highlighted the variation in selected tree species contributions to community parameters by showing differences in composition and structure across study sites and among individual plots within each of the study sites. Thus, it has been concluded in this study that fire treatment did not significantly affect or alter significantly the floristic composition of the woody vegetation along the rainfall gradient. Therefore, short-term effect of fire (2 – 3 years) will maintain high species richness and diversity and above all will support the re-establishment of the tree species after a fire.

The study has shown that fire has significant effects on density and composition of natural regeneration and shrub layer. Shrub and pioneer tree species are quick to take up patches created by fire, as observed in other studies (Boaler & Sciwale, 1966; Gotosa *et al.*, 2013). In addition, regeneration in height size class of 0 – 0.5 m are highly stimulated by fire (especially on wetter sites), hence influencing the density and composition. The study has shown that selected tree species are regenerating as well although they have shown high regeneration limitation across the study sites. Some selected tree species such as *G. coleosperma* and *S. rautanenii* showed high sensitivity to fire, while species like *T. sericea* showed high density but *P. angolensis* had poor regeneration as expected. In addition, the study has shown that *B. plurijuga*, *G. coleosperma* and *S. rautanenii* have episodic regeneration as seen from their height classes. These tree species showed good regeneration on plots that burned between 1 and 3 years within the 14-year period. The selected tree species have

the ability and the potential to regenerate naturally. It has been shown from this study that *B. plurijuga*, *B. africana*, *G. coleosperma*, *P. angolensis*, *S. rautanenii* and *T. sericea* are all recruiting but not at all sites. The evidence of recruiting is based on the presence of the seedlings, saplings, > 1 seedling to sapling ratios and negative OLS regressions of height SCD. However, it is also evident that the selected tree species are not surviving in large numbers in the next size classes to stabilise the populations. These are important aspects that could point to the future trends of a species. The evidence from this study suggest that fire treatment has an effect on density and composition of natural regeneration. With the above conclusions in mind, tree species such as *B. plurijuga*, *G. coleosperma* and *S. rautanenii* may require interventions to boost their regeneration and survival to the next stage of life. Even though some tree species showed more density in recently burnt treatment, they also showed better regeneration with low to medium fire frequency. Hence, complete fire protection at 2 - 3 year interval, and early burning (Geldenhuys, 1977; Trapnell, 1959), may boost their chances of robust recruitment of new individuals in the population but early and late burning treatment effects were not tested in this study.

The findings from this study make several contributions to the understanding of the stand structure of the selected species along a rainfall gradient. It has been shown that diameter increase had a positive association with the height increase of the species even though in some cases the strength of the relationship was weak. The study has shown that a unit increase in diameter for any of the selected tree species had a corresponding increase in height based on the slope in the equation. For instance, *B. plurijuga* and *T. sericea* had a weak relationship strength at Sesheke (medium MAP) and Caprivi (medium MAP) sites; whereas *B. africana*, *P. angolensis* and *S. rautanenii* displayed strong relationships at Kabompo (high MAP) and weak relationships at Kanovlei site (low MAP). The general conclusion from this observation is that trees in open and dry forests do not invest much resource in height growth compared to those in closed forest at wetter sites where there is high competition for light and the need to break into the canopy level as quickly as possible. Water limitation cannot be ruled on influencing the relationship strength since Kabompo displayed strong relationships compared to the other sites. The diameter SCD showed the population of *B. plurijuga*, *B. africana* and *S. rautanenii* as unstable, lacking rejuvenation and declining while the population of *P. angolensis* and *T. sericea* were stable and rejuvenating in the smaller size classes. Some selected tree species showed significant differences in some of the diameter classes due to fire treatment but most of the differences in diameter SCD were non-significant. The mean diameters of the selected species were non-significantly different among sites except for that of *B. plurijuga* in the bigger size classes at Sesheke. The results showed almost the same trend for basal area of selected species with the only significantly different and higher basal area recorded at Sesheke site. In conclusion,

therefore, fire treatment did not highly influence the stand structure of the selected species along a rainfall gradient.

The results from this study have shown the highly significant effect of fire treatment on the vegetation cover along the rainfall gradient. The canopy cover at study sites was significantly influenced by the interaction between site and fire treatment with less cover experienced at the dry site compared to the wetter sites and generally less cover in the recently burnt treatments (but only on the wetter sites). The former conclusion is made after ignoring the Caprivi site, which had too many anthropogenic influences. The substrate cover on the forest floor showed trends that were influenced by the fire treatment with an exception of herbs that were non-significantly affected. The grasses on the forest floor reduced with fire at Kanovlei, Sesheke and Kabompo but they increased significantly at Caprivi. The shrubs reduced at Kanovlei, Caprivi and Kabompo sites but increased at Sesheke site due to fire. Coverage by bare ground on the forest floor increased with fire at all the four sites. These findings highlight the potential usefulness of fire treatment on vegetation cover for a few years since it opens up the canopy and reduce substrates that are competitors to tree species. Thus, the effect of fire treatment may lead to high density of natural regeneration along a rainfall gradient. These conclusions strongly support the idea of carrying out periodic burning in managing natural forests. Ideally, a 2 -3 year period will suffice to reduce competition on the forest floor and opening up shrub and small tree canopies to give chance to new recruits. Natural regeneration could be boosted and by the time the subsequent fire returns the new recruits could have grown to a level where they are able to withstand the effect of fire especially of early burning, as noted by Trapnell (1959) and Geldenhuys (1977).

This study set out to gain a better understanding of the fire treatment impact on the density and composition of natural regeneration of woody species along the rainfall gradient in miombo woodlands. Having evaluated the impact of fire treatment on floristic composition, natural regeneration density, stand structure of selected species and the vegetation cover, two of the objectives can be answered affirmatively. It is concluded that fire treatment affects the density and composition of natural regeneration and vegetation cover in the short-term of two to three years. Natural regeneration and substrates on the forest floor can be seasonal with varying trends of being present in one season and absent in the next. It has been shown that floristic composition and stand structure are not highly affected by fire treatment in the short-term. Therefore, this study has echoed the findings from other studies on the interaction of fire and site (MAP) along a rainfall gradient such as those conducted by Sankaran *et al.*, (2005) and Staver *et al.* (2011).

6.3. Recommendations for future research

The findings from this study contributes to the understanding of fire and site interaction along a rainfall gradient in the miombo woodlands and provide a basis for further research.

- i. This study focused on evaluation of natural regeneration of selected tree species in a single sample for one or two seasons. This may not have brought out the full understanding of natural regeneration trends of the tree species. This is because natural regeneration is highly variable, fluctuating from season to season (Kabajani, 2016) depending on the prevailing factors for that particular season. Therefore, to fully understand the regeneration trends, there is need to set up permanent sample plots in the respective state forests for long-term regeneration studies where data can be collected over time to help understand the trends for tree species that are commercially valuable and those that are valuable to local communities.
- ii. In this study, all regeneration was assumed to be from seedlings but could not have been the case for all regenerations since mode of regeneration was not assessed. Assessing mode of regeneration could be an important component of long-term regeneration studies. This could help in pointing out which of the tree species highly regenerate from sexual regeneration or asexual regeneration. Similarly, such a study could also contribute enormously to understanding seed banks and the behaviour of seeds given the local conditions of the site (*sensu* Wilson & Witkowski, 2003). Therefore, it is highly recommended that future studies in the state forests take into account mode of regeneration for tree species in the long-term regeneration studies.
- iii. There were many factors relating to individual tree species and site that were not investigated in this study but could have played a significant role in the findings of the study. Among others were soil nutrients, climatic conditions, water surplus and deficit that are required by an individual plant or that may be available at each of the sites (*sensu* Thornthwaite, 1948; Trapnell *et al.*, 1976; Fisher & Binkley, 2000; Burke, 2006). Therefore, it is highly recommended to investigate other site factors that may not seem too obvious but could be key in understanding, especially the unexpected, findings of every study.
- iv. The fire assessment used in this study was based on the MODIS burned area product with a spatial resolution of 500 X 500 m pixel size (Stellmes *et al.*, 2013). The period of 2001 – 2015 was good and long enough for credible results on fire study. However, the ideal situation is to assess natural regeneration, tree parameters and stand parameters repeatedly just after each fire. In doing this, evidence of fire effect on tree species will be better shown in the results. In addition, fire regime, which is fire frequency, intensity and season (Bond & Van Wilgen,

1996; Smit *et al.*, 2010), would be better understood for each of the state forest. At present, most fire studies apply the blanket cover principles based on the pioneers of fire studies in the miombo woodland (Geldenhuys, 1977; Trapnell, 1959). Since each site has localised conditions, fire studies at each of the state forest could help in understanding fire trend and the responses of the tree species. Hence, the recommendation to conduct fire studies at each of the state forest. It is also recommended that the raw data from fire studies that has been conducted by the Directorate of Forest for some time now at Kanovlei state forest be published. It is from such tailor made studies that fire management plans based on the observed trends can be developed from for sustainable natural forest management.

REFERENCES

- Angombe, S., Selanniemi, T. & Chakanga, M. 2001. *Wood resources report of M'kata pilot forest area*. Windhoek.
- Aune-Lundberg, L. & Strand, G.-H. 2014. Comparison of variance estimation methods for use with two-dimensional systematic sampling of land use/land cover data. *Environmental Modelling & Software*. 61:87–97.
- Backéus, I., Pettersson, B., Strömquist, L. & Ruffo, C. 2006. Tree communities and structural dynamics in miombo (*Brachystegia-Julbernardia*) woodland, Tanzania. *Forest Ecology and Management*. 230(1–3):171–178.
- Banda, T., Schwartz, M.W. & Caro, T. 2006a. Woody vegetation structure and composition along a protection gradient in a miombo ecosystem of western Tanzania. *Forest Ecology and Management*. 230(1–3):179–185.
- Banda, T., Schwartz, M.W. & Caro, T. 2006b. Effects of fire on germination of *Pterocarpus angolensis*. *Forest Ecology and Management*. 233(1):116–120.
- Barbour, M.G., Burk, J.H., Pitts, W.D., Gilliam, F.S. & Swartz, M.W. 1987. *Terrestrial plant ecology*. Third edit ed. New York: Addison Wesley Longman, Inc.
- Barnes, J.I., MacGregor, J.J., Nhuleipo, O. & Muteyauli, P.I. 2010. The value of Namibia's forest resources: Preliminary economic asset and flow accounts. *Development Southern Africa*. 27(2):159–176.
- Baskerville, G.L. 1972. Use of Logarithmic Regression in the Estimation of Plant Biomass. *Canadian Journal of Forestry*. 2:49–53.
- Blackie, R., Baldauf, C., Gautier, D., Gumbo, D., Kassa, H., Parthasarathy, N., Paumgarten, F., Sola, P., et al. 2014. *Tropical dry forests: The state of global knowledge and recommendations for future research*. Bogor.
- Boaler, S.B. & Sciwale, K.C. 1966. Ecology of a Miombo Site, Lupa North Forest Reserve, Tanzania: III. Effects on the Vegetation of Local Cultivation Practices. *Journal of Ecology*. 54(3):577–587.
- Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H. & White, J.-S.S. 2009. Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology and Evolution*. 24(3):127–135.

- Bond, W. 2001. Fires , ecological effects of. *Encyclopedia of Biodiversity Vol. 2*. 2:745–753.
- Bond, W.J. & Keeley, J.E. 2005. Fire as a global “herbivore”: The ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution*. 20(7):387–394.
- Bond, W.J. & Midgley, J.J. 2001. Ecology of sperouting in woody plants: the persistence niche. *Trends in Ecology and Evolution*. 16(1):45–51.
- Bond, W.J. & Van Wilgen, B.W. 1996. *Fire and plants*. First Edit ed. London: Chapman & Hall.
- Borghetti, M. & Giannini, R. 2005. Natural regeneration in woodland management. *Encyclopedia of Life Support Systems (EOLSS)*. I:5. [Online], Available: <http://www.eolss.net/Sample-Chapters/C12/E1-67-02-04.pdf>.
- Bourne, C.M., Regular, P.M., Sun, B., Thompson, S.P., Trant, A.J. & Wheeler, J.A. n.d. Generalized linear model analysis in ecology. 1–33.
- Brown, J.A., Salehi M., M., Moradi, M., Panahbehagh, B. & Smith, D.R. 2013. Adaptive survey designs for sampling rare and clustered populations. *Mathematics and Computers in Simulation*. 93:108–116.
- Burke, A. 2002. *Fire impacts on the vegetation in the Kavango Region for Environmental Profile of the Kavango Region project*. Windhoek.
- Burke, A. 2006. Savanna trees in Namibia - Factors controlling their distribution at the arid end of the spectrum. *Flora: Morphology, Distribution, Functional Ecology of Plants*. 201(3):189–201.
- Burke, A., Juola, V., Korhonen, K., Selanniemi, T. & Chakanga, M. 2001. *Field instructions for the NFI Namibia*. Windhoek: Directorate of Forestry.
- Busby, P.E., Vitousek, P. & Dirzo, R. 2010. Prevalence of tree regeneration by sprouting and seeding along a rainfall gradient in Hawai’i. *Biotropica*. 42(1):80–86.
- Campbell, B. Ed. 1996. *The Miombo in Transition: Woodlands and Welfare in Africa*. [Online], Available: <http://books.google.com/books?hl=nl&lr=&id=rpildJJVdU4C&pgis=1>.
- Campbell, B.M., Angelsen, A., Cunningham, A.B., Katerere, Y., Siteo, A.A. & Wunder, S. 2007. Miombo woodlands – opportunities and barriers to sustainable forest management. *Observatory*. 41. [Online], Available: http://www.cifor.cgiar.org/miombo/docs/Campbell_BarriersandOpportunities.pdf.
- Caro, T.M., Sungula, M., Schwartz, M.W. & Bella, E.M. 2005. Recruitment of *Pterocarpus angolensis* in the wild. *Forest Ecology and Management*. 219(2–3):169–175.

- De Cauwer, V. 2012. *Field measurement method to determine standing timber stock of woodlands in the Okavango catchment within “ The future Okavango Project” (SP05, Task 6)*. Windhoek.
- De Cauwer, V., Muys, B., Revermann, R. & Trabucco, A. 2014. Potential, realised, future distribution and environmental suitability for *Pterocarpus angolensis* DC in southern Africa. *Forest Ecology and Management*. 315:211–226.
- De Cauwer, V., Geldenhuys, C.J., Aerts, R., Kabajani, M. & Muys, B. 2016. Patterns of forest composition and their long term environmental drivers in the tropical dry forest transition zone of southern Africa. *Forest Ecosystems*. 3(1):3–23.
- De Cauwer, V., Fichtler, E., Beeckman, H., Graz, F.P., Mertens, J., Van Holsbeeck, S. & Muys, B. 2017. Predicting site productivity of the timber tree *Pterocarpus angolensis*. *Southern Forests: a Journal of Forest Science*. 1–10.
- Ceccon, E., Huante, P. & Rincón, E. 2006. Abiotic factors influencing tropical dry forests regeneration. *Brazilian archives of biology and technology*. 49(2):305–312.
- Central Statistical Office. 2011. *Zambia 2010 Census of population and Housing: Preliminary Population Figures*. Lusaka.
- Central Statistical Office. 2013. *2010 census of population and housing: Population and demographic projections 2011 - 2035*. Lusaka.
- Chakanga, M., Selanniemi, T. & Korhonen, K. 1998. *Forest Inventory Report of Caprivi State Forest*. Windhoek.
- Chen, W., Moriya, K., Sakai, T., Koyama, L. & Cao, C. 2014. Post-fire forest regeneration under different restoration treatments in the Greater Hinggan Mountain area of China. *Ecological Engineering*. 70:304–311.
- Chidumayo, E.N. & Gumbo, D.J. Eds. 2010. *The Dry Forests and Woodlands of Africa: Managing for products and services*. Earthscan.
- Chidumayo, E.N. 1987a. Species Structure in Zambian Miombo Woodland. *Journal of Tropical Ecology*. 3(2):109–118.
- Chidumayo, E.N. 1987b. Woodland structure, destruction and conservation in the Copperbelt area of Zambia. *Biological Conservation*. 40(2):89–100.
- Chidumayo, E.N. 1988. A Re-Assessment of Effects of Fire on Miombo Regeneration in the Zambian Copperbelt. *Journal of Tropical Ecology*. 4(4):361–372.

- Chidumayo, E.N. 1997. *Miombo ecology and management: an introduction*. London: IT Publications in association with the Stockholm Environment Institute.
- Chidumayo, E.N. 2016. Distribution and abundance of a keystone tree, *Schinziophyton rautanenii*, and factors affecting its structure in Zambia, southern Africa. *Biodiversity and Conservation*. 25(4):711–724.
- Childes, S.L. 1984. The population dynamics of some woody species in the Kalahari sand vegetation of Hwange National Park. University of the Witwatersrand.
- Clewer, A.G. & Scarisbrick, D.H. 2001. *Practical Statistics and Experimental Design for Plant and Crop Science*. Chichester: John Wiley & Sons, Ltd.
- Coates-Palgrave, M. 2002. *Keith Coates-Palgrave Trees of Southern Africa*. 3rd ed. Cape Town: Struik (Pty) Ltd.
- Condit, R. 2008. *Methods for estimating aboveground biomass of forest and replacement vegetation in the tropics*. Center for Tropical Forest Science Research Manual, 73 pages.
- Condit, R., Sukumar, R., Hubbell, S.P. & Foster, R.B. 1998. Predicting Population Trends from Size Distributions: A Direct Test in a Tropical Tree Community. *The American Naturalist*. 152(4):495–509.
- Cooper, C.F. 1957. The Variable Plot Method for Estimating Shrub Density. *Journal of Range Management*. 10(3):111–115.
- Curtis, B. & Mannheimer, C. 2005. *Tree atlas of Namibia*. Windhoek, Namibia: National Botanical Research Institute.
- Deiller, A.-F., Walter, J.-M.N. & Tremolieres, M. 2003. Regeneration strategies in a temperate hardwood floodplain forest of the Upper Rhine: Sexual versus vegetative reproduction of woody species. *Forest Ecology and Management*. 180(1–3):215–225.
- Deweese, P.A., Campbell, B.M., Katerere, Y., Siteo, A., Cunningham, A.B., Angelsen, A. & Wunder, S. 2011. *Managing the Miombo Woodlands of Southern Africa*. Washington.
- DFSC. 2001. *Conservation Plan for Genetic Resources of Zambezi teak (Baikiaea plurijuga) in Zambia*. Humlebaek.
- Diserud, O.H. & Ødegaard, F. 2007. A multiple-site similarity measure. *Biological Letters*. 38(2):35–37.
- Edgell, S.E. & Noon, S.M. 1984. Effect of violation of normality on the t test of the correlation

coefficient. *Psychological Bulletin*. 95(3):576–583.

- Everard, D.A., Midgley, J.J. & van Wyk, G.F. 1995. Dynamics of some forests in Kwa Zulu-Natal, South Africa, based on ordinations and size-class distributions. *South African Journal of Botany*. 61(6):283–292.
- Fanshawe, D.B. 2010. Vegetation Descriptions of the Upper Zambezi Districts of Zambia. *Biodiversity Foundation for Africa*. (22).
- FAO. 2003. *Role of Planted Forests and Trees Outside Forests in Sustainable Forest Management in the Republic of Namibia*, by I. Thomas and M. Chakanga. (Working Paper FP/30E). Rome.
- FAO. 2009. *Country pasture / forage resource profiles: ZAMBIA*. Rome.
- FAO. 2010. *Global Forest Resources Assessment 2010 Country Report Namibia*. (FRA2010/142). Rome.
- FAO. 2012. *Forest Resources Assessment 2015: Terms and Definitions*. Rome. [Online], Available: <http://www.fao.org/docrep/017/ap862e/ap862e00.pdf>.
- Faraway, J.J. 2006. *Extending the linear model with R: generalized linear, mixed effects and nonparametric regression models*. Boca Raton: Chapman & Hall/CRC Taylor & Francis Group.
- Faraway, J.J. 2015. *Linear Models with R*. Second Edition. Boca Raton: Taylor & Francis Group, LLC. [Online], Available: <https://books.google.pt/books?id=7tHhAwAAQBAJ>.
- Fichtler, E., Trouet, V., Beeckman, H., Coppin, P. & Worbes, M. 2004. Climatic signals in tree rings of *Burkea africana* and *Pterocarpus angolensis* from semiarid forests in Namibia. *Trees*. 18(4):442–451.
- Fisher, R.F. & Binkley, D. 2000. *Ecology and Management of Forest soils*. Third Edition. New York: John Wiley & Sons, Inc.
- Friedel, M.H. & Chewings, V.H. 1988. Comparison of crown cover estimates for woody vegetation in arid rangelands. *Australian Journal of Ecology*. 13(4):463–468.
- Gambiza, J., Bond, W., Frost, P.G.H. & Higgins, S. 2000. Special Section: Land use options in dry tropical woodland ecosystem in Zimbabwe: A simulation model of miombo woodland dynamics under different management regimes. *Ecological Economics*. 33(3):353–368.
- Gandiwa, E. 2011. Effects of repeated burning on woody vegetation structure and composition in a semi-arid Southern Africa savanna. *International journal of environmental sciences*. 2(2):458–471. [Online], Available:

<http://www.ipublishing.co.in/jesvollno12010/volumetwo/EIJES3046.pdf>.

- Gautam, M.K., Manhas, R.K. & Tripathi, A.K. 2016. Patterns of diversity and regeneration in unmanaged moist deciduous forests in response to disturbance in Shiwalik Himalayas, India. *Journal of Asia-Pacific Biodiversity*. 9(2):144–151.
- Geldenhuys, C.J. 1977. The Effect of Different Regimes of Annual Burning on Two Woodland Communities in Kavango. *South African Forestry Journal*. 103:32–42.
- Giess, W. 1998. A preliminary vegetation map of Namibia. *Dinteria*. 4:5–15.
- Glèlè Kakaï, R. & Sinsin, B. 2009. Structural description of two *Isobertia* dominated vegetation types in the Wari-Marô Forest Reserve (Benin). *South African Journal of Botany*. 75(1):43–51.
- Goldammer, J.G., Frost, P.G.H., Jurvélius, M., Evelien, M., Kruger, T., Moody, S.I. & Pogeyed, M. 2002. Community participation in integrated forest fire management : experiences from Africa , Asia and Europe. In P. Moore, D. Ganz, L.C. Tan, T. Enters, & P.B. Durst (eds.). Bangkok: Food and Agriculture Organization of the United Nations *Communities in flames: proceedings of an international conference on community involvement in fire management*. 33–55.
- Gonçalves, F.M.P., Revermann, R., Gomes, A.L., Aidar, M.P., Finckh, M. & Jürgens, N. 2017. Tree species diversity and composition of Miombo woodlands in south- central Angola, a chronosequence of forest recovery after shifting cultivation. *International Journal of Forestry Research*. 13.
- Gotosa, T.N., Gotosa, J., Katsvanga, C.A.T., Nyakudya, I.W., Kundhlande, A. & Mapaure, I. 2013. Woody vegetation diversity and regeneration in an abandoned sugar estate in semi-arid Zimbabwe. *Journal of Animal and Plant Sciences*. 23(5):1370–1378.
- Govender, N., Trollope, W.S.W. & Van Wilgen, B.W. 2006. The effect of fire season, fire frequency, rainfall and management on fire intensity in savanna vegetation in South Africa. *Journal of Applied Ecology*. 43(4):748–758.
- Graz, F.P. 2002. Description and ecology of *Schinziophyton rautanenii* (Schniz) Radel.-Sm. in Namibia. *Dinteria*. 27(27):19–35.
- Graz, F.P. 2003. Fire damage to *Schinziophyton rautanenii* (Schinz) trees in North-eastern Namibia. *Dinteria*. 28:39–43.
- Graz, F.P. 2004a. Description and ecology of *Pterocarpus angolensis* in Namibia. *Dinteria*. 29(29):27–39. [Online], Available: http://www.nbri.org.na/fileadmin/user_upload/publications/Dinteria/Dinteria_29D.pdf.

- Graz, F.P. 2004b. Structure and diversity of the dry woodland savanna of northern Namibia. Georg-August-Universität.
- Hammer, Ø. 2016. *PAST PAleontological STatistics Version 3.14*. Oslo.
- Hammer, Ø. & Harper, D.A.T. 2006. *Paleontological Data Analysis*. Boston: Blackwell Publishing.
- Handavu, F., Syampungani, S. & Chisanga, E. 2011. The influence of stump diameter and height on coppicing ability of selected key Miombo woodland tree species of Zambia: A guide for harvesting for charcoal production. *Journal of Ecology and the Natural Environment*. 3(14):461–468.
- Hoffmann, W.A. 1998. Post-burn reproduction of woody plants in a neotropical savanna: The relative importance of sexual and vegetative reproduction. *Journal of Applied Ecology*. 35(3):422–433.
- Holck, M.H. 2008. Participatory forest monitoring: An assessment of the accuracy of simple cost-effective methods. *Biodiversity and Conservation*. 17(8):2023–2036.
- Van Holsbeeck, S., De Cauwer, V., De Ridder, M., Fichtler, E., Beeckman, H. & Mertens, J. 2016. Annual diameter growth of *Pterocarpus angolensis* (Kiaat) and other woodland species in Namibia. *Forest Ecology and Management*. 373:1–8.
- IRDNC. 2015. *Timber Trade in Namibia*. Windhoek: Integrated development and nature conservation/Southern Africa science service centre for climate change and adaptive land management.
- Isango, J.A. 2007. *Stand structure and tree species composition of Tazania Miombo woodlands: A case study from Miombo woodlands of community based forest management in Iringa district*.
- JAICAF. 2008. *Agriculture and Forestry in Zambia: Present situation and issues for development*. Tokyo. [Online], Available: http://statbank.ssb.no/statistikkbanken/Default_FR.asp?Productid=10.04&PXSid=0&nvl=true&PLanguage=0&tilside=selecttable/MenuSelP.asp&SubjectCode=10.
- Jayaraman, K. 1999. *A Statistical Manual For Forestry Research*. Bangkok: Food and Agriculture Organisation.
- Jew, E.K.K., Dougill, A.J., Sallu, S.M., O’Connell, J. & Benton, T.G. 2016. Miombo woodland under threat: Consequences for tree diversity and carbon storage. *Forest Ecology and Management*. 361:144–153.
- Kabajani, M. 2013. *Population structure of Pterocarpus angolensis (Kiaat) in response to harvesting in the western Kavango Region, Namibia*. Windhoek. [Online], Available:

<http://edugreen.teri.res.in/explore/forestry/what.htm>.

- Kabajani, M.W. 2016. An assessment of the natural regeneration of valuable woody species in the kavango regions of North eastern Namibia. Namibia University of Science and Technology.
- Kalaba, F.K., Quinn, C.H., Dougill, A.J. & Vinya, R. 2013. Floristic composition, species diversity and carbon storage in charcoal and agriculture fallows and management implications in Miombo woodlands of Zambia. *Forest Ecology and Management*. 304:99–109.
- Kayofa, F. 2015. Natural Regeneration Potential of *Pterocarpus angolensis* (Kiaat Tree) in the dry forests of northern Namibia. Stellenbosch University.
- Kennard, D.K., Gould, K., Putz, F.E., Fredericksen, T.S. & Morales, F. 2002. Effect of disturbance intensity on regeneration mechanisms in a tropical dry forest. *Forest Ecology and Management*. 162(2–3):197–208.
- Kikula, I.S. 1986. The Influence of Fire on the Composition of Miombo Woodland of SW Tanzania. *Oikos*. 46(3):317–324.
- Kozlowski, T.T. & Ahlgren, C.. 1974. *Fires and ecosystems*. New York: Academic Press.
- Kruger, L.M., Midgley, J.J. & Cowling, R.M. 1997. Resprouters vs reseeders in South African forest trees; a model based on forest canopy height. *Functional Ecology*. 11(1):101–105.
- Lamprecht, H. 1989. *Silviculture in the tropics: tropical forest ecosystems and their tree species - possibilities and methods for their long-term utilisation*.
- Lavorel, S., McIntyre, S. & Grigulis, K. 1999. Plant response to disturbance in a Mediterranean grassland: How many functional groups? *Journal of Vegetation Science*. 10(5):661–672.
- Lohmann, D., Tietjen, B., Blaum, N., Joubert, D.F. & Jeltsch, F. 2014. Prescribed fire as a tool for managing shrub encroachment in semi-arid savanna rangelands. *Journal of Arid Environments*. 107:49–56.
- Luoga, E.J., Witkowski, E.T.F. & Balkwill, K. 2002. Harvested and standing wood stocks in protected and communal miombo woodlands of eastern Tanzania. *Forest Ecology and Management*. 164(1–3):15–30.
- Luoga, E.J., Witkowski, E.T.F. & Balkwill, K. 2004. Regeneration by coppicing (resprouting) of miombo (African savanna) trees in relation to land use. *Forest Ecology and Management*. 189(1–3):23–35.
- Lykke, A.M. 1998. Assessment of species composition change in savanna vegetation by means of

- woody plants' size class distributions and local information. *Biodiversity and Conservation*. 7(10):1261–1275.
- Mackenzie, C. 2014. *Forest Governance and Timber Trade Flows Within , to and from Eastern and Southern African Countries Zambia Study*. Lusaka.
- Madow, W.G. & Madow, L.H. 1944. On the theory of systematic sampling, I. *The Annals of Mathematical Statistics*. 1–24.
- Magurran, A.E. 2004. *Measuring Biological Diversity*. Boston: Blackwell Science Ltd.
- Maliondo, S.M.S., Abeli, W.S., Meiludie, R.E.L.O., Migunga, G.A. & Applegate, G.B. 2005. Tree Species Composition and Potential Timber Production of Communal Miombo Woodland in Handeni District Tanzania. *Journal of Tropical Forest Science*. 17(1):104–120.
- Mapaure, I. 2013. Short-term responses of shrub layer communities to dry season fires and tree thinning in semi-arid miombo woodlands of north-western Zimbabwe. *African Journal of Plant Science*. 7(9):414–425.
- Margalef, R. 1972. Homage to Evelyn Hutchinson, or why is there an upper limit to diversity. *Transactions of the Connecticut Academy of Arts and Sciences*. 44:211–235.
- McDonald, J.H. 2014. *Handbook of Biological Statistics*. Third Edit ed. Baltimore: SPARKY HOUSE PUBLISHING Baltimore,.
- Mendelsohn, J. & el Obeid, S. 2005. *Forests and Woodlands of Namibia*. Windhoek: RAISON (Research Information Services of Namibia).
- Mendelsohn, J., Jarvis, A., Roberts, C. & Robertson, T. 2002. *The Atlas of Namibia: A Potrait of the land and its people*. Windhoek: David Philip Publishers.
- Mensah, S., Houehanou, T.D., Sogbohossou, E.A., Assogbadjo, A.E. & Glèlè Kakaï, R. 2014. Effect of human disturbance and climatic variability on the population structure of *Azelia africana* Sm. ex pers. (Fabaceae-Caesalpinioideae) at country broad-scale (Benin, West Africa). *South African Journal of Botany*. 95:165–173.
- Ministry of Agriculture, Water and Forestry. 2011. *A Forest Research Strategy for Namibia (2011–2015)*. Windhoek: Directorate of Forestry.
- Minitab. 2016. *Getting Started with Minitab 17*. New York: Minitab Inc. [Online], Available: https://www.minitab.com/uploadedFiles/Documents/getting-started/Minitab17_GettingStarted-en.pdf.

- Mmolotsi, R.M., Obopile, M., Kwerepe, B.C., Sebolai, B., Rampart, M.P., Segwagwe, A.T., Ramolemana, G., Maphane, T.M., et al. 2012. Studies on Mukwa (*Pterocarpus angolensis*, D. C.) Dieback in Chobe Forest Reserves in Botswana. *Journal of Plant Studies*. 1(2):2010–2013.
- Mojeremane, W. & Lumbile, A.U. 2016. A Review of *Pterocarpus angolensis* DC (Mukwa) an Important and Threatened Timber Species of the Miombo Woodlands. *Research Journal of Forestry*. 21(1):314–317.
- Moses, M. 2013. Assessment of trade-offs between timber and carbon values of *Pterocarpus angolensis* (Kiaat) in the Kavango Region of Namibia - a comparison of current and potential values. Stellenbosch University.
- Moyo, H., Scholes, M.C. & Twine, W. 2015. Resprouting after coppicing is negatively influenced by browsing in *Terminalia sericea*. *Ecological Research*. 30(5):899–907.
- Mugasha, W.A., Bollandås, O.M. & Eid, T. 2013. Relationships between diameter and height of trees in natural tropical forest in Tanzania. *Southern Forests: a Journal of Forest Science*. 75(4):221–237.
- Muhoko, E.M. & Kamwi, J.M. 2014. *Woody Resource Inventory Report for Kanovlei State Forest*. Windhoek.
- Mukosha, J. & Siampale, A. 2009. *Integrated land use assessment (ILUA) Zambia 2005 - 2008*. Lusaka.
- Muller-landau, H.C., Wright, S.J., Calderón, O., Hubbell, S.P. & Foster, R.B. 2002. Recruitment Limitation : Concepts , Methods and Case-studies from a Tropical Forest. In D.J. Levey, W.R. Silva, & M. Galetti (eds.). Chicago: CAB International *Seed Dispersal and Frugivory: Ecology, Evolution and Conservation*. 51–69.
- Munishi, P.K.T., Temu, R.P.C. & Soka, G. 2011. Plant communities and tree species associations in a Miombo ecosystem in the Lake Rukwa basin , Southern Tanzania : Implications for conservation. *Journal of ecology and the natural environment*. 3(2):63–71.
- Munkert, H.C. 2009. Sexual and vegetative regeneration of three leguminous tree species in South African savannas. *South African Journal of Botany*. 75(3):606–610.
- Mwabumba, L., Chirwa, P.W., Lowore, J.D. & Munthali, C.R.Y. 1999.
- Mwakalukwa, E.E., Meilby, H. & Treue, T. 2014. Floristic Composition, Structure, and Species Associations of Dry Miombo Woodland in Tanzania. *ISRN Biodiversity*. 1–15.
- Mwavu, E.N. & Witkowski, E.T.F. 2009. Population structure and regeneration of multiple-use tree

- species in a semi-deciduous African tropical rainforest: Implications for primate conservation. *Forest Ecology and Management*. 258(5):840–849.
- Mwitwa, J.P. 2004. Growth and Physiological Parameters related to Shoot Dieback in *Pterocarpus angolensis* DC Seedlings. Stellenbosch University.
- National Statistics Agency. 2006. *Republic of Namibia Population Projections : 2001 - 2031*. Windhoek.
- National Statistics Agency. 2011. *Namibia 2011 Population and Housing Census Indicators*. Windhoek.
- Ncube, Z. & Mufandaedza, E. 2013. Effects of fire on coppice shoot production and growth in an African savanna woodland dominated by *Pterocarpus angolensis* and *Baikiaea plurijuga*. *Academia Journal of Agricultural Research*. 1(2):28–35.
- Nepolo, E. & Mapaure, I. 2012. Short-Term Influence of Fire on Herbaceous Composition, Diversity and Grass Biomass Production in Semi-Arid Savanna Woodland in Windhoek, Namibia. *International Journal of Ecosystem*. 2(6):154–160.
- Ngoma, J., Speer, J.H., Vinya, R., Kruijt, B., Moors, E. & Leemans, R. 2015. The dendrochronological potential of *Baikiaea plurijuga* in Zambia. *Dendrochronologia*. 1–13.
- O'Brien, E.M. 1993. Climatic Gradients in Woody Plant Species Richness : Towards an Explanation Based on an Analysis of Southern Africa's Woody Flora. *Journal of Biogeography*. 20(2):181–198.
- O'Hara, R.B. & Kotze, D.J. 2010. Do not log-transform count data. *Methods in Ecology and Evolution*. 1(2):118–122.
- Obiri, J., Lawes, M. & Mukolwe, M. 2002. The dynamics and sustainable use of high-value tree species of the coastal Pondoland forests of the Eastern Cape Province, South Africa. *Forest Ecology and Management*. 166(1–3):131–148.
- Ohl, C. 2005. Characterisation of post-fire vegetation regeneration , successional patterns and underlying plant regeneration mechanisms in south-eastern Spain. Martin-Luther-University.
- Okitsu, S. 2005. Factors controlling geographical distribution in savanna vegetation in Namibia. *African Study Monographs*. Suppl. 30:135–151.
- Orwa, C., Mutua, A., Kindt, R., Jamnadass, R. & Anthony, S. 2009. [Online], Available: <http://www.worldagroforestryorg/sites/treedbs/treedatabases.asp>.

- Oyedeeji, S., Onuche, F.J., Animasaun, D.A., Ogunkunle, C.O., Agboola, O.O. & Isichei, A.O. 2016. Short-term effects of early-season fire on herbaceous composition, dry matter production and soil fertility in Guinea savanna, Nigeria. *Archives of Biological Sciences*. 68(1):7–16.
- Pardos, M., Montero, G., Cañellas, I. & Ruiz del Castillo, J. 2005. Ecophysiology of natural regeneration of forest stands in Spain. *Forest Systems*. 14(3):434–445. [Online], Available: <http://revistas.inia.es/index.php/fs/article/view/939>.
- Pohlert, T. 2014. The Pairwise Multiple Comparison of Mean Ranks Package (PMCMR). *R package*. 27.
- Quantum GIS Development Team. 2016. [Online], Available: www.qgis.oseo.org.
- R Core Team. 2017. *An Introduction to R - Notes on R: A Programming Environment for Data Analysis and Graphics Version 3.4.0*. R Core Team.
- Reinhart, K.O., Dangi, S.R. & Vermeire, L.T. 2016. The effect of fire intensity, nutrients, soil microbes, and spatial distance on grassland productivity. *Plant and Soil*. 409(1–2):203–216.
- Revermann, R., Finckh, M., Stellmes, M., Strohbach, B.J., Frantz, D. & Oldeland, J. 2016. Linking land surface phenology and vegetation-plot databases to model terrestrial plant α -diversity of the okavango basin. *Remote Sensing*. 8(5).
- Ribeiro, N.S., Shugart, H.H. & Washington-Allen, R. 2008. The effects of fire and elephants on species composition and structure of the Niassa Reserve, northern Mozambique. *Forest Ecology and Management*. 255(5–6):1626–1636.
- Roberts-Pichette, P. & Gillespie, L. 1999. *Terrestrial Vegetation Biodiversity Monitoring Protocols*. Ontario.
- Rutherford, M.C. 1981. Survival, regeneration and leaf biomass changes in woody plants following spring burns in *Burkea africana*-*Ochna pulchra* Savanna. *Bothalia*. 13(3 & 4):531–552.
- Ryan, C.M. & Williams, M. 2011. How does fire intensity and frequency affect miombo woodland tree populations and biomass? *Ecological Applications*. 21(1):48–60.
- Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D.J., Cade, B.S., Gignoux, J., Higgins, S.I., et al. 2005. Determinants of woody cover in African savannas. *Nature*. 438(7069):846–849.
- Sankaran, M., Ratnam, J. & Hanan, N.P. 2008. Woody cover in African savannas: The role of resources, fire and herbivory. *Global Ecology and Biogeography*. 17(2):236–245.

- SASSCAL. 2016. *Forest regeneration , growth , threats and trends in different forest types (Task ID 038)*. Windhoek: Southern Africa science service centre for climate change and adaptive land management.
- Schelstraete, M. 2016. Assessment of fire damage on the forest population near Hamoye, Kavango, Namibia. Universiteit Gent.
- Scholes, A.R.J., Dowty, P.R., Caylor, K., Parsons, D.A.B. & Frost, P.G.H. 2002. Trends in Savanna Structure and Composition along an Aridity Gradient in the Kalahari. *Journal of Vegetation Science*. 13(3):419–428.
- Scholes, R.J., Frost, P.G.H. & Tian, Y. 2004. Canopy structure in savannas along a moisture gradient on Kalahari sands. *Global Change Biology*. 10(3):292–302.
- Schwartz, M.W., Caro, T.M. & Banda-Sakala, T. 2002. Assessing the sustainability of harvest of *Pterocarpis angolensis* in Rukwa Region, Tanzania. *Forest Ecology and Management*. 170:259–269.
- Seifert, T., Seifert, S., Seydack, A., Durrheim, G. & Gadow, K. von. 2014. Competition effects in an afrotemperate forest. *Forest Ecosystems*. 1(1):13.
- Shackleton, C.M. 2002. Growth patterns of *Pterocarpus angolensis* in savannas of the South African lowveld. *Forest Ecology and Management*. 166(1–3):85–97.
- Shackleton, C.M. & Scholes, R.J. 2011. Above ground woody community attributes, biomass and carbon stocks along a rainfall gradient in the savannas of the central lowveld, South Africa. *South African Journal of Botany*. 77(1):184–192.
- Shackleton, C.M., Griffin, N.J., Banks, D.I., Mavrandonis, J.M. & Shackleton, S.E. 1994. Community Structure and Species Composition along a Disturbance Gradient in a Communally Managed South African Savanna. *Vegetatio*. 115(2):157–167.
- Sheuyange, A., Oba, G. & Weladji, R.B. 2005. Effects of anthropogenic fire history on savanna vegetation in northeastern Namibia. *Journal of Environmental Management*. 75(3):189–198.
- Siljander, M. 2009. Predictive fire occurrence modelling to improve burned area estimation at a regional scale: A case study in East Caprivi, Namibia. *International Journal of Applied Earth Observation and Geoinformation*. 11(6):380–393.
- Singh, A.K., Kushwaha, M., Rai, A. & Singh, N. 2017. Changes in soil microbial response across year following a wildfire in tropical dry forest. *Forest Ecology and Management*. 391:458–468.
- Smit, I.P.J., Asner, G.P., Govender, N., Kennedy-Bowdoin, T., Knapp, D.E. & Jacobson, J. 2010.

- Effects of fire on woody vegetation structure in African savanna. *Ecological Applications*. 20(7):1865–1875.
- Sop, T.K., Oldeland, J., Schmiedel, U., Ouedraogo, I. & Thiombiano, A. 2011. Population structure of three woody species in four ethnic domains of the sub-sahel of Burkina Faso. *Land Degradation and Development*. 22(6):519–529.
- Stahle, D.W., Mushove, P.T., Cleaveland, M.K., Roig, F. & Haynes, G.A. 1999. Management implications of annual growth rings in *Pterocarpus angolensis* from Zimbabwe. *Forest Ecology and Management*. 124(2–3):217–229.
- Staver, A.C., Archibald, S. & Levin, S. 2011b. Tree cover in sub-Saharan Africa: Rainfall and fire constrain forest and savanna as alternative stable states. *Ecology*. 92(5):1063–1072.
- Staver, A.C., Archibald, S. & Levin, S. 2011a. *The Global Extent and Determinants of savanna and forest as Alternative Biome States*.
- Stellmes, M., Frantz, D., Finckh, M., Revermann, R., Roder, A. & Hill, J. 2013. Fire frequency, fire seasonality and fire intensity within the Okavango region derived from MODIS fire products. *Biodiversity and Ecology*. 5:1–13.
- Strohbach, B.J. & Petersen, A. 2007. Vegetation of the central Kavango woodlands in Namibia: An example from the Mile 46 Livestock Development Centre. *South African Journal of Botany*. 73(3):391–401.
- Stromgaard, P. 1986. Early Secondary Succession on Abandoned Shifting Cultivator Plots in the Miombo of South Central Africa. *Biotropica*. 18(2):97–106.
- Sumida, A., Miyaura, T. & Torii, H. 2013. Relationships of tree height and diameter at breast height revisited: Analyses of stem growth using 20-year data of an even-aged *Chamaecyparis obtusa* stand. *Tree Physiology*. 33(1):106–118.
- Sweet, J. & Burke, A. 2006. *Country pasture / forage resource profiles: Namibia*. Windhoek.
- Syampungani, S., Geldenhuys, C.J. & Chirwa, P.W. 2016. Regeneration dynamics of miombo woodland in response to different anthropogenic disturbances: forest characterisation for sustainable management. *Agroforestry Systems*. 90(4):563–576.
- Therrell, M.D., Stahle, D.W., Mukelabai, M.M. & Shugart, H.H. 2007. Age, and radial growth dynamics of *Pterocarpus angolensis* in southern Africa. *Forest Ecology and Management*. 244(1–3):24–31.
- Thornthwaite, C.W. 1948. An Approach toward a Rational Classification of Climate. *Geographical*

Review. 38(1):55–94.

- Thunström, L. 2012. Population size structure and recruitment rate in *Pterocarpus angolensis*, an exploited tree species in miombo woodlands, Tanzania. Uppsala University.
- Timberlake, J. & Chidumayo, E. 2011. *Miombo Ecoregion Vision report*. Bulawayo.
- Traoré, L., Sop, T.K., Dayamba, S.D., Traoré, S., Hahn, K. & Thiombiano, A. 2012. Do protected areas really work to conserve species? A case study of three vulnerable woody species in the Sudanian zone of Burkina Faso. *Environment, Development and Sustainability*. 14(4):663–686.
- Trapnell, C.G. 1959. Ecological Results of Woodland and Burning Experiments in Northern Rhodesia. *Journal of Ecology*. 47(1):129–168.
- Trapnell, C.G., Friend, M.T., Chamberlain, G.T. & Birch, H.F. 1976. The Effects of Fire and Termites on a Zambian Woodland Soil. *Journal of Ecology*. 64(2):577–588.
- Venables, W.N. & Dichmont, C.M. 2004. GLMs, GAMs and GLMMs: An overview of theory for applications in fisheries research. *Fisheries Research*. 70(2–3 SPEC. ISS.):319–337.
- Verma, S. & Jayakumar, S. 2015. Post-fire regeneration dynamics of tree species in a tropical dry deciduous forest, Western Ghats, India. *Forest Ecology and Management*. 341:75–82.
- Vermeulen, W.J. 1990. *A Monograph on Pterocarpus angolensis*. Pretoria: Department of Environmental Affairs Water Affairs and Forestry.
- Vieira, D.L.M. & Scariot, A. 2006. Principles of natural regeneration of tropical dry forests for restoration. *Society for Ecological Restoration International*. 14(1):11–20.
- Wale, H.A., Bekele, T. & Dalle, G. 2012. Floristic diversity, regeneration status, and vegetation structure of woodlands in Metema Area, Amhara National Regional State, Northwestern Ethiopia. *Journal of Forestry Research*. 23(3):391–398.
- Warton, D.I. & Hui, F.K.C. 2011. The arcsine is asinine: the analysis of proportions in ecology. *Ecology*. 91(1):3–10.
- West, A.G., Midgley, J.J. & Bond, W.J. 2000. Regeneration failure and the potential importance of human disturbance in a subtropical forest. *Applied Vegetation Science*. 3:223–232.
- White, F. 1983. *The vegetation of Africa: A descriptive memoir to accompany the UNESCO/AETFAT/UNSO vegetation map of Africa*. Paris.
- Whittaker, R.H. 1972. Evolution and Measurement of Species Diversity. *Taxon*. 21(2):213–251.
- Wiafe, E.D. 2014. Tree Regeneration after Logging in Rain -forest Ecosystem. *Research Journal of*

Biology. 2:18–28. [Online], Available: [phd+literature%5CWiafe2014.pdf](#).

Wilson, E.G. & Witkowski, E.T.F. 2003. Seed Banks , Bark Thickness and Change in Age and Size Structure (1978-1999) of the African Savanna Tree , *Burkea africana*. *Plant Ecology*. 167(1):151–162.

www.sasscal.org. 2016. *SASSCAL – Southern Africa Science Service Centre for Climate Change and Adaptive Land Management*. [Online], Available: www.sasscal.org.

van Wyk, B. & van Wyk, P. 1997. *Field Guide to Trees of Southern Africa*. Cape Town: Struik Publishers.

Zeileis, A., Kleiber, C. & Jackman, S. 2008. Regression Models for Count Data in R. *Journal Of Statistical Software*. 27(8):1076–84.

Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A. & Smith, G.M. 2009. *Mixed Effects Models and Extensions in Ecology with R*. Vol. 1. New York: Springer Science+Business Media, LLC.

APPENDICES

Appendix 1: Data record sheets

Appendix 1.1: Stand parameters record sheet

STAND INFORMATION		Land type	
Plot no		1 = Escarpment (steep, long ridge, extending over more than 10 km)	
Area		2 = Hill crest (top of a ridge)	
Date		3 = Mid slope	
Team		4 = Foot slope	
Lat		5 = Valley floor	
Long		6 = Plateau	
Accuracy (m)		7 = Depression/Pan	
Elevation (m)		8 = Inselberg (isolated mountain)	
Land type		9 = Plain	
Land use		10 = Dry river bed	
Crown coverage trees		11 = River bank (current)	
Crown coverage shrubs		12 = River/alluvial terrace (old bank)	
Grass coverage		13 = Dune base	
Herb coverage		14 = Dune slope and crest	
Bare soil coverage		15 = Interdune, (always sand, between dunes)	
Human influence		16 = Rocky outcrop	
Domestic mammals		17 = Water course	
Damage by forest fire		18 = Flood plain	
Recent fire (Y/N)		19 = Oshana	
Main cause damage plot		20 = Oshana	
Ptan nearby (Y/N)			
Remarks			

Land use	Cause damage
1 = Naturally vegetated land (including farmland used for grazing)	1 = Forest fire
2 = Agro-silvopastoral (Agro-Forestry)	2 = Mammals wild
3 = Forestry (commercial tree production, plantation)	3 = Mammals domestic
4 = Agriculture (crop production)	4 = Insect
5 = Mining	5 = Human
6 = Bare soil	6 = Frost
	7 = Drought
	8 = Erosion
	9 = Storm
	0 = Unknown

Damage
0 = No damage/influence
1 = Mild damages, not affecting the trees
2 = Moderate damages, decreasing vitality of trees
3 = Serious damages, seriously affecting several trees
4 = Fatal damages, several trees are dying or dead

Cause damage
10=fungi

Appendix 1.2: Tree parameters record sheet

TREES														
Plot no 0														
Radius 30m: cbh > 141 cm ; Radius 20m : cbh > 62.8cm ; Radius 10m : cbh > 15,7 cm														
no	Species	cbh (cm)	Alive	Dist (m)	Lat	Long	Timb q	I log (m)	Crown cl	Phenol.	Height(m)	Damage	Cause dam	Remarks
1														
2														
3														
4														
5														
6														
7														
8														
9														
10														
11														
12														
13														
14														
15														
16														
17														
18														
19														
20														
21														
22														
23														
24														
25														
26														
27														
28														
29														
30														

Alive
1 = Alive tree
2 = Standing dead tree
3 = Dead, lying
4 = Stump
5 = stump with coppice

Timber or pole quality
4 = Expected good
5 = Expected medium
6 = Expected poor
7 = no timber or pole quality

Crown class
1 = Open crown
2 = Dominant
3 = Co-dominant
4 = Intermediate
5 = Overtopped

Phenology
L(eaves), F(owers), F(ruits)
Damage : see sheet 1

Cause damage
1 = Forest fire
2 = Mammals wild
3 = Mammals domestic
4 = Insect
5 = Human
6 = Frost
7 = Drought
8 = Erosion
9 = Storm
0 = Unknown

Species names : list first three letters of genus and species, unless in cases where confusion can arise e.g. Combsp and Commsp.
 unknown species are listed as Specq, but it is advised to take a picture and a leave sample to identify it afterwards.
 Multiple stems : dbh and log length needs to be recorded for each stem.

Appendix 1.5: Substrate cover record sheet

Line intercept

Date

plot no

Line length

Transect location

Substrate	Start distance	End distance

Appendix 2: GPS coordinates for sample plots

Kanovlei plots				Sesheke plots			Caprivi plots			Kabompo plots		
Plot #	Latitude (S)	Longitude (E)	Elevation	Latitude (S)	Longitude (E)	Elevation	Latitude (S)	Longitude (E)	Elevation	Latitude (S)	Longitude (E)	Elevation
1	-19.2844867	19.4517729	1235.1	-17.3792517	24.5136429	1016.5	-17.68176	23.921683	991.4	-13.7273074	24.24943208	1146.9
2	-19.2844662	19.4507369	1135.3	-17.3793715	24.512403	1000.3	-17.6807993	23.9216226	993	-13.72648376	24.2501015	1152.6
3	-19.2846961	19.4498165	1234.4	-17.3793473	24.5112812	1003.7	-17.6799879	23.92201014	996.7	-13.7257984	24.2508005	1161
4	-19.2854831	19.4560043	1237.3	-17.3652217	24.5150729	966.7	-17.66804246	23.9143263	996.9	-13.7435062	24.258299	1157
5	-19.2856873	19.4550315	1231.7	-17.3646731	24.513614	1008.3	-17.6674028	23.9150296	1004.2	-13.7426987	24.259143	1163
6	-19.2858404	19.4540679	1231.1	-17.3654233	24.5129763	1019.5	-17.6665782	23.9153357	998.6	-13.7420084	24.2596985	1176.5
7	-19.2882029	19.4478033	1233.7	-17.3734769	24.5129963	1021.4	-17.6661729	23.9143076	997.4	-13.7605347	24.26760906	1168.7
8	-19.2881889	19.4487628	1234.3	-17.3739342	24.5108108	1021.5	-17.5874052	24.0601425	1002.7	-13.759631	24.26795071	1147
9	-19.2882525	19.4497332	1230.1	-17.3746891	24.5089451	1002.0	-17.5865154	24.0603117	1006.4	-13.7584793	24.268814	1172.3
10	-19.288115	19.4526983	1229.2	-17.3743217	24.5198537	1006.9	-17.5857894	24.060781	1008.1	-13.7320341	24.210749	1153.7
11	-19.2882223	19.4536407	1226.5	-17.3745464	24.5207489	1002.6				-13.732746	24.2101844	1152.6
12	-19.2883117	19.4545788	1237.1	-17.3745186	24.5218263	1002.1				-13.7335217	24.2094646	1156.6
13	-19.29409	19.44753	1205	-17.3802132	24.5179494	1009.4				-13.7442301	24.2133225	1162.1
14	-19.2940544	19.4465934	1232	-17.3802434	24.518831	997.8				-13.7449284	24.2122784	1163.8
15	-19.2940753	19.4456468	1230	-17.3806603	24.5212611	1000.3				-13.74550507	24.2114748	1157.5
16	-19.293138	19.4507802	1225.9	-17.3622583	24.5229649	1000.7				-13.7563708	24.2198775	1153.6
17	-19.2929638	19.4517413	1230.4	-17.3618332	24.5249288	1000.8				-13.756683	24.2188616	1152
18	-19.2928277	19.453208	1228.5	-17.3607848	24.5252138	1005.7				-13.7570401	24.2179485	1148.2
19	-19.2933095	19.4497659	1230.6									

Appendix 3: Methodology extras

Appendix 3.1: Fire frequency across study sites

Study site	Number of burns per plot (2001 – 2015)																		
	i	ii	iii	iv	v	vi	vii	viii	ix	x	xi	xii	xiii	xiv	xv	xvi	xvii	xviii	xix
Kanovlei	5	5	5	5	5	5	6	6	6	4	4	4	4	5	6	3	3	3	5
Sesheke	3	3	3	8	8	9	5	5	5	4	4	4	1	1	1	9	9	9	ns
Caprivi	2	2	2	7	7	9	7	9	9	9	ns	ns	ns	ns	ns	ns	ns	ns	ns
Kabompo	8	8	12	1	1	2	2	3	6	4	3	3	1	1	1	2	0	0	ns

Note:

Plot numbers are in Roman numerals (i, ii....xix)

NS: plot not sampled

Fire frequency classification: 0 burn: - none (no fire on the plot),

1 -3 burns: - Low

4 – 6 burns: - Medium

> 6 burns: - High

Appendix 3.2: Formulae used for species community parameters

Abundance: total number of individuals of each species in the total area of the sample.

Basal area: cross-section area of tree stems.

Density: average number of individuals of a species on a unit area basis.

$$D = \frac{\text{Number of individuals in the sample}}{\text{Total area of individuals of the sample (m}^2\text{)}}$$

Dominance: area a species occupies in a stand (for trees use basal area) on a unit area basis.

$$Dom = \frac{\text{Basal area of individual species in the sample (m}^2\text{)}}{\text{Total area of the sample (m}^2\text{)}}$$

Frequency: distribution of a species through a stand, i.e. percentage of plots in the sample area in which a given species occurs.

$$F = \left(\frac{\text{Number of plots in which a species occurs}}{\text{Total number of plots in a sample}} \right) \times 100$$

Appendix 4: Figures and Tables from results

Appendix 4.1: List of all species recorded across study sites

Species	Family	Study site
<i>Ozoroa okavangensis</i> R. Fern. & A. Fern.	Anacardiaceae	Caprivi
<i>Rhus tenuinervis</i> Engl.	Anacardiaceae	Sesheke
<i>Friesodielsia obovata</i> (Berth.)Verdc.	Annonacea	Sesheke
<i>Uvariastrum hexaloboides</i> R.E. Fr.	Annonacea	Kabompo
<i>Xylopiya odoratissima</i> Welw. Ex Oliv.	Annonacea	Kabompo
<i>Steganotaenia araliacea</i> Hochst.	Apiaceae	Sesheke
<i>Diplorhynchus condylocarpon</i> (Müll.Arg.) Pichon	Apocynaceae	Caprivi, Kabompo
<i>Strophanthus welwitschii</i> (Baill.) K. Schum.	Apocynaceae	Sesheke
<i>Afzelia quanzensis</i> Welw.	Caesalpinioideae	Kabompo
<i>Baikiaea plurijuga</i> Harms	Caesalpinioideae	Sesheke, Caprivi
<i>Bauhinia petersiana</i> Bolle	Caesalpinioideae	Kanovlei, Caprivi, Kabompo
<i>Brachystegia longifolia</i> Benth.	Caesalpinioideae	Kabompo
<i>Brachystegia speciformis</i> Benth.	Caesalpinioideae	Kabompo
<i>Burkea africana</i> Hook.	Caesalpinioideae	Kanovlei, Caprivi, Kabompo
<i>Cryptosepalum exfoliatum</i> De Wild.	Caesalpinioideae	Kabompo
<i>Dialium engleranum</i> Henriq.	Caesalpinioideae	Kabompo
<i>Erythrophleum africanum</i> (Welw. Ex Benth.) Harms	Caesalpinioideae	Caprivi
<i>Guibourtia coleosperma</i> (Benth.) J. Léonard	Caesalpinioideae	Kabompo
<i>Julbernardia paniculata</i> (Benth) Troupin	Caesalpinioideae	Kabompo
<i>Boscia albitrunca</i> (Burch.) Gilg & Gilg-Ben.	Capparaceae	Caprivi
<i>Garcinia livingstoneii</i> T. Anderson	Clusiaceae	Kabompo
<i>Combretum celastroides</i> Welw. ex M.A. Lawson	Combretaceae	Sesheke
<i>Combretum herereonse</i> Schinz	Combretaceae	Sesheke
<i>Combretum molle</i> R. Ex G. Don	Combretaceae	Kabompo

<i>Combretum psidioides</i> Welw.	Combretaceae	Caprivi
<i>Combretum species</i> Loefl.	Combretaceae	Kanovlei, Caprivi
<i>Combretum zeyheri</i> Sond.	Combretaceae	Kabompo
<i>Pteleopsis anisoptera</i> (M.A.Lawson) Engl. & Diels	Combretaceae	Kabompo
<i>Terminalia sericea</i> Burch. Ex DC	Combretaceae	Kanovlei, Caprivi, Kabompo
<i>Diospyros botocana</i> Hiern	Ebenaceae	Kabompo
<i>Croton gratissimus</i> Burch.	Euphorbiaceae	Caprivi
<i>Maprounea africana</i> Müll.Arg.	Euphorbiaceae	Kabompo
<i>Pseudolachnostylis maprouneifolia</i> Pax	Euphorbiaceae	Sesheke, Kabompo
<i>Schinziophyton rautanenii</i> (Schinz) Radcl.-Sm	Euphorbiaceae	All sites
<i>Memecylon sapini</i> L.	Melastomataceae	Kabompo
<i>Acacia ataxacantha</i> DC.	Mimosoideae	Kanovlei, Sesheke, Caprivi
<i>Acacia erioloba</i> E.Mey.	Mimosoideae	Caprivi
<i>Acacia fleckii</i> Schinz	Mimosoideae	Caprivi
<i>Albizia adianthifolia</i> (Schumach.) W.Wight	Mimosoideae	Kabompo
<i>Albizia versicolor</i> Oliv.	Mimosoideae	Kabompo
<i>Dichrostachys cinerea</i> Wight & Arn	Mimosoideae	Sesheke, Caprivi
<i>Ficus sycomorus</i> L.	Moraceae	Sesheke
<i>Ochna pulchra</i> Hook.	Ochnaceae	Kanovlei, Caprivi, Kabompo
<i>Ximenia caffra</i> Sond.	Olacaceae	Caprivi
<i>Schrebera trichoclada</i> Welw.	Oleaceae	Sesheke, Kabompo
<i>Baphia bequaertii</i> De Wild.	Papilionoideae	Kabompo
<i>Baphia Massaiensis</i> Taub.	Papilionoideae	Kanovlei, Sesheke, Caprivi
<i>Dalbergia nitidula</i> Welw. Ex Baker	Papilionoideae	Kabompo
<i>Pericorpsis angolensis</i> (Baker) Meeuwen	Papilionoideae	Kabompo
<i>Philenoptera violacea</i> (Klotze) Schrire	Papilionoideae	Sesheke
<i>Philenoptera nelsii</i> (Schinz) Schrire	Papilionoideae	Caprivi
<i>Pterocarpus angolensis</i> DC	Papilionoideae	Kanovlei, Caprivi, Kabompo
<i>Pterocarpus lucens</i> Guill. & Perr	Papilionoideae	Sesheke
<i>Swartzia madagascariensis</i> Desv.	Papilionoideae	Caprivi, Kabompo
<i>Paropsia brazzeana</i> Baill.	Passifloraceae	Kabompo
<i>Securidaca longepedunculata</i> Fresen.	Polygalaceae	Kanovlei, Kabompo
<i>Faurea rochetiana</i> (A.Rich.) Pic.Serm.	Proteaceae	Kabompo
<i>Berchemia discolor</i> (Klotzsch) Hemsl.	Rhamnaceae	Sesheke

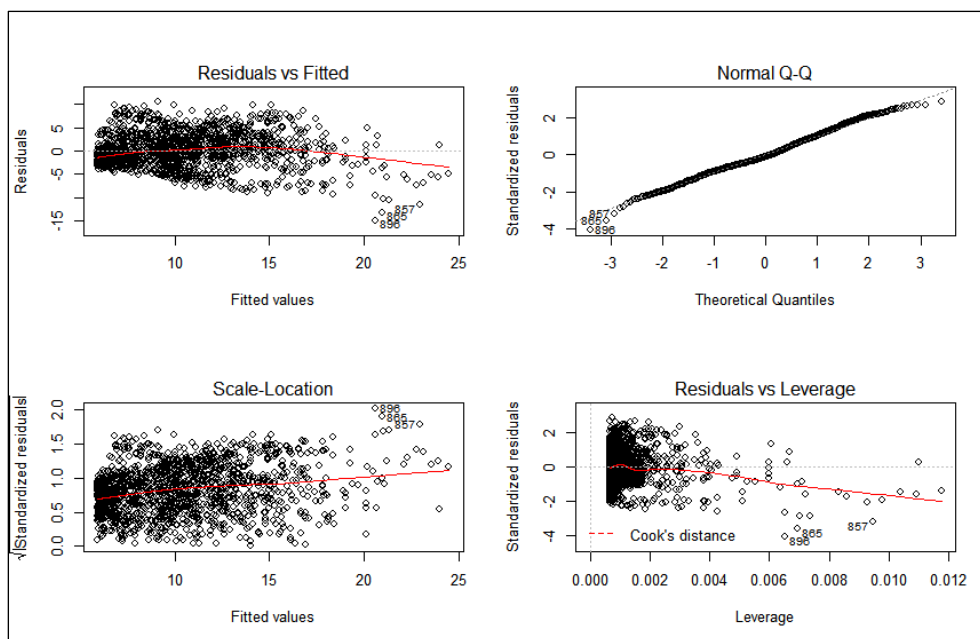
<i>Psyrdrax livida</i> (Hiern) Bridson	Rubiaceae	Sesheke, Caprivi
<i>Rothmannia englerana</i> Thunb.	Rubiaceae	Kabompo
<i>Vangueria infausta</i> Burch.	Rubiaceae	Caprivi
<i>Vangueriopsis lanciflora</i> (Hiern) Robyns	Rubiaceae	Kabompo
<i>Allophylus africanus</i> P.Beauv.	Sapindaceae	Sesheke
<i>Englerophytum magalimontanum</i> (Sond.) T.D. Penn	Sapotaceae	Kabompo
<i>Strychnos cocculoides</i> Baker	Strychnaceae	Sesheke, Kabompo
<i>Strychnos pungens</i> Soler.	Strychnaceae	Caprivi, Kabompo
<i>Strychnos spinosa</i> Lam.	Strychnaceae	Sesheke
<i>Grewia avellana</i> Hiern	Tiliaceae	Kanovlei, Caprivi
<i>Munyamatila</i>	Unidentified	Sesheke
<i>Namulomo</i>	Unidentified	Sesheke

Appendix 4.2: Regeneration limitation of selected species

Study site	Species	Regeneration limitation (%)		
		Recently burnt	Recently unburnt	Site overall
Kanovlei	<i>B. plurijuga</i>	100	100	100
Sesheke		94.4	94.4	94.4
Caprivi		16.7	92.9	70
Kabompo		100	100	100
Kanovlei	<i>B. africana</i>	75	61.1	68.4
Sesheke		100	100	100
Caprivi		66.7	92.9	85
Kabompo		55.6	44.4	50
Kanovlei	<i>G. coleosperma</i>	100	100	100
Sesheke		100	100	100
Caprivi		100	100	100
Kabompo		100	94.4	97.2
Kanovlei	<i>P. angolensis</i>	100	100	100
Sesheke		100	100	100
Caprivi		100	92.9	95

Kabompo		72.2	55.6	63.9
Kanovlei	<i>S. rautanenii</i>	95	100	97.4
Sesheke		88.9	100	94.4
Caprivi		83.3	100	95
Kabompo		66.7	100	83.3
Kanovlei	<i>T. sericea</i>	20	22.2	21.1
Sesheke		100	100	100
Caprivi		33.3	14.3	92.9
Kabompo		94.4	100	97.2

Appendix 4.3: Diagnostic plots for correlation of diameter and height data



Appendix 4.4: Glm test P-values of *B. africana* height SCD

Size class (m)	Fire return interval	MAP (Study site)	Interaction effect
0 – 0.5	0.092	0.000	0.002
0.51 - 1	0.918	0.000	0.710
1.01 - 2	0.043	0.078	0.999
2.01 - 3	0.952	0.068	0.326

3.01 - 5	0.133	< 0.0001	0.149
5.01 - 10	0.964	< 0.0001	0.000
10.01 - 15	0.005	0.003	0.029
15.01 - 20	0.999	< 0.0001	1.000
20.1 - 25	0.999	< 0.0001	1.000
> 25	n.a	n.a	n.a

Appendix 4.5: Glm test P-values of *T. sericea* height SCD

Size class (m)	Fire return interval	MAP (Study site)	Interaction effect
0 – 0.5	0.117	< 0.0001	0.161
0.51 - 1	0.889	0.006	0.003
1.01 - 2	0.991	0.000	0.022
2.01 - 3	0.003	< 0.0001	0.181
3.01 - 5	0.017	< 0.0001	1.000
5.01 - 10	0.999	< 0.0001	1.000
10.01 - 15	n.a	n.a	n.a
15.01 - 20	n.a	n.a	n.a
20.1 - 25	n.a	n.a	n.a
> 25	n.a	n.a	n.a

Appendix 4.6: Glm test P-values of *P. angolensis* diameter SCD

Size class (cm)	Fire return interval	MAP (Study site)	Interaction effect
5 – 10	0.120	0.008	0.662
10.1 - 15	0.048	0.0003	0.108
15.1 - 20	0.000	0.000	0.001
20.1 - 25	0.001	0.000	0.109
25.1 – 30	0.004	< 0.000	0.012
30.1 – 35	0.001	< 0.000	0.004
35.1 – 40	0.000	< 0.000	0.014
40.1 – 45	0.999	< 0.000	1.000
45.1 – 50	0.050	< 0.000	0.004
50.1 – 55	0.999	0.000	1.000
55.1 – 60	n.a	n.a	n.a

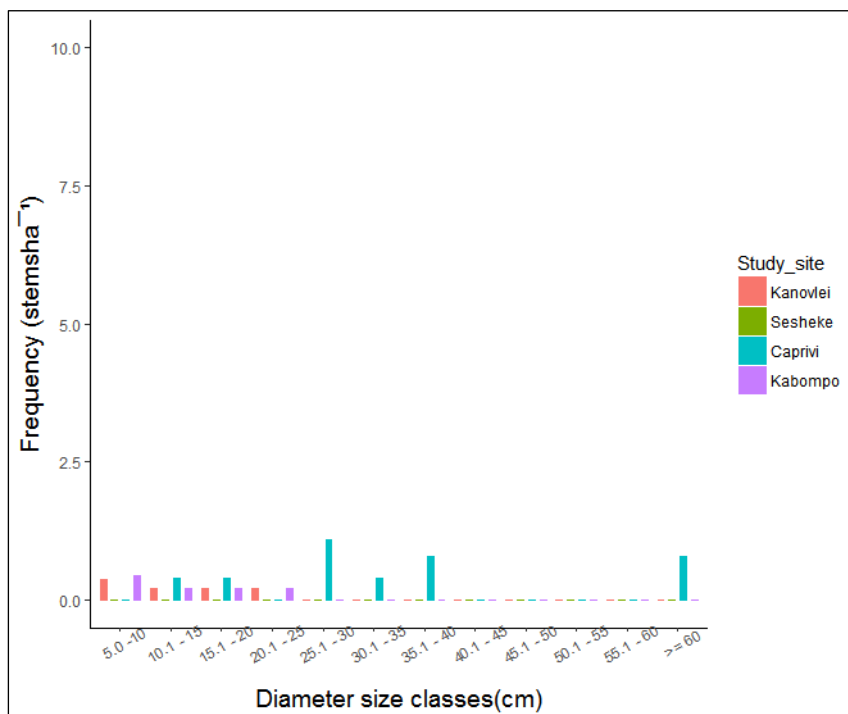
> 60

0.999

0.000

1.000

Appendix 4.7: Diameter SCD of *S. rautanenii*



Appendix 4.8: Post-hoc Dunn test on mean diameter of selected species

Species	p-value
<i>Baikiaea plurijuga</i> vs <i>Burkea africana</i>	0.000
<i>Baikiaea plurijuga</i> vs <i>Guibourtia coleosperma</i>	0.004
<i>Baikiaea plurijuga</i> vs <i>Pterocarpus angolensis</i>	< 0.0001
<i>Baikiaea plurijuga</i> vs <i>Schinziophyton rautanenii</i>	< 0.0001
<i>Baikiaea plurijuga</i> vs <i>Terminalia sericea</i>	0.001
<i>Guibourtia coleosperma</i> vs <i>Pterocarpus angolensis</i>	0.001
<i>Guibourtia coleosperma</i> vs <i>Schinziophyton rautanenii</i>	0.001
<i>Guibourtia coleosperma</i> vs <i>Terminalia sericea</i>	0.002

Appendix 4.9: The Anova results on mean height of all woody species.

Parameter	Sum Sq.	df	F-value	P-value
Fire frequency	17.927	3	4.491	0.008
Study site	290.237	2	109.064	<0.0001

Fire frequency*Study site	16.335	3	4.092	0.012
Residuals	61.207	46		

Appendix 4.10: The Anova results on mean height of selected species

Parameter	Sum Sq.	df	F-value	P-value
Fire frequency	393.40	3	14.336	<0.0001
Species	746.58	5	16.323	<0.0001
Fire frequency* Species	447.69	11	4.449	<0.0001
Residuals	887.29	97		

Appendix 4.11: Post-hoc Dunn test on basal area of selected species

Species	p-value
<i>Baikiaea plurijuga</i> vs <i>Burkea africana</i>	0.006
<i>Baikiaea plurijuga</i> vs <i>Guibourtia coleosperma</i>	0.001
<i>Baikiaea plurijuga</i> vs <i>Pterocarpus angolensis</i>	< 0.0001
<i>Baikiaea plurijuga</i> vs <i>Schinziophyton rautanenii</i>	< 0.0001
<i>Baikiaea plurijuga</i> vs <i>Terminalia sericea</i>	< 0.0001
<i>Burkea africana</i> vs <i>Schinziophyton rautanenii</i>	0.000
<i>Pterocarpus angolensis</i> vs <i>Schinziophyton rautanenii</i>	0.000

Appendix 4.12: Glm post-hoc test on canopy cover

Study site	p-value
<i>Kanovlei state forest</i> vs <i>Sesheke state forest</i>	0.001
<i>Kanovlei state forest</i> vs <i>Kabompo state forest</i>	0.001
<i>Sesheke state forest</i> vs <i>Caprivi state forest</i>	0.003
<i>Sesheke state forest</i> vs <i>Kabompo state forest</i>	0.001
<i>Caprivi state forest</i> vs <i>Kabompo state forest</i>	0.001