IMPACT OF AGRICULTURAL INTENSIFICATION ON BIODIVERSITY AND SECONDARY SUCCESSION IN THE MID-ZAMBEZI VALLEY, NORTHERN ZIMBABWE

By

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Abstract

In this thesis, the impact of woodland clearing, cultivation intensity and fallowing on tree species diversity, physiognomy and dominance was tested in the semi-arid mid-Zambezi valley, northern Zimbabwe. This study further tested the impact of agriculture on arthropod biodiversity using butterflies in the genus *Charaxes* (Lepidoptera: Nymphalidae) and beetles in the subfamily Cetoniinae (Coleoptera: Scarabaeidae). This study also investigated vegetation dynamics within recovering fallow areas. Results showed that, in both intensified and less intensified agricultural areas, conversion of natural woodland areas to cropped fields result in rapid decrease in tree diversity at field level, while at landscape level agricultural activities do not affect woody species diversity, at least in the short term. Instead, through selective cutting down of trees and subsequent crop cultivation negative impacts of woodland conversion is on tree physiognomy and dominance. At a higher trophic level, results showed that woodland clearing resulted in a decrease in abundance and loss of diversity of fruit-feeding butterflies of the genus *Charaxes* and beetles of the subfamily Cetoniinae with loss recorded in areas homogeneous intensively cultivated areas. Furthermore, results showed that, vegetation dynamics within fallow areas area characterised by a change in floristic composition with high rates of above-ground biomass accumulation driven by dominance of invasive species such as *Acacia tortilis* subsp *spirocarpa*. All sets of results in this study indicated that any efforts to conserve biodiversity within agricultural landscapes in the mid-Zambezi valley may have to focus on implementing an agricultural paradigm that maintains a mosaic of different land-use units, each in a different phase of clearance-cultivation-abandonment-recovery-clearance cycle.

Keywords: biodiversity; agroecosystems; bio-indicators; agriculture intensity; conservation, vegetation dynamics; biomass accumulation

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Declaration 1: Originality

I, Edwin Munyaradzi Tambara declare that the thesis/dissertation, which I hereby submit for the degree of Master of Philosophy at the University of Zimbabwe, is my own work and has not previously been submitted by me for a degree at this or any other tertiary institution. I further declare that all sources cited or quoted are indicated by means of a comprehensive list of references.

SIGNATURE:

DATE: September 2012

Declaration 2: Publication

Details that form part and/ or include research presented in this thesis include publications in preparation, submitted, in press and published and give details of the contributions of each author to the experimental work and writing of each publication.

Publication 1

Tambara E.[1], Murwira A.[2], Kativu S.[3], Torquebiau E.[4] (2012) **Farming does not necessarily conflict with tree diversity in the mid-Zambezi valley, Zimbabwe.** *Agroforestry Systems,* **84** (2): 299-309.

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Biodiversity can simply be defined as the total richness and variability of life on earth (Gaston, 1996; Altieri, 1999; Jeffries, 2006; World Bank 2006; Brussaard *et al.*, 2010). Within an ecological context, several workers, among them Hooper *et al.* (2005), Gaston (1996) and Mooney (2002) defined the term biodiversity as a concept that encompasses a broad spectrum of biotic scales, from genetic variation within species and habitats, to biome distribution on earth. It contributes to ecosystem structure, stability and productivity as well as facilitates functioning of ecosystems and drives such ecological processes as primary productivity, biogeochemical cycling, pollination, seed dispersal, regulation of climates, hydrological cycling and recycling of nutrients (Power and Flecker, 1996; Mooney, 2002; Scholes and Biggs, 2004; Hooper *et al.*, 2005; Hendrickx *et al.*, 2007; Colwell, 2009). These regulating functions are progressively lost under agriculture (Swift and Anderson, 1993; Naeem *et al.*, 1999). The net outcome of biodiversity loss is the replacement of natural ecosystems by artificial ecosystems that require human manipulation to fulfill ecosystem processes (Altieri, 1994).

In agricultural landscapes, biodiversity is under threat from various pressures, which include use of agrochemicals, cattle grazing, soil erosion, increased collection of firewood, invasive alien species, over-exploitation, pollution and contaminants, incidental mortality, disease, forest fires, mono-culturing and climate change (Fritz *et al.*, 2003; Sodhi *et al.*, 2004; Mattison and Norris, 2005; Butler and Laurance, 2008; Haines-Young, 2009; LERU, 2010; Perrings, 2010). Although some species, such as ruderal herbs respond positively disturbed landscapes (Walkowiak and Simonetti, 1980), the majority only show limited tolerance of increasingly widespread and rapid changes to agroecosystems (Naeem *et al.*, 1999). It is important, therefore, that conservation concerns be accommodated within agricultural practice.

1.1. Integrating crop farming and biodiversity conservation

The greatest challenge for conservation practitioners and land-use managers is the achievement of a balance between ever-growing demand for agricultural products and biodiversity conservation (Tilman *et al.*, 2001; Harvey *et al.*, 2008; Gardner *et al.*, 2009). “Land sparing” and “wildlife-friendly” farming are presently two conflicting strategies that aim to balance conservation and agricultural production (Matson and Vitousek, 2006; Fischer *et al.*, 2008). The two paradigms stem from different scientific views, thus posing difficulties in implementing biodiversity conservation in agroecosystems (Fischer *et al.*, 2008).

Land sparing involves intensification of existing cropland, and separation of reserves aimed at biodiversity conservation. In contrast, wildlife-friendly farming integrates conservation and crop farming within landscapes. In essence, wildlife-friendly farming emphasizes heterogeneity, resilience, and ecological interactions between farmed and unfarmed areas (Fischer *et al.*, 2008). The key feature of agricultural intensification is the reduced number of crops species, which often leads to monoculture (Giller *et al.*, 1997; Matson *et al.*, 1997). Intensification of crop agriculture is, thus often perceived to be a major threat to biodiversity in forest and woodland ecosystems (Debinski and Holt, 2000; Balmford *et al.*, 2005; Bruckmann *et al.*, 2010; Kadoya and Washitani, 2011). Today, the common landscape pattern within many tropical areas is woodland patches embedded in an agricultural mosaic matrix (Teodoro *et al.*, 2011). Information on whether and how much these agricultural landscapes in developing countries are conserving biodiversity is, however, still rudimentary.

The two paradigms, land-sparing and wildlife friendly farming, have unique advantages and disadvantages which present difficulties of implementation in integrating crop farming and biodiversity conservation (Fischer *et al.*, 2008). For example, with intensified agriculture, the use of fertilizers, herbicides and pesticides may pose a serious threat to non-targeted organisms and decrease the permeability of agriculture matrix by isolating populations living in natural patches (Goulart *et al*., 2009; Goulart *et al*., 2011), while with wildlife friendly farming, agricultural yields tend to be lower per unit area (Green *et al*. 2005). Thus a larger land area is typically needed to produce the same agricultural yield. In order to prescribe the ideal paradigm that suits a particular landscape, there is need to understand the nature of biodiversity response to the current cultivation practices. This focus raises the need to assess the impact of crop cultivation and farmland expansion on different aspects of biodiversity within agricultural landscapes and adjacent areas.

1.2. Impact of crop cultivation on woody species community structure and diversity

Through clearance of land for crop cultivation, subsequent abandonment, and selective harvesting of trees for various purposes, farmers directly affect woodland cover. Thus, land clearance for crop cultivation affects biodiversity directly through habitat conversion, and indirectly through fragmentation and alteration of flows of energy, materials and organisms (Giller *et al*., 1997). The impact of primary woodland loss on species diversity is, however, still poorly understood, given the enormous variety of taxonomic groups with possibly differing response to human disturbance (Beck and Schulze, 2000).

To understand the impact of crop farming on biodiversity, it is important to first focus on woody species which, as primary producers, form the keystone component in most trophic cascades (Woodwell and Whittaker, 1968; Persson, 1999; Cauwer *et al*., 2006; Thebault and Loreau, 2006; Ihenyen *et al*., 2010). Forest or woodland conversion is known to follow two alternative patterns: (a) floristic conversion that involves qualitative and quantitative changes in vegetation composition (Mafuta and Makuvise, 2000; Chinuwo *et al*., 2010), and (b) physiognomic conversion that is more common, and involves changes in vegetation structure (Crawley, 1997; SADC/IUCN/SARDC, 2000). Scientific studies have suggested that, the effect of crop agriculture on woodlands mainly results in negative changes in floristic composition (Bierregaard *et al*., 1992; Oba *et al*., 2002; Ogunleye *et al*., 2004 ; Augusseau *et al*., 2006). To the our knowledge, at least within tropical landscapes, such change is not confined to floristic composition, neither is it always negative. Thus, there is need to test hypotheses on such ecological patterns as tree diversity respond to woodland clearing and crop cultivation.

1.3. Biological indicators of agricultural ecosystem disturbance

It is important to test whether the impact of woodland clearing and crop farming is limited to primary producers (woody species), or is felt at a higher trophic level (primary consumer level such as arthropods). In studies focusing on assessing impacts of ecosystem disturbance on biodiversity, it is not ecologically sound to make observations at one trophic level or on a single species and use them as a conservation umbrella for other trophic levels (Kremen, 1992; Hilty and Merenlender, 2000; Maes and Van Dyck, 2005). The reason for this is that different species at different trophic levels respond differently to a given ecosystem disturbance (Landres *et al*., 1988; Prendergast *et al*., 1993). Within landscapes of conservation importance, it is critical to consider all levels of biodiversity in addressing conservation concerns (Kremen, 1992; Maes and Van Dyck, 2005).

At any given time or scale, it has proved difficult to employ total biota in assessing ecosystem disturbances on a large area (Brown, 1991; New, 2005). Reduced sets of taxonomic groups are, therefore, used as bioindicators (McGeoch, 1998; Niemela *et al*., 2000; McGeoch *et al*., 2002; Heink and Kowarik, 2010). Although the effectiveness of the concept of biological indicators has rarely been tested (Andelman and Fagan, 2000; Maes and Van Dyck, 2005), indicator species have been used for decades as a convenient assay of environmental conditions (Landres *et al*., 1988). In essence, bio-indicators are species sensitive to slight ecosystem changes in a predictive manner that allows the detection and measurement of the effect of various anthropogenic pressures on ecosystems (Paoletti, 1999; Bouyer *et al.*, 2007). Confounded criteria used to select species, and absence of precise definitions and procedures, severely weaken the effectiveness and credibility of biological indicators (Noss, 1999).

Several studies (Beccaloni and Gaston, 1995; Beck and Schulze, 2000; Fermon *et al.*, 2000; Ramos, 2000; Thomas, 2005; Bobo *et al.*, 2006; Bouyer *et al.*, 2007) investigated the relation between ecosystem disturbance and fruit-feeding insects. Although, the concept of bioindicators has found considerable application in other tropical countries, its application and documentation in Zimbabwe and the region is still limited. In the present study, therefore, bioindicators are used in answering the following question: "How does insect diversity respond to cultivation-generated ecosystem disturbance within tropical landscapes?".

1.4. Vegetation dynamics within recovering fallow areas

It has long been established that, as part of an agricultural system, vegetation in fallows provide services and functions in the landscape, which may contribute to food security or aid in carbon recovery and storage (Aweto, 1981; Breman and Kessler, 1995). This is because fallow vegetation facilitates the recovery of the biogeochemical stability as it has a fundamental effect on other natural processes, such as the restoration of soil nutrition, primary productivity and water regulation. More importantly, fallow vegetation can act as refugia for relict species through creation of habitats within agricultural landscapes. Today fallow vegetation remains important for smallholder farmers who strongly rely on natural fertility of the soil (Baudron *et al.*, 2009). Recent population pressure in the tropics has led to land scarcity, in many areas, which in turn has brought about the shortening or complete abandonment of the crucial fallow period, resulting in decrease in soil fertility (Peters and Neuenschwander, 1988; Sarmiento *et al.*, 2003). In the end, this has resulted in poor crop yields and further clearing of forests for fertile arable land.

The critical question that remains unanswered is, "is ecosystem disturbance through crop agriculture in tropical areas, bringing in novel ecosystems (also termed "emerging ecosystems", e.g. see Milton, 2003 and Hobbs *et al*., 2006)?". There are two main processes which may lead to emergence of novel ecosystems, both degradation and invasion of natural habitats or abandonment of intensively managed systems (Hobbs *et al*., 2006). As various efforts are being made to integrate agriculture with biodiversity conservation in tropical areas, such as the mid-Zambezi valley, it is important to understand the role of fallow vegetation. An important step towards this process is an in-depth study on the mechanisms, rates and pathways of secondary succession dynamics of fallow vegetation in this ecosystem. This, will give an insight into whether agricultural disturbance is pushing the ecosystem towards a threshold to a new or different state which could be transient or stable.

1.5. Thesis objectives

The main objective of this study was to evaluate how agricultural intensification affects woody species diversity and physiognomy along an agricultural intensification gradient, with examples being provided by woody vegetation species. In addition, the study aimed to establish changes in diversity and abundance of fruit-feeding arthropods. Also this study investigated prospects of ecosystem recovery by assessing vegetation dynamics in fallow areas. To achieve these objectives, the study was subdivided into three subthemes. The study tested the general hypothesis that biodiversity at varying trophic levels is progressively lost under progressive agricultural intensification.

1.5.1. Specific objectives

Specifically, the study tested the predictions that biodiversity decreases with increasing agricultural intensity and that woodland structure and composition are permanently altered by cultivation. Thus,

1. Woody species diversity, physiognomy and dominance were assessed within three land-use zones along an agricultural intensification gradient.
2. Diversity and abundance of bio-indicator fruit-feeding arthropods, were investigated in different habitats, in order to determine the impact of habitat conversion on arthropod diversity, and
3. Vegetation dynamics in fallows on clayey (*mutapo*) and sandy (*bandate*) soils, were analysed by the change in woody species richness, dominance, life-form spectrums and above-ground biomass accumulation.

1.5.2. Research Questions

To answer the broad question on how biodiversity respond to agricultural intensification, the following questions were raised:

1. How does woodland conversion for cultivation purposes affect woody species?
2. What is the response of arthropod species to loss and conversion of habitats for agricultural purposes?
3. What are the vegetation dynamics within recovering fallow areas?
4. What are the above-ground tree biomass accumulation rates within fallow areas on different soils?
5. What are the inter-annual variations in biomass accumulation in fallows?
6. Is the vegetation within fallows indicating a recovery towards the natural vegetation state or emergence of a new different state?

1.5.3. Justification of study

Agricultural expansion in the mid-Zambezi valley is the primary process of increasing crop production. This is characterised by the increase in arable field sizes and reduction or complete abandonment of fallowing practices. Factors such as, increased human immigration, increased access to structured markets, agricultural inputs and implements, and the current development in infrastructure have translated into increased demand for arable land in the mid-Zambezi valley. Thus, the observed trend has been that of increased expansion of agricultural landscapes through conversion of natural woodland areas into farmland (Murwira *et al*. 2010). Moreover, in some parts of the valley agricultural activities have gone beyond expansion to intensively cultivated landscapes characterised by use of pesticides and herbicides, increased labour input per unit area, high livestock density and monocultures with a strong bias towards cotton farming (Baudron *et al*. 2011). The expected result, therefore, will be continual disappearance and or over simplification of habitats, accompanied by a decline in biodiversity at all trophic levels.

Past studies in the mid-Zambezi valley mainly focused on assessing change in land use, general biodiversity assessments and ways of improving agricultural yield (Derman, 1996; Biodiversity Project, 2002; Baudron *et al*., 2009; Baudron *et al*., 2011a; Baudron *et al*., 2011b). Few studies (Cumming and Lynam, 1997; Fritz *et al*., 2003; Poilecot and Gaidet, 2010) focused on response of biodiversity during and after human activities. Very few, if any, focused on the response of plant and small animal taxa to the continued loss of habitats due to agricultural expansion and intensification. The present study critically examines how the dominant farming practice (agriculture intensification) affects biodiversity in a landscape of global importance for biodiversity conservation, the mid-Zambezi valley. Thus, findings from the present study could assist in formulating policies for conservation practices and biodiversity recovery processes.

1.6. Thesis outline

This thesis consists of seven chapters in total. Chapter 2, gives a description of physical attributes of the study area and defines the agriculture intensification gradient; Chapter 3, the focuses on the study design used in the whole study and provides a description of the materials and methods used in data collection. Chapters 4 to 6 address components of the research questions and are similar in style to separate papers. Chapter 7 is a synthesis for the whole thesis.

Chapter 4: is a study addressing the following questions: (1) what is the role of soil type on woody species diversity?; (2) does cultivation have any significant effect on woody species diversity or, alternatively, tree species physiognomy?; (3) does cultivation intensity significantly affect woody species diversity and physiognomy?; and (5) what is the relationship between period of cultivation and woody species diversity?

Chapter 5: explores the effect of habitat conversion through woodland clearing for cultivation purposes on butterfly (Nymphalidae: *Charaxes*) and beetle (Coleoptera: Scarabaeidae: Cetoniinae) species richness and abundance along a gradient of cultivation intensity.

Chapter 6: explores and compares secondary succession and above-ground biomass accumulation between fallow areas with clayey (*mutapo*) and sandy (*bandate*) soils, analysing the change in woody species richness, dominance and in the life-form spectrums of the vegetation.

Chapter 7: synthesizes findings of the thesis and discusses the implications of these with regards to agricultural production and biodiversity conservation in the mid-Zambezi valley and similar areas.

The mid-Zambezi valley partly occurs in Mozambique, Zambia and Zimbabwe. This study was carried out in Mushumbi Pools and Angwa areas that are located within communal lands of Dande in the Mbire Rural District Council area, Zimbabwe (Figure 2.1). Communal lands are a land category characterized by collective or community land ownership and they are subdivided into administrative or management units called wards. Farmers in the communal lands have user rights that only prohibit sale of land otherwise they own the land and are able to transfer it to their siblings. The study area in Angwa consists of Ward 2 (781km²) and in Mushumbi Pools it consists of Ward 3 (356km²) and Ward 9 (Biodiversity Project, 2002). The mid-Zambezi valley is a semi-arid, low lying area, with an average altitude of 400 meters above sea level (a. s. l.). It extends between longitudes 30° east and 31° east, and latitudes 15° 30 south and 16° 20. It is dominated to the south by an escarpment (Mavuradonha Mountain) which rises to 1400 m a. s. l., and stretches northwards to the Zambezi River.

2.1.1. Climate and hydrology

Zimbabwe is divided into agro-ecological regions based on rainfall, temperature and soil capability (Vincent and Thomas, 1962). The study area lies within Agro-ecological Region IV. This region is characterized by a tropical dry climate, with a mean annual temperature of 25°C and minimum and maximum temperatures of 10°C (June and July) and 40°C (October and November), respectively (Vincent and Thomas, 1962). The area has two distinct seasons: a rainy season (November to March) and a dry season (April to October). The rainy season receives rainfall amounts that range from 350 mm to 650 mm, falling within a 36 day period (Vincent and Thomas, 1962). The area is composed of former floodplains of the Zambezi River, with dense, complex hydrological network. Three perennial rivers (Angwa, Kadzi and Manyame) dominate the hydrology of the area, and they all flow across Dande Communal Lands into the Zambezi River.

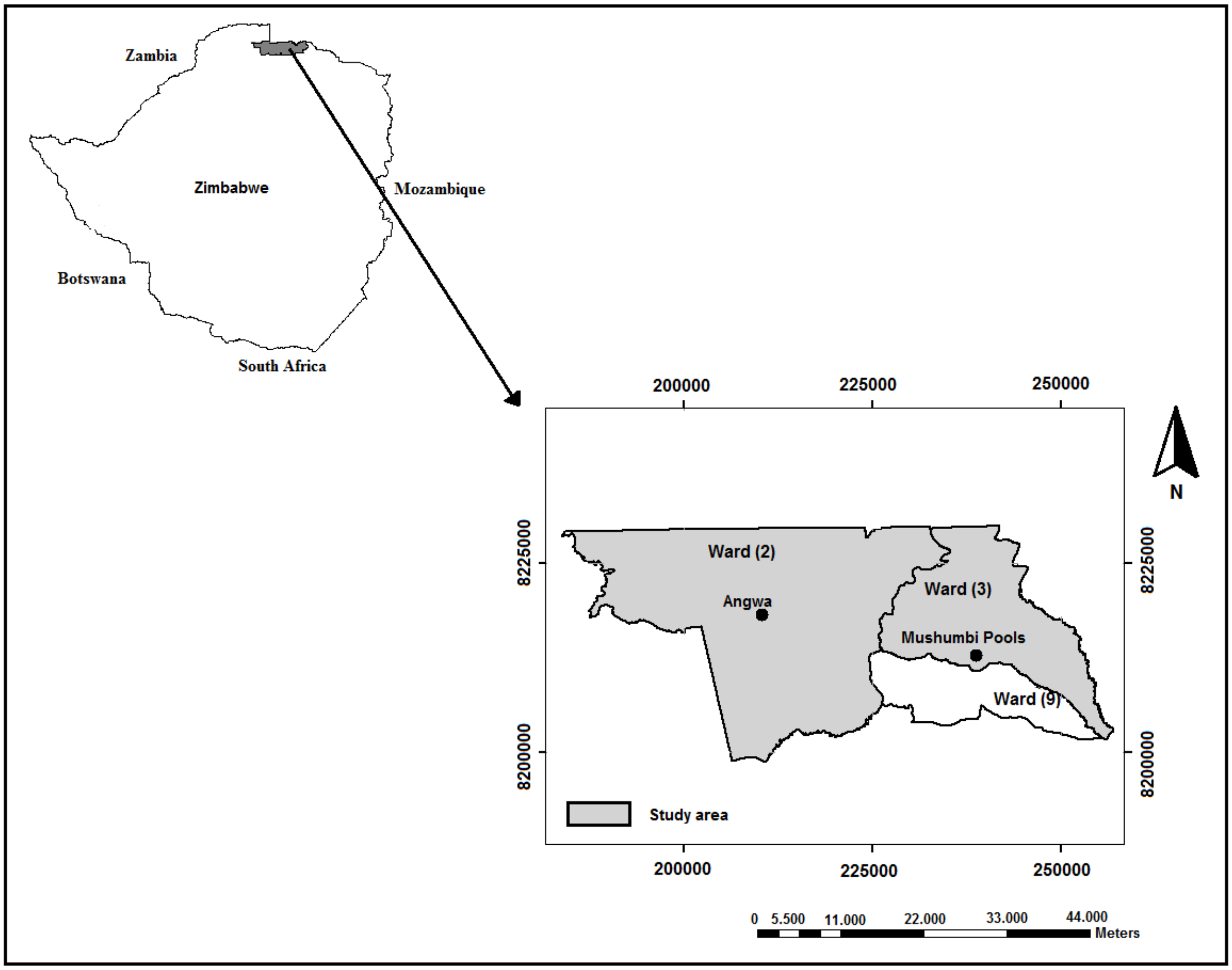


Figure 2.1. Study area covering Ward 2 and 3 in Mbire area in the mid-Zambezi valley. Map coordinates are in metres UTM zone 36 South.

2.1.2. Flora and fauna

The mid-Zambezi valley is host to an estimated 700 plant species (Timberlake, 1988; Biodiversity project, 2002). On a broader note, the natural vegetation cover of the area is deciduous, dry savannah, and is dominated by *Colophospermum mopane* (Figure 2.2), with associations of *Acacia nilotica*, *Adansonia digitata*, *Combretum elaeagnoides*, *Diospyros kirkii*, *Kirkia acuminata*, *Sclerocarya birrea*, *Terminalia brachystemma*, *Terminalia sericea*, *Terminalia stuhlmanni* and *Ziziphus mucronata* (Timberlake *et al*., 1988; Gaidet *et al*., 2003; Fritz *et al*., 2003). Other dominant woodland types in the area include: the *Diospyros kirkii/Combretum apiculatum* low open woodland (Figure 2.3), commonly occurring on shallow sandstone soils, *Brachystegia allenii*/*C. mopane* open woodlands on stony and deep soils, and *Julbernardia*/*Combretum*/mopane woodland on rocky sandstone slopes.

The area still hosts an important diversity of mammals, several of which are emblems of big African game (Frizt *et al*., 2003; CIRAD, 2004), with more than 40 species of large mammals and 200 bird species. Invertebrate groups partially studied for the area include Arachnids (127 species from 27 families), Coleoptera and Lepidoptera (90 species listed, mainly from Nymphalidae, Papilionidae and Lycaenidae families). The area includes an abundance and diversity of invertebrates of economic importance such as the mopane worm, *Imbrasia belina* (Gardiner and Niemeyer, 1998; Biodiversity Project, 2002; Gondo *et al*., 2010).

Figure 2.2. Mopane woodland dominated by *Colophospermum mopane*

Figure 2.3. *Diospyros kirkii*/*Combretum apiculatum* open woodland

2.1.3. Soil

The underlying geology of the area consists mainly of Dande sandstone. Soils vary, but are generally sandy, locally shallow, often rich in sodium, but lacking organic matter (Mvuriye *et al.*, 2001). Agricultural activities are primarily carried out on two soil types, *mutapo* and *bandate* in Mushumbi Pools and Angwa. *Mutapo* soils are eutrophic, and associated with sodic/saline areas. The soils are heavy, with high moisture holding capacity, and a depth reaching up to 3 meters (Mvuriye *et al.*, 2001). *Bandate* soils are moderately heavy soils that make huge dust clouds when ploughed, and have high moisture retention capacity. Fertility rate in the latter is high, and the soil gives good yields of cotton, maize, millet and sorghum, particularly during drier years (Baudron *et al.*, 2011a). Baudron *et al*. (2012) provided a detailed analysis on characteristics of the two soil types (Table 2.1)

Table 2.1. Soil characteristics of the two dominant soil types, *mutapo* and *bandate*, used for crop farming in the study area.

Standard errors are given after the signs '±'.

Soil type[*]	**Depth (cm)**	**pH KCL**	**SOC (g/kg)**	**N (mg/kg)**	**P (cmol/kg)**	**K (cmol/kg)**	**Clay %**	**Silt %**	**Sand %**
Bandate	0-10	6.2 ± 0.5	10 ± 3.2	0.8 ± 0.3	17.3 ± 13.4	0.68 ± 0.3	13.6 ± 4.0	18.2 ± 11.2	68.2 ± 14.6
	10-20	5.9 ± 0.6	8.7 ± 2.8	0.7 ± 0.2	11.7 ± 11.8	0.54 ± 0.3	15.5 ± 5.3	18.0 ± 10.8	66.4 ± 15.3
Mutapo	0-10	6.1 ± 0.5	7.7 ± 3.0	0.6 ± 0.2	11.8 ± 8.2	0.80 ± 0.3	14.9 ± 3.0	17.5 ± 5.5	67.7 ± 7.5
	10-20	5.9 ± 0.5	7.4 ± 2.6	0.6 ± 0.2	8.26 ± 5.7	0.69 ± 0.3	16.5 ± 3.9	17.9 ± 5.6	65.6 ± 8.5

[*] Local names, **Source**: Baudron *et al*. (2012)

2.1.4. Gradient of human population and anthropogenic disturbances

Human population in Mbire District has increased exponentially with time (Pwiti, 1996; Biodiversity Project, 2002; Baudron *et al.*, 2011a). Tsetse fly eradication and associated provision of roads and infrastructures were some of the major drivers of population increase in the mid-Zambezi (Chizarura, 2003). Tsetse eradication, the primary factor, was however, not uniform across the landscape. Human settlement in the area, therefore, followed a gradient of tsetse fly eradication (Cumming and Lynam, 1997; Derman, 1996; Murwira *et al.*, 2010). A significant increase in human population was recorded between 1992 and 2002 in Mbire District. For instance, the total population in Wards 2 and 3 increased from 9872 in 1992 to 14109 in 2002, and 20312 in 2003 (Biodiversity Project, 2002). In 2002, population density in wards (Ward 3 and 9) around Mushumbi Pools ranged from 29.2 to 42.9 inhabitants km^{-2}, while for Ward 2 around Angwa, it was 5.7 inhabitants km^{-2} (Baudron *et al.*, 2011a). The increase was 1.5 times greater in Mushumbi Pools than in Angwa (Chizarura, 2003). Immigration in the study area was actively promoted by the Mid-Zambezi Resettlement Development Project (MZRDP), a government sponsored project that was earmarked to resettle 3000 families from degraded, overcrowded communal lands (Pwiti, 1996; Ivy, 1998). Each family received a one-acre residential plot and a 12-acre plot for farming (Derman, 1996).

2.1.5. Deforestation

Densely populated areas, such as Mushumbi Pools, have seen exploitation of natural woodlands going beyond mere gathering of dead wood to deliberate and indiscriminate cutting of live trees. People have to walk distances of more than 5 km from their homesteads in search of fuel wood in this part of the valley (Baudron *et al.*, 2009). Mushumbi Pools is also experiencing a rapid growth in fuel wood market, where people sell stakes of fuelwood along major roads. The most impacted vegetation type is mopane woodland which is intensively used by local communities

(Poilecot and Gaidet, 2010). Pressure on woodlands for fuelwood purposes is very low in the more remote Angwa area where people utilize fuelwood only for domestic purposes.

The need for agricultural land has emerged as the major driver of deforestation in the mid-Zambezi valley (Biodiversity Project, 2002; Baudron *et al.*, 2011a). Initial massive deforestation was brought about by the Mid-Zambezi Rural Development Programme (MZRDP) policies, particularly its policy of opening up arable lands which resulted in the burning and clearing of thousands of acres of forest (Derman, 1996). The situation still continues today largely due to farmers' practices, particularly larger-scale cotton growers (Biodiversity Project, 2002). Again the most impacted area is Mushumbi Pools, where agricultural landscapes are continuing to expand into natural woodlands.

2.1.6. Tsetse control

The valley has a long history of programmes aimed at tsetse fly (*Glossina morsitans morsitans* Westwood and *G. pallipides* Austen) and trypanosomiasis eradication (Derman, 1996). Eradication programmes involved the immigration and emigration of people and animals, killing of game, exclusion of cattle from infested areas, construction of game and cattle fences, development and use of trypanomicides, cattle dips, and more recently, the erection of targets to attract and kill tsetse flies. Up until the late 1980s, however, the greater part of the mid-Zambezi valley remained infested by the tsetse fly. Presence of the tsetse fly was perceived as a major limiting factor for smallholder agricultural development in the area. In order to stimulate development, large scale operations of aerial and ground spraying of insecticides were conducted in the mid-1980s. This was gradually, replaced by the more environmental-friendly method of deploying target traps (Derman, 1996). Tsetse control was intensive and more successful in the Eastern part of the valley, and was accompanied by massive human immigration into cleared areas. Mushumbi Pools is now tsetse-free, while West Angwa remains infested. East Angwa represents an intermediate area (Cumming and Lynam, 1997; Biodiversity Project, 2002; Baudron *et al*., 2011a).

2.1.7. Agriculture dynamics in the mid-Zambezi valley

Until the 1980s, the mid-Zambezi valley had remained an area of subsistence agriculture with deeply modified cropping patterns. According to Poilecot (2002), after Zimbabwe's independence in 1980, cropland in the mid-Zambezi valley quadrupled in less than 16 years. Today, farming is at the center of the valley's economy, with cotton, maize and sorghum being the dominant crops (Baudron *et al*. 2009). The level of agricultural production and farming practices in the mid-Zambezi valley are no longer homogenous across the landscape. Instead, agricultural activity follows the anthropogenic and tsetse gradient from north-west-west to south-east-east of the valley (Baudron *et al*. 2009; Baudron *et al*. 2011a).

The intensity of agricultural activity the mid-Zambezi valley has been defined using the average area under cultivation per household, number of crops grown, labour per cultivated area, livestock density, use of animal drawn ploughs and quantity of inorganic inputs (Baudron *et al*. 2011). Using these variables a gradient of agricultural intensification has been defined, characterised increasing intensity from West Angwa to East Angwa towards Mushumbi Pools (Baudron *et al.* 2009). Apart from the anthropogenic and tsetse gradients, access to markets and infrastructure are critical factors that strongly modified and developed the existing agricultural gradient. Mushumbi Pools being the center of the District and hosting cotton depots and other agroservices has been able to support agricultural activities through good infrastructure (e.g. roads, schools, government institutions, clinics), increased access to markets, inputs (e.g. fertilizers, hybrid seeds, pesticides, herbicides) and reduced transport costs for both inputs and outputs (Derman 1996; Baudron *et al*. 2009). To this effect farmers in Mushumbi Pools are more able to capitalize revenues from agricultural produce (particularly cotton) in animal traction, agricultural implements and inputs, allowing cultivation of more land per household.

The density of cultivated fields in West Angwa increases from an average surface per farm of 1.56 ha to 3.03 ha in East Angwa and 3.66 ha in Mushumbi Pools. In West Angwa, households have on average 96m of transect length under cultivation, 134m in East Angwa and 196m in Mushumbi Pools. The median of the cultivated area per working adult on the farm is also much lower in West Angwa than in East Angwa or Mushumbi Pools. The number of crops grown per household increases from an average of three crop types in Mushumbi Pools, were the commonly grown crops are, cotton, maize and sorghum, to five crop types in Angwa were crops grown include, cotton, maize, sorghum, millet, groundnuts, roundnuts and cowpea.

CHAPTER 3: Materials and Methods

3.1. Study design

The study was carried out along a belt transect cutting across the mid-Zambezi Valley, northern Zimbabwe (Figure 3.1). The belt transect (about 40 kilometers) represents a spatial agricultural intensification gradient transcending the Zimbabwean side of the valley, from Angwa towards Mushumbi Pools (Biodiversity Project, 2002; Baudron *et al*., 2009; Baudron *et al*., 2011a). An interpreted soil map was produced as part of the Biodiversity Project (2002) in the mid-Zambezi valley. The map was used in the present study to stratify the study area, Angwa and Mushumbi Pools, into two zones based on the two dominant soil types of the area, grey-white clays (*mutapo*) and sandy clay soil (*bandate*). Two sites, Mubairakuenda (Site 1) and Vinyu (Site 2), are located in Angwa and the other two, Shange (Site 3) and Nyambudzi (Site 4), are located in Mushumbi Pools (Figure 3.1).

At each site, a 1 km radius study area was demarcated after traversing and identifying areas at relatively similar elevation (using a handheld Global Positioning System: Garmin eTrex HGPS Navigator), rainfall, temperature and topography. In each study site three land cover types were identified: (1) an area where natural vegetation has been cleared for agricultural purposes, with the crops grown including cotton, sorghum, maize and millet, denoted as **cultivated area**; (2) previously cleared or cultivated areas of different ages with recovering vegetation, denoted as **fallow area**; and (3) woodland with no recent signs of disturbance, denoted as **natural woodland**. All fields and fallows within the four sites were located, and their area determined using a hand held GPS. Data on each field in the study area were captured from Baudron *unpublished* (2009); Biodiversity Project (2002), and included name of farmer, current crop, field age and fallow age. To determine field and fallow age, cultivation cycles, data on the history of each cultivated and fallow field was captured from previous studies (Baudron *unpublished*, 2009; Biodiversity Project, 2002) and more information was provided by key informants who has a good knowledge of the area, Mr Knowledge Mataya in Mushumbi Pools and Mr Edwin Chimusimbe in Angwa. All the information gathered for each field and fallow was compared to that provided by the farmers.

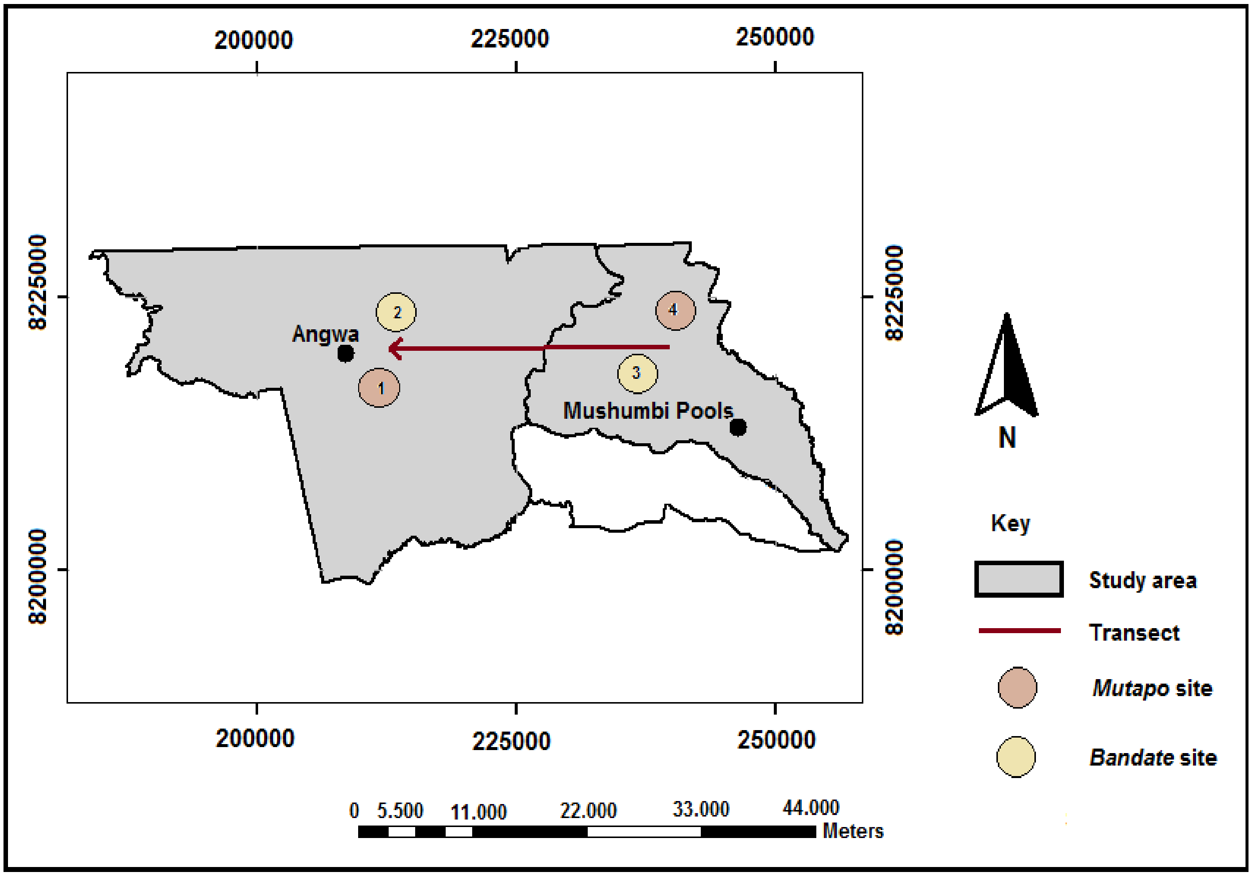


Figure 3.1. A map showing distribution of sites in the study area.

3.2. Data collection

3.2.1. Tree species data collection and analysis

Data were collected from three different categories at each site; cultivated, fallow and natural woodland (Table 3.1). The number of plots sampled was different from one site to the other due to differences in cultivation intensity among the four sites. The density of cultivated fields increased from site 1 to site 4 whilst that of fallows decreased. The average size of cultivated fields across the study area is 1.2 hectares and that of fallows was 0.97 ha. This difference in size, justified the establishment of larger sample plots (30 m x 100 m) in the cultivated fields and natural woodlands. Sampling plots were randomly selected from the field database for data collection in the cultivated fields and fallows. This was done to provide the basis for computing a valid estimate of the sampling error by justifying the assumption of independence of sample units over the fields. Randomisation was achieved by entering information on all plots into a spread sheet and arranging all cultivated fields and fallows by age, ranging from 1 to 30 years for cultivated fields, and 1 to 12 years for fallows. Fields or fallows within each particular age were allocated numbers and for each particular age, random selection of plots was achieved by generating random numbers from a calculator. Selection of sampling plots in the natural woodland was achieved by moving a distance of 300 meters from a randomly selected cultivated field into the adjacent natural woodland.

Table 3.1. Number of sampled fields, fallows and woodland plots in each site

	Cultivated fields	**Fallows**	**Natural woodland**	**Total**
Plot size (m)	30 x 100	30 x 30	30 x 100	
Site 1	19	10	16	**45**
Site 2	17	14	14	**45**
Site 3	27	9	22	**58**
Site 4	33	8	24	**65**
Total	**96**	**41**	**76**	

In this study, woody species were defined as tree plants in the form of stumps, seedlings, saplings, shrubs and adult trees. On each sampling plot, all woody species encountered were identified in situ by their botanical and local names and recorded. Species were identified by local name and also by botanical name using, Drummond and Coates-Palgrave (1973), Timberlake *et al*. (1993), Van Wyk and Van Wyk (1997), Coates-Palgrave (2002). Frequency of occurrence and tree size (height) in each sampling plot were used to determine tree dominance in each site. Ten dominant woody species were selected for detailed analysis of physiognomic status in each of the three land-use zones. Within each sampling plot, individuals of the ten dominant species were grouped by habit and height into four classes (1) seedlings and immature saplings, woody plants with a height less than 0.5 m (< 0.5 m); (2) stumps and saplings, woody plants with a height between 0.51 and 1 m (0.51–1 m) (3) shrubs, woody plants with a height between 1.1 and 3 m (1.1–3 m) and (4) mature trees, woody plants with a height more than 3 m (> 3 m). The physiognomic status of the dominant species in each land-use category was assessed on the basis of frequency of individuals within each size class. Observations were made to identify tolerated woody species within the agricultural landscape.

3.2.2. Arthropod data collection

An assessment was conducted to establish species richness and abundances in the genus *Charaxes* (Lepidoptera: Nymphalidae) and subfamily Cetoniinae (Coleoptera: Scarabaeidae). The effectiveness and credibility of these two arthropod groups as indicator species assemblages for monitoring ecosystem health has been tested in the W-Regional Park between Benin, Burkina Faso and Niger (Bouyer *et al*., 2007).

Sampling

Sites were stratified into two zones: agricultural area (Zone 1) and natural woodland (Zone 2). In total, eight land-use zones were sampled in the four sites. Two transects of approximately 1.6 km in length were demarcated in each zone. Transect 1: natural woodland interior (800 m from cultivated edge), transect 2: natural woodland (100 m from cultivated land into natural woodland to avoid edge effect), transect 3: cultivated (100 m from woodland edge into cultivated area to avoid the edge effect) and transect 4: cultivated interior (800 from woodland edge). The adopted distances were selected based on exploratory surveys which showed that at 100 m edge effects can be discerned whereas at 800 m none were detected. Along each transect, five trapping locations were placed some 400 m apart. In locations were trees to install traps were not on the transect line, traps were installed within a distance of 20 m from the transect line. In total, sixteen transects were laid out in the study area, with 80 trapping locations.

The selected insect fauna was trapped using attractive traps. Traps were designed and set as described by De Vries (1988), De Vries (1997), Bouyer *et al*. (1997) and, Fermon *et al*. (2000). The first trap (Cetoniinae, Figure 3.2a) consisted of a plastic water bottle (about 32 cm in height or 1.5 litres in capacity) into which two windows of about 8 x 5 cm were made in the middle part. A string for tying the container to a tree was attached to the bottle top. The second trap (*Charaxes*), consisted of a cylindrical net (mosquito net material) of 60 cm in height and 30cm in diameter, placed some 5 cm over a circular plank (30 cm diameter) with a string attached to the top side (Figure 3.2b). In total, 160 (80 of each type) banana baited traps (20 in each site) were installed at least 1.0 m above ground on tree branches. Figure 3.3 shows some of the installed traps in the study area.

In order to maximize sampling efficiency, sampling was confined to periods of adult insects' maximum activity and density. To this end, sampling was carried out during the hot and humid season between January and March 2011, which is the hatching season for the insects and therefore populations of adults were at peak. Each trapping location consisted of two traps, one for Cetoniinae and the other for *Charaxes* as described by De Vries (1988), Fermon *et al*. (2000), and Bouyer *et al*. (2007). Traps were installed at least 1.0 m above ground on tree branches. Trapping geographic location was recorded using a handheld Global Positioning System (Garmin eTrex H). Traps were set and monitored between the 10 and 22 of January 2011, 12 and 24 of February 2011 and between 5 and 18 of March 2011. Due to the distance between Angwa and Mushumbi, it was not possible to monitor traps in all sites in one day. Therefore, in all the three sampling periods, sites were sampled in pairs, Site 1 and Site 2 and then Site 3 and Site 4. Thus, in each site traps were emptied between 12.00 hrs and 14.00 hrs every second day.

Butterflies in the family Nymphalidae were identified using pictures and descriptions by D'Abrera (1980), Larsen (1996), and Picker *et al*. (2004) and. Species of genus *Charaxes* were identified from D'Abrera (1980), Henning (1989) and Larsen (1996). Cetoniinae species were identified from guides from the following: Beetles of Africa (http://www.beetlesofafrica.com) and Tree of Life (http://tolweb.org/Cetoniinae/105064). Species identification for both butterflies and beetles was verified by the assistance of Dominique Dulieu, an expert in these faunal groups. Habitat associations for species in the genus *Charaxes* were adopted from Larsen (1996). From each trap, insects were recorded by species and apparent density per trap (ADT) according to Bouyer *et al*. (2007). Butterflies and beetles species were identified in the field and released after marking (using small stickers) in order to avoid recording the same individuals. Insects were identified using the pictorial literature of Picker *et al*. (2004).

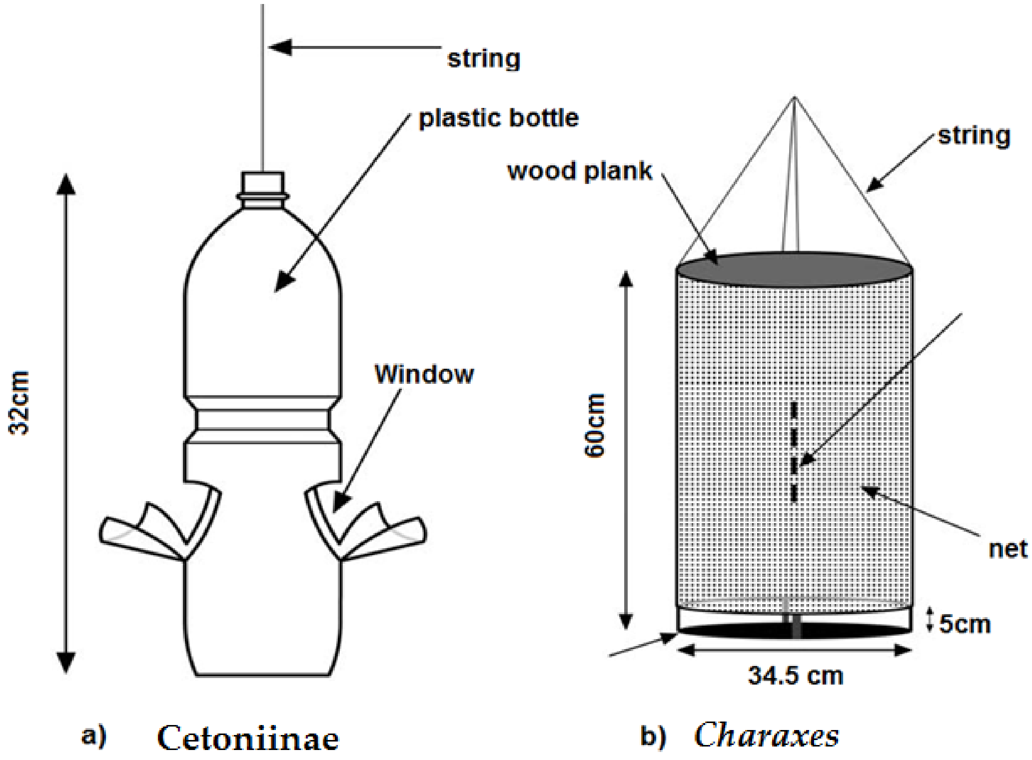


Figure 3.2. Banana baited Cetoniinae and *Charaxes* traps as described by De Vries (1988), Bouyer *et al*. (1997), De Vries (1997), and Fermon *et al.* (2000).

Figure 3.3. The two types of traps installed in the study area

3.2.3. Fallow vegetation dynamics

Data were collected from fallows of different ages on two soil types (*mutapo* and *bandate*), ranging from 1 to 12 years. In total 41 fallows were sampled, 19 on *mutapo* soil and 22 on *bandate* soil. In this study, each fallow was divided into four equal parts using a rope and each of the four blocks was allocated a number (from 1 to 4). Using random numbers generated from a calculator two blocks in each fallow were randomly selected and one sampling plot measuring 30 m x 30 m was established at the centre of each block (Figure 3.4). In each sampling plot, the following vegetation characteristics were determined: number of woody species, tree height, shrub density, tree density and tree diameter at breast height (dbh). Shrubs were defined as a woody plant with a height less than 3 m while a tree was defined as a woody plant with a height of more than 3 m and a diameter at breast height of > 5 cm. This part of the study was done between May and July 2010.

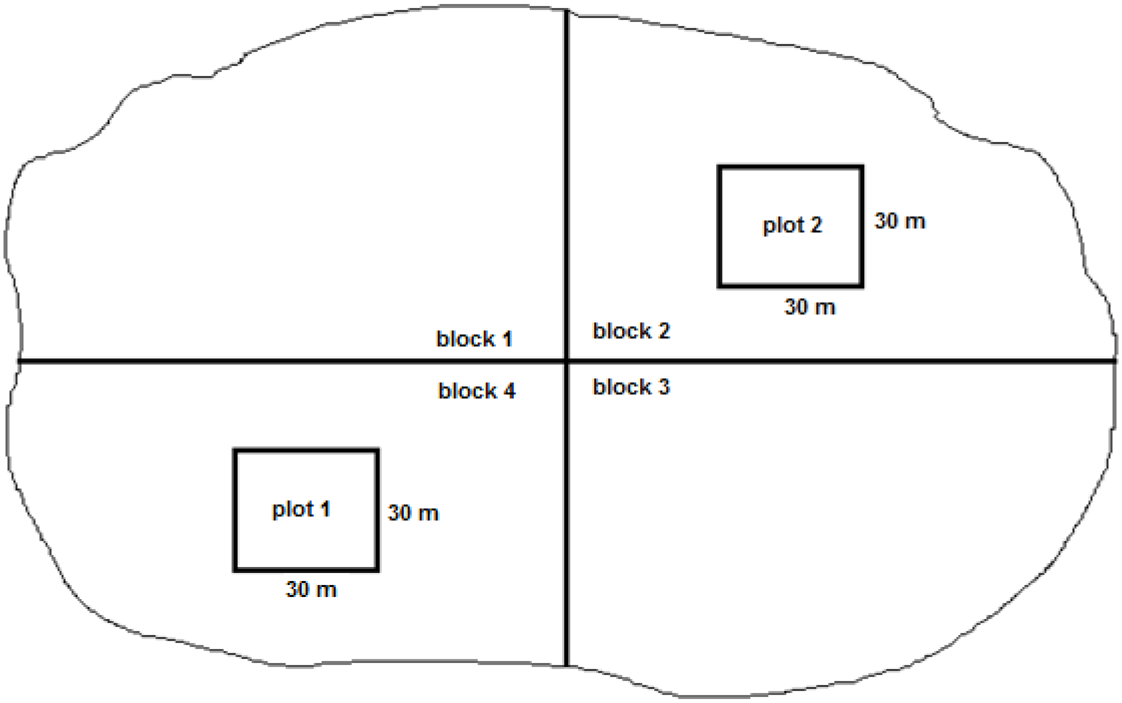


Figure 3.4. A layout of sampling plots in each fallow field. Fallows in the study area have a variable area, ranging from 0.2 ha to 1.9 ha with an average of 0.97 ha

3.2.4. Above-ground biomass

In the same plots used to analyze secondary succession, estimation of tree above-ground biomass within fallows was carried out on ten dominant species on each soil type. The Reference unit method after Snowdon *et al.* (2002) was used in estimating shrub biomass and allometric equations were used to estimate tree biomass.

3.2.4.1. Shrub biomass: Reference Unit Method

The first stage of shrub biomass estimation involved establishing three height classes (1) seedlings and saplings, woody plants with a height less than 0.5 m (< 0.50 m), (2) saplings and regenerating stumps, woody plants with a height between 0.51 and 1m (0.51–1 m), (3) shrubs, woody plants with a height between (1.1–3 m). In each size class for each of the ten species, three individual plants were sampled for calibration purposes. This involved counting the total number of primary shoots on each selected shrub. Three average size primary shoots (at least, 15-20% of the shrub) were clipped and weighed. The clipped material was kept within large envelopes for further analysis. In the laboratory, the clipped material was oven dried for 48 hours at 70˚C. This temperature level (70˚C) is adequate enough to dry the material but do not remove all moisture from plant tissues whereas higher temperatures (> 80°C) may cause a loss of dry weight due to decomposition of some organic compounds and volatilisation of some vegetative oils (Snowdon *et al*. 2002). The dried plant material was weighed. Mean dry weight was calculated for the three primary shoots. For each of the three sampled plants, dry weight (biomass) of the whole plant was calculated multiplying, the number of primary shoots per plant with dry weight of one primary shoot.

To obtain total biomass in each size class, for each species, the mean biomass of the three sampled plants was multiplied with the number of individual plants in each size class (Equation 1).

$$\boldsymbol{B}_{(\boldsymbol{species})} = \frac{\sum \boldsymbol{MB}_{(\boldsymbol{tree})} \times \boldsymbol{NP}_{(\boldsymbol{class})}}{\boldsymbol{SA}} \qquad \textbf{Equation (1)}$$

Where:

$\boldsymbol{B}_{(species)}$= total species biomass per plot (tons/ha)
$\boldsymbol{MB}_{(tree)}$ = mean tree biomass
$\boldsymbol{NP}_{(class)}$ = number of trees per plot
$\boldsymbol{SA}$ = total sampled area

3.2.4.2. Tree biomass calculation

Tree diameter at breast height 1.3 m from the soil level (DBH) was used to estimate tree biomass via allometric equations. Diameter at breast height gives a level of consistency and is easy to measure (Philip, 1994). All living woody plants with > 3 m height and > 5 cm DBH were measured and for trees forking below 1.3 m from the ground level, each stem was measured and recorded separately (Philip, 1994). Allometric equations that relate components of above-ground tree biomass to stem dimensions are commonly used to obtain estimates of biomass in forests, woodlands and savannahs (Nickless *et al.*, 2011). These equations show the relationship between tree oven dried biomass and tree diameter at breast height. According to Chidumayo (1997); Specht and West (2003) this relationship is established through destructive methods where trees are felled, divided into foliage, branches, stem wood and stem bark and weighed fresh. Sub-samples are then oven dried and weighed. In the present study, because of resource constraints, it was not possible to develop the equations *de novo* in order estimate tree biomass. Thus, allometric equations were obtained from several published sources (Table 3.2).

In this study, species with closely related physiognomic characteristics were grouped with species with known allometric equations (*C. mopane*, *A. tortilis*) forming the core species in each group. Each equation in Table 3.2 was used to estimate the biomass of each tree, after which the average of the estimates was calculated per tree. The total amount of biomass per hectare in each fallow was estimated by summing the biomass of individual trees in each sampling plot and then dividing by the area of each plot (0.09 ha) after which the average of the biomass estimates from the two sampling plots was calculated per fallow (Equation 2).

$$\boldsymbol{B}_{(\boldsymbol{fallow})} = \frac{\sum \boldsymbol{MB}_{(\boldsymbol{tree})}}{\boldsymbol{PA}} \qquad \textbf{Equation (2)}$$

Where:

$\boldsymbol{B}_{(fallow)}$ = total above-ground biomass per fallow (tonnes/ha)

$\boldsymbol{MB}_{(tree)}$ = mean biomass per tree (tonne)

$\boldsymbol{PA}$ = Plot area (ha)

Table 3.2. Predictive equations for tree biomass estimation

Equation	**Source**
Eq.1 $B = (D^{2.4575} * 0.0544)$	Guy, 1981
Eq.2 $B = 34.47 - 8.067D + 0.6589(D)^2$	Brown *et al*. 1989
Eq.3 $B = 16\ DBH - 157$	Chidumayo, 1997
Eq.4 $B = 2.23\ DBH - 6.44$ (DBH> 11 cm)	Chidumayo, 1997
Eq.5 $B = 1.4DBH - 3.1$ (DBH <11 cm)	Chidumayo, 1997
Eq.7 $B = 3.6225 * DBH \hat{} 1.4924$	REDD (2012)
Eq.6 $B = -3.04 + 1.287 * (DBH)^2$	Scholes, 1990

Where:
B = estimated above-ground biomass (kg)
DBH = diameter (cm) at breast height (1.3 m)

3.3. Data analysis

To determine tree species alpha diversity (diversity within plots), beta diversity (diversity within land-cover types) and gamma (diversity at each site), the Simpson's reciprocal and Shannon-Wiener index (Magurran, 1988) were computed for each plot, land-cover type and site. Data were tested for normality using the Kolmogorov-Smirnov test. Since the data were normally distributed, Two-Way ANOVA was used to analyze the data with tree density, Simpson's and Shannon-Wiener indices as response variables, while soil type and land-use as the two fixed factors, soil type with two levels (*mutapo* versus *bandate*) and land-use with three levels (cultivated versus fallow versus woodland). Comparison between individual sites was carried out using Tukey multiple comparisons of means at $P < 0.05$ confidence level. To explore the relationship between field age and number of woody species and diversity regression analysis was used with field age (years) as the explanatory variable. Regression analysis was also used to establish the relationship between fallow age and number of woody species, woody species diversity, tree diameter at breast height and accumulated above-ground biomass.

To determine arthropod (Charaxes and Cetoniinae) species alpha diversity, data (number of species and number of individuals per species) from each trapping location (20 samples per site) were used to compute the Shannon-Weiner diversity index (Magurran, 1988) and Morisita-Horn similarity index (M-H) (Wolda, 1981). Among quantitative similarity indices, the Morisita-Horn index is the only one not strongly influenced by species richness and sample size. The M-H index is, however, sensitive to the most abundant species (Wolda, 1981). Next in this analysis, the Kolmogorov-Smirnov test was used to test for normality of data and it showed that data followed a normal distribution. The data was then analyzed using Two-Way ANOVA with average abundance and Shannon-Wiener index obtained in each trap (i.e. sample size = 80) as the response variables, while habitat and distance as the two fixed factors, each factor with two levels (Woodland versus Cultivated and Interior versus Edge transect respectively). All diversity indices were computed using EstimateS statistical software: version 8 (Colwell, 2006) and all statistical analyses were performed using R Version 2.14.0 and SPSS version 16.0.

RESULTS AND DISCUSSION

CHAPTER 4: Tree diversity in agricultural landscapes in the mid-Zambezi valley

4.1. Results

4.1.1. Tree diversity as a function of soil type and land-use

A total of 82 woody species from 29 families were recorded across the study area (Appendix 1). Eight species (9 %) were present in all four sites. There were more species present on *bandate* soil, 53 species (63 %) than on *mutapo* soil, 43 species (52.4 %). Among sites, the highest number of species was recorded in Site 3 (43 species), followed by Site 4 (42 species), Site 1 (39 species) and lowest in Site 2 (38 species) (Figure 4.1). Woody species diversity was significantly higher ($F = 3.13_{2,36}$, $p = 0.021$) on *bandate* soil than *mutapo* soil. Table 4.1 shows mean values for tree density and the two diversity indices (Simpson's and Shannon-Weiner) in each category and Table 4.2 shows results on Two-Way ANOVA test showing significant effect of soil type and land-use and significant interaction ($p < 0.05$). Tukey's pairwise comparison tests between land-use zones on *bandate* soil, revealed that, woodland areas had significantly higher ($p < 0.05$) woody species diversity while on *mutapo* soil, woody species diversity was significantly higher ($p < 0.05$) within fallow areas. Results on the Morisita-Horn sample similarity index are presented in Table 4.3.

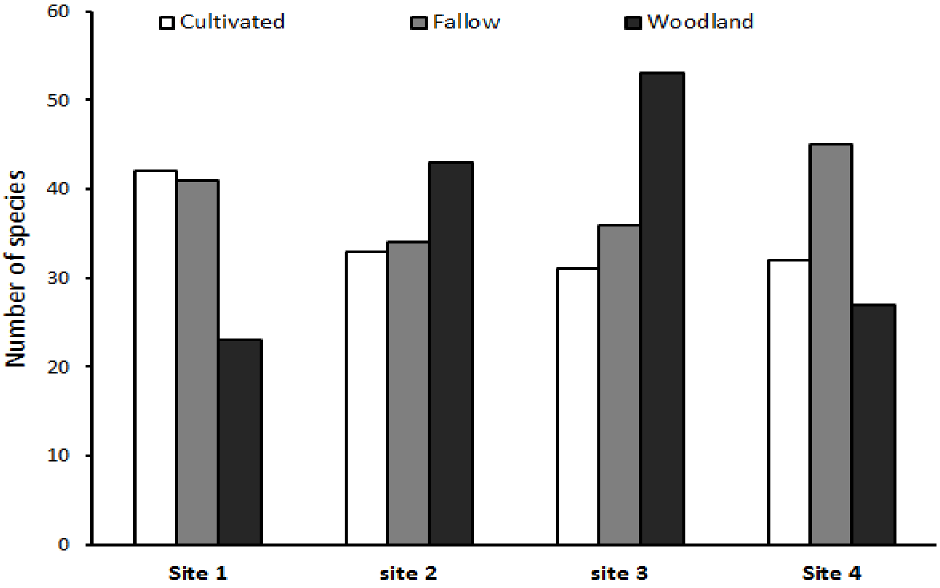


Figure 4.1. Number of woody species within each land-use zone in each site.

Table 4.1. Woody species diversity (Simpson's index and Shannon-Weiner index) and density among land-use zones in each site

	Land-use	No. of species	Simpson's index	Shannon-Weiner index	Tree density (n/ha)
Site 1	C		6.48	1.89	707.32
	F		7.88	2.86	1138.10
	W		3.75	1.34	1721.63
Site 2	C		12.05	3.16	616.24
	F		16.22	4.06	1479.48
	W		14.03	3.85	1701.28
Site 3	C		4.57	2.15	450.46
	F		16.20	4.62	946.55
	W		12.57	2.41	1483.20
Site 4	C		3.05	1.31	362.71
	F		12.90	2.71	1072.78
	W		1.80	1.04	2024.18

C = cultivated; F = fallow; W = woodland

Table 4.2. Two-Way ANOVA test on the effect of soil type (*mutapo* versus *bandate*) and land-use (cultivated versus fallow versus woodland) on woody species diversity.

Measure	**Df**	**F-value**	**p-value**
Shannon-Wiener index			
Soil type	1	10.60	0.00 **
Land-use	2	27.08	0.00 ***
Soil type: Land-use	4	7.24	0.00**
Simpson's index			
Soil type	1	7.59	0.00 **
Land-use	2	23.74	0.00***
Soil type: Land-use	4	3.59	0.03*

The asterisks * represents a statistically significant variable effect (* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$)

Table 4.3. Shared woody species and similarity statistics for the four sites in the study area

First Sample	Second Sample	Shared Species Observed	Morisita-Horn
Site 1	Site 2	11	0.51
Site 1	Site 3	15	0.55
Site 1	Site 4	25	0.81
Site 2	Site 3	28	0.88
Site 2	Site 4	9	0.47
Site 3	Site 4	16	0.59

Morisita-Horn sample similarity index calculated using EstimateS (Colwell, 2006)

4.1.2. Tree diversity as a function of cultivation intensity

A comparison of sites on the same soil type, showed that, woody species diversity and tree density were higher in less intensified sites, in Site 1 than site 4 (Figure 4.2) and Site 2 than Site 3 (Figure 4.3). The Tukey multiple comparison tests, giving more attention to sites with the same soil type (Table 4.4), showed significantly higher ($p < 0.05$) woody tree density in Site 1 compared to Site 4 and higher woody tree density in Site 2 than Site 3. The same test showed no significant differences ($p > 0.05$) in woody species diversity between Site 1 and Site 4 and between Site 2 and Site 3.

Table 4.4. Comparison of tree density and diversity between cultivated areas based on Tukey HSD multiple comparisons of means at 95 % confidence level

Parameter	**Site comparisons**	**p - value**
Tree density (n/ha)	Site 2-Site 1	0.99
	Site 3-Site 1	0.00*
	Site 4-Site 1	**0.00***
	Site 3-Site 2	**0.01***
	Site 4-Site 2	0.00*
	Site 4-Site 3	0.82
Shannon-Wiener index	Site 2-Site 1	0.00*
	Site 3-Site 1	0.01*
	Site 4-Site 1	**0.08**
	Site 3-Site 2	**0.07**
	Site 4-Site 2	0.01*
	Site 4-Site 3	0.00*
Simpson's index	Site 2-Site 1	0.00*
	Site 3-Site 1	**0.07**
	Site 4-Site 1	**0.11**
	Site 3-Site 2	0.41
	Site 4-Site 2	0.04*
	Site 4-Site 3	0.01*

The asterisks * represents a statistically significant difference (* $p < 0.05$); Pairs in bold represent sites with the same soil type

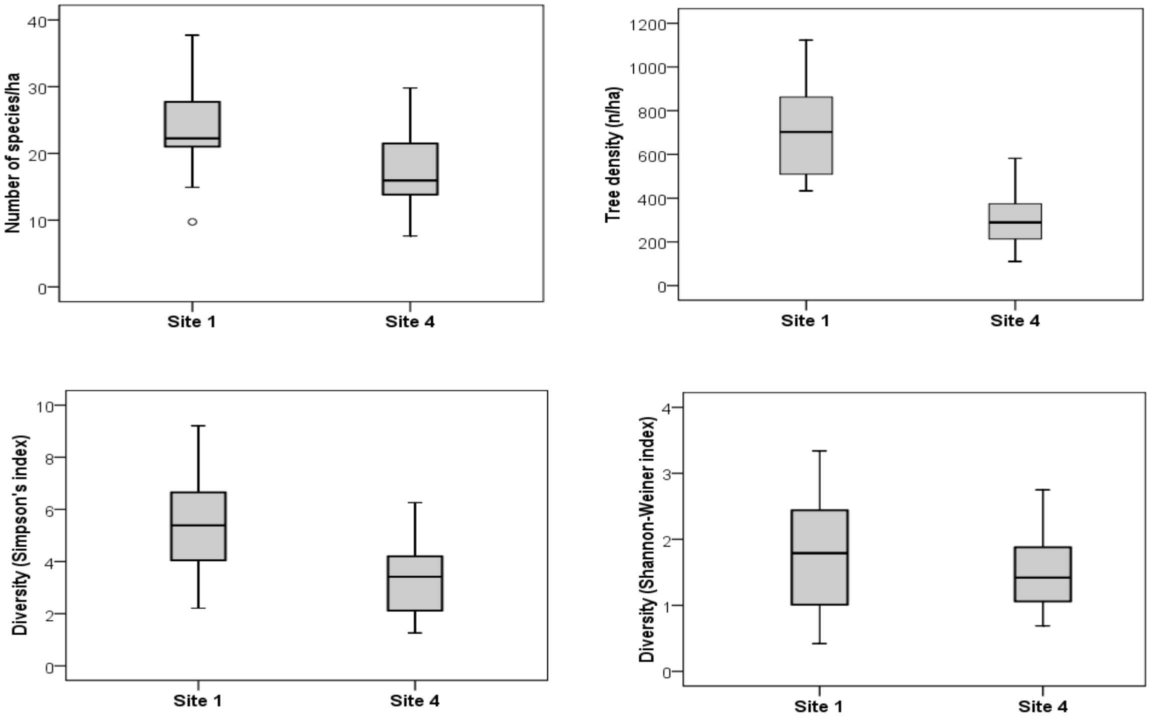


Figure 4.2. A comparison of tree diversity measures between cultivated areas in Site 1 and Site 4

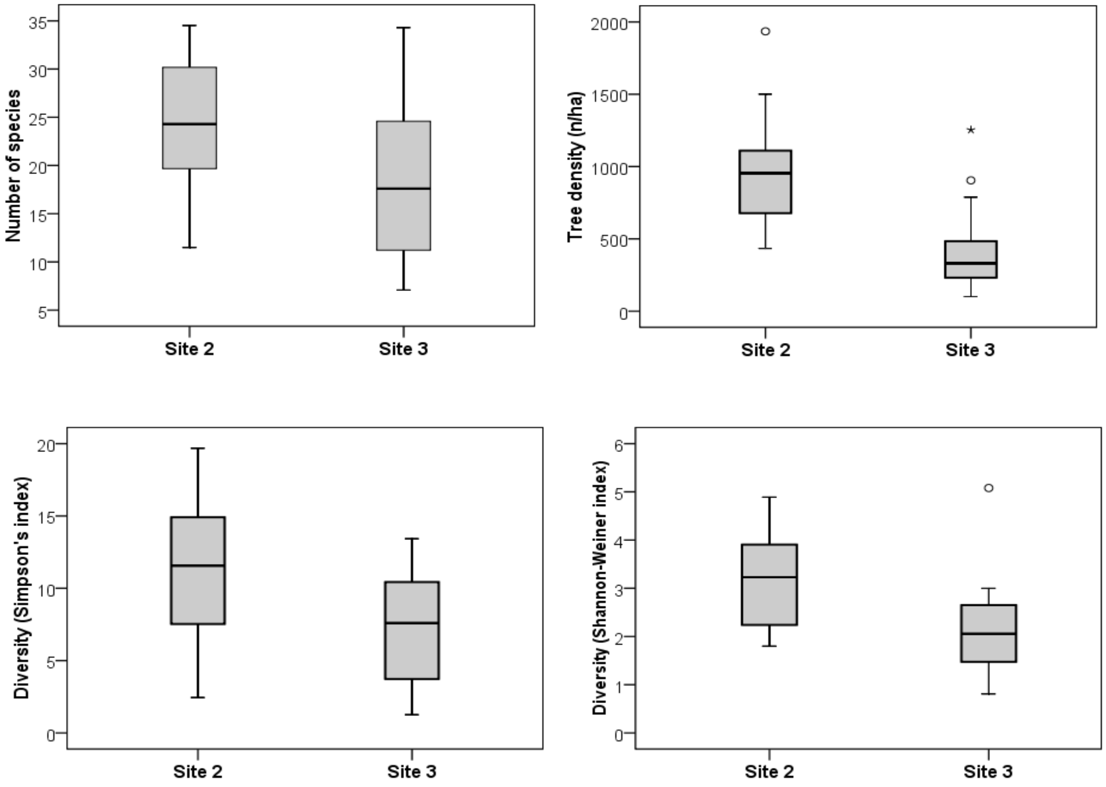


Figure 4.3. A comparison of tree diversity measures between cultivated areas in Site 2 and Site 3

4.1.3. The relationship between field age and woody species diversity

In all sites, woody species richness, tree density and species diversity decreased with increasing cultivation period (Figure 4.4 to Figure 4.7). Both tree density and species diversity were high within newly opened fields, then decreased with increasing field age. The decreasing trend was characterised by a steep decrease in the first 10 year years after which there was no decrease. Non-linear regression analysis, showed significant ($p < 0.05$) relationships between field age and all tree diversity measures. The relationship between field age and woody species diversity was an exponential decrease in diversity with increasing age. Low R^2 values ($R^2 < 0.5$ in all cases) indicate a poor fit of the exponential trend, except for number of species in Site 2, were $R^2 = 0.61$ (Figure 4.5a), Site 4, $R^2 = 0.65$ (Figure 4.7a). The Shannon-Weiner index in Site 4 followed a linear decrease with increasing field age (R^2 value of 0.65).

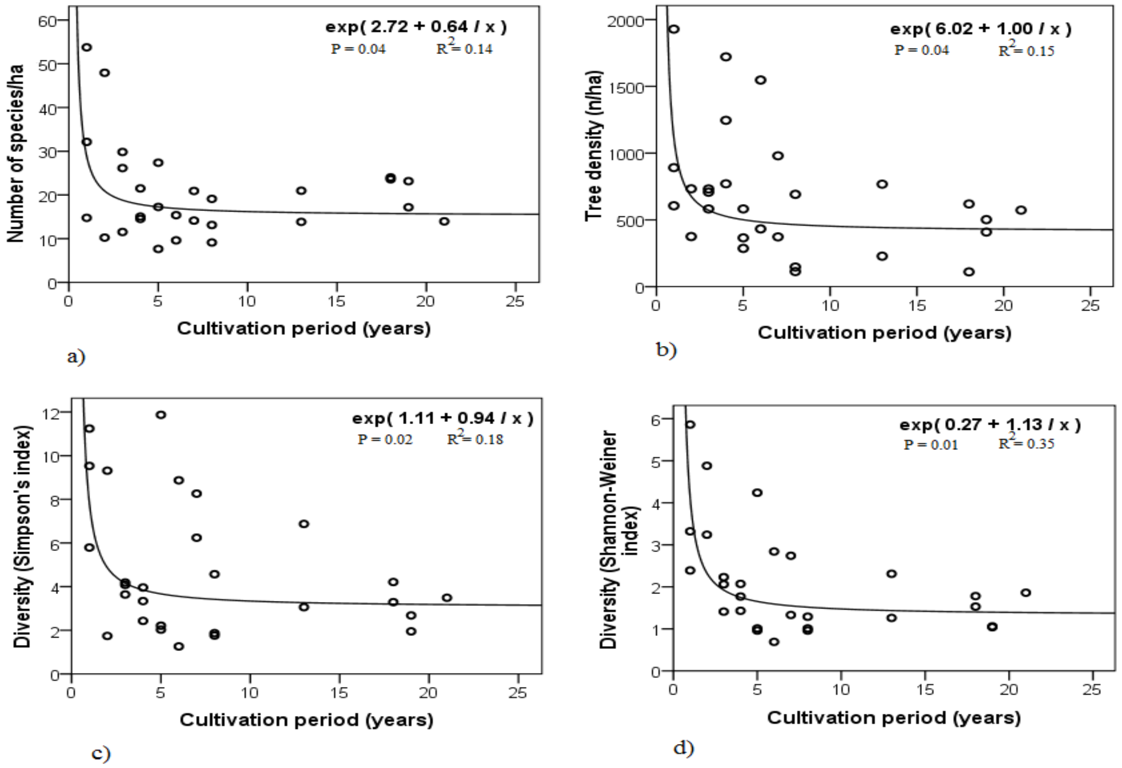


Figure 4.4. Relationship between woody species diversity and field age at site 1

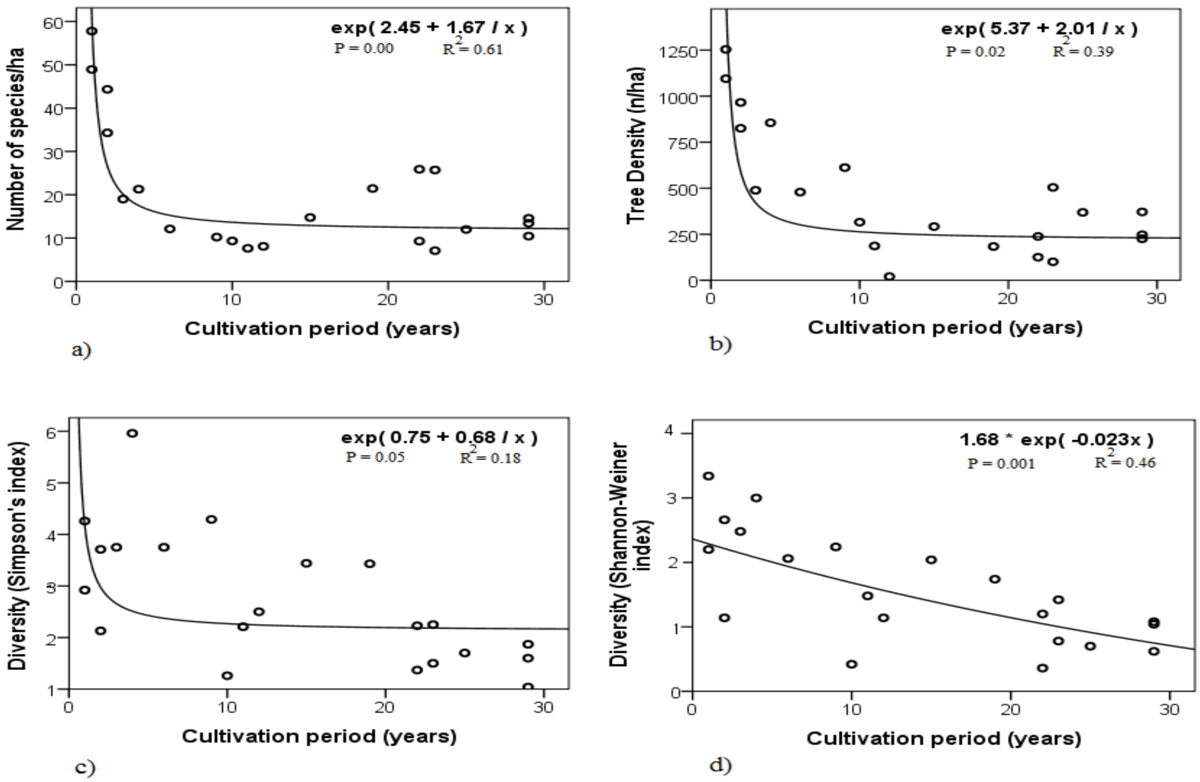


Figure 4.5. Relationship between woody species diversity and field age at site 2

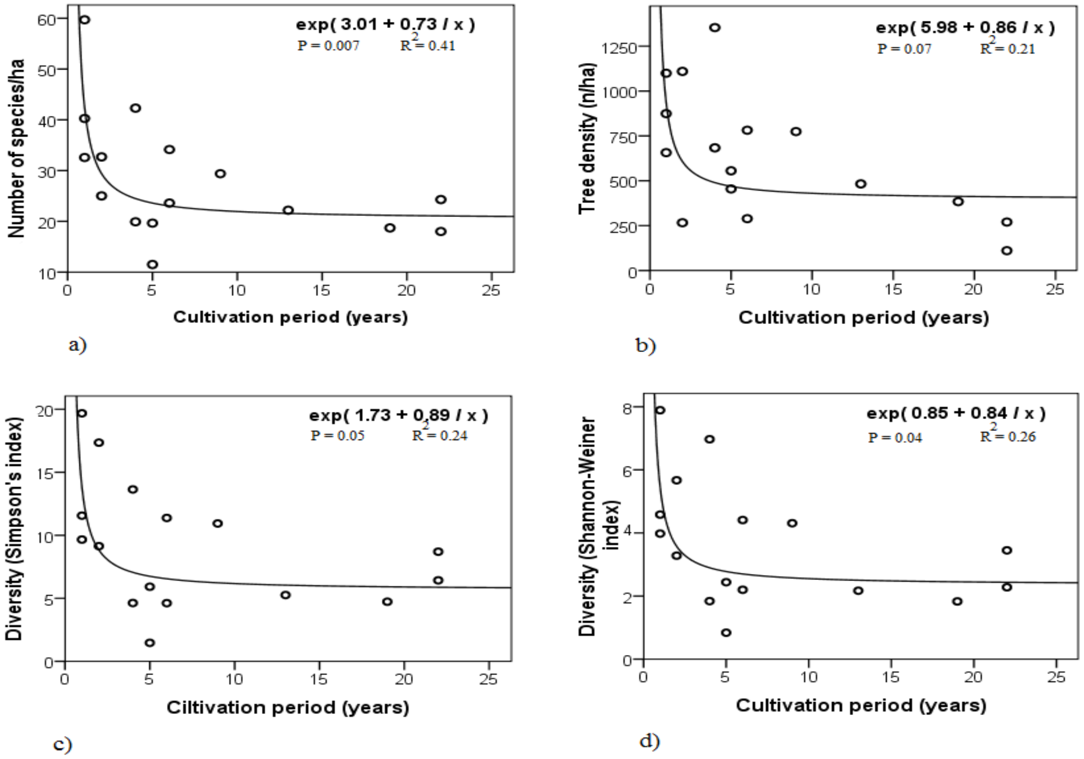
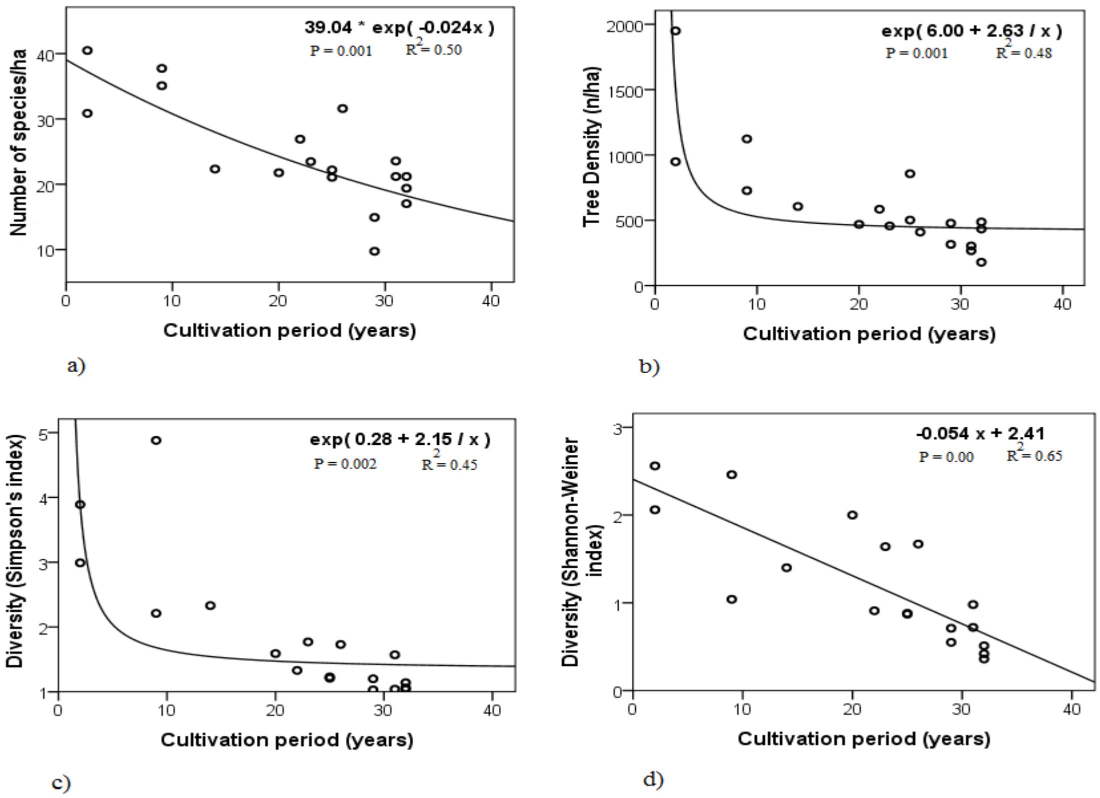


Figure 4.6. Relationship between woody species diversity and field age at site 3



a) b)

c) d)

Figure 4.7. Relationship between woody species diversity and field age at site 4

4.1.4. Impacts of woodland clearing and cultivation on woody species dominance

Table 4.5 shows sixteen most frequent woody species in the study area. High frequencies of shrubs of the ten species were recorded within cultivated fields and fallows while more trees were recorded within natural woodlands and also within fallows. Woody species with the highest frequency changed among the three land-cover types at each site. Among the sets of tolerated species within fields and field edges in the agricultural areas, fruit trees were observed in high abundance than other tree species. In total eight species of fruit trees were recorded compared to six species of medicinal importance (Table 4.6). Figure 4.8 to Figure 4.15 show frequencies of ten species measured in each land-cover category at each site. The common trend was high frequency of *C. mopane* trees in natural woodlands on *mutapo* soil and *D. kirkii* and *X. torreana* trees in natural woodland on *bandati* soil. In cultivated areas and fallows *A. tortilis subsp.*

spirocarpa, *C. elaeagnoides* and *L. cappassa* recorded high frequencies of both shrubs and trees (Figure 4.8 to Figure 4.15).

Table 4.5. Dominant woody species recorded in the study area

Species	**Site 1**	**Site 2**	**Site 3**	**Site 4**
Acacia tortilis	+	+	+	+
Albizia versicola	+	-	-	+
Azanza garckeana	-	+	-	-
Colophospermum mopane	+	+	+	+
Combretum elaeagnoides	+	+	+	+
Combretum mossambicense	+	-	+	+
Dichrostachys cinerea	+	+	+	+
Diospyros kirkii	-	+	+	-
Diospyros qualoensis	+	-	-	+
Grewia monticola	+	+	-	+
Lonchocarpus cappassa	-	-	+	-
Markhamia zanzibarica	+	+	+	+
Sclerocarya birrea	-	+	-	-
Terminalia sericea	-	-	+	-
Xeroderris stulhmannii	-	+	-	-
Ziziphus mauritiana	+	-	+	+

The positive sign (+) means present and negative sign means (-) absent

Table 4.6. Woody species tolerated within agricultural landscapes in mid-Zambezi valley

Species	**Family**	**Attributes**
Azanza garckeana	Malvaceae	sweet edible fruits
Adansonia digitata	Bombacaceae	edible fruits and leaves, bark yields excellent fibre, seed edible and germinates fairly easily, used in traditional medicinal
Acacia nilotica	Mimosaceae	it is used for fuel wood, charcoal, fodder and construction, edible gum.
A. tortilis subsp. *spirocarpa*	Mimosaceae	useful for timber, fuel wood, charcoal, fodder, construction, N fixation and soil stabilization, bark used medicinally.
Annona senegalensis	Annonaceae	it has medicinal properties and edible fruit
Cassia abreviata Oliv.	Caesalpiniaceae	roots used medicinally (to treat black water)
Diospyros kirkii	Ebenaceae	an edible berry fruit much sought after,
Faidherbia albida	Mimosaceae	typical agroforestry African species important for fodder, construction, N fixation and soil stabilization, bark used medicine.
Kigelia africana	Bignoniaceae	easily cultivated, high carbon fixing ability and a very large toxic fruit with high medicinal properties (skin disorders)
Pseudolachnostylis maprouneifolia	Phyllanthaceae	fire resistant, respond well to cultivation, good honey tree, bark and leaves used medicinally
Pterocarpus rotundifolius	Fabaceae	good carbon fixer given
Lonchocarpus capassa	Fabaceae	heavy, hard, durable wood, leaves browsed by stock, roots and leaves used medicinally
Sclerocarya birrea	Anacardiaceae	edible fruit, bark used for medicinal purposes, (proven antihistamine and anti-diarrhoea properties), useful wood.
Tamarindus indica	Caesalpiniaceae	edible acid sweet fruit used for preserves, flowers, leaves and crushed seed added to relishes
Vangueria infausta	Rubiaceae	edible fruits
Ziziphus mauritiana	Rhamnaceae	edible fruits

Sources for attributes: Coates-Palgrave (1995); van Wyk and van Wyk (1997) and Biodiversity Project (2002)

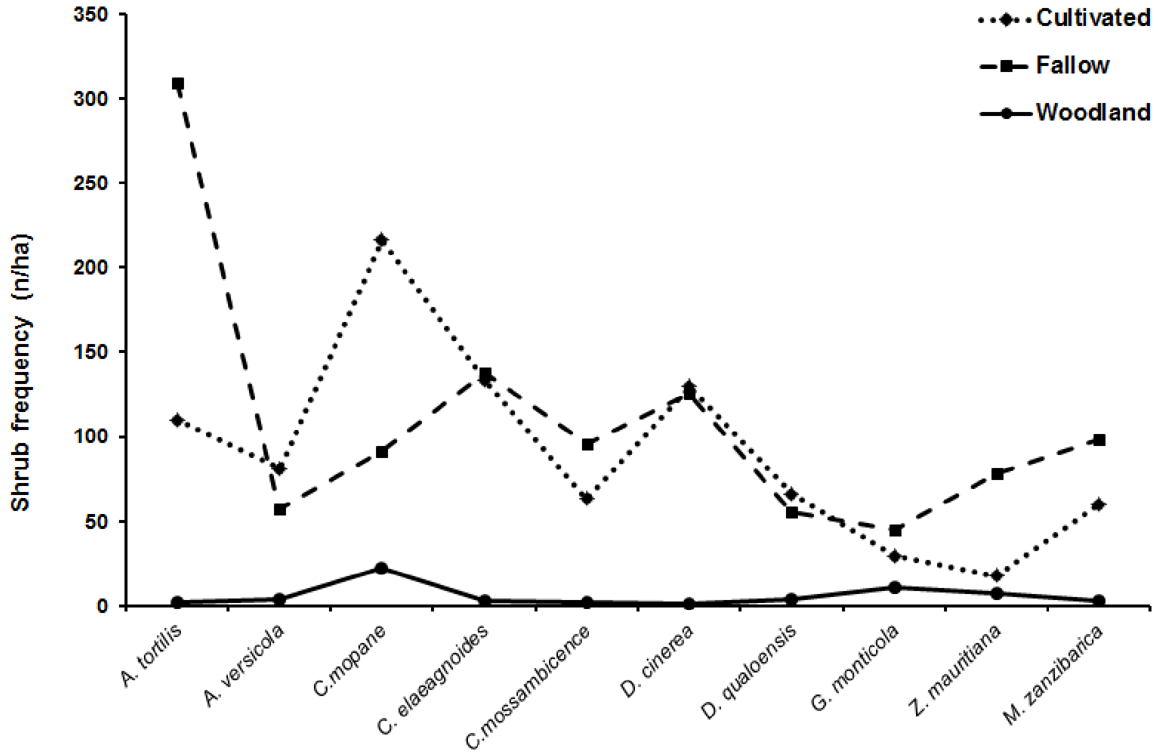


Figure 4.8. Frequency of shrubs of ten woody species in three land-cover types at site 1

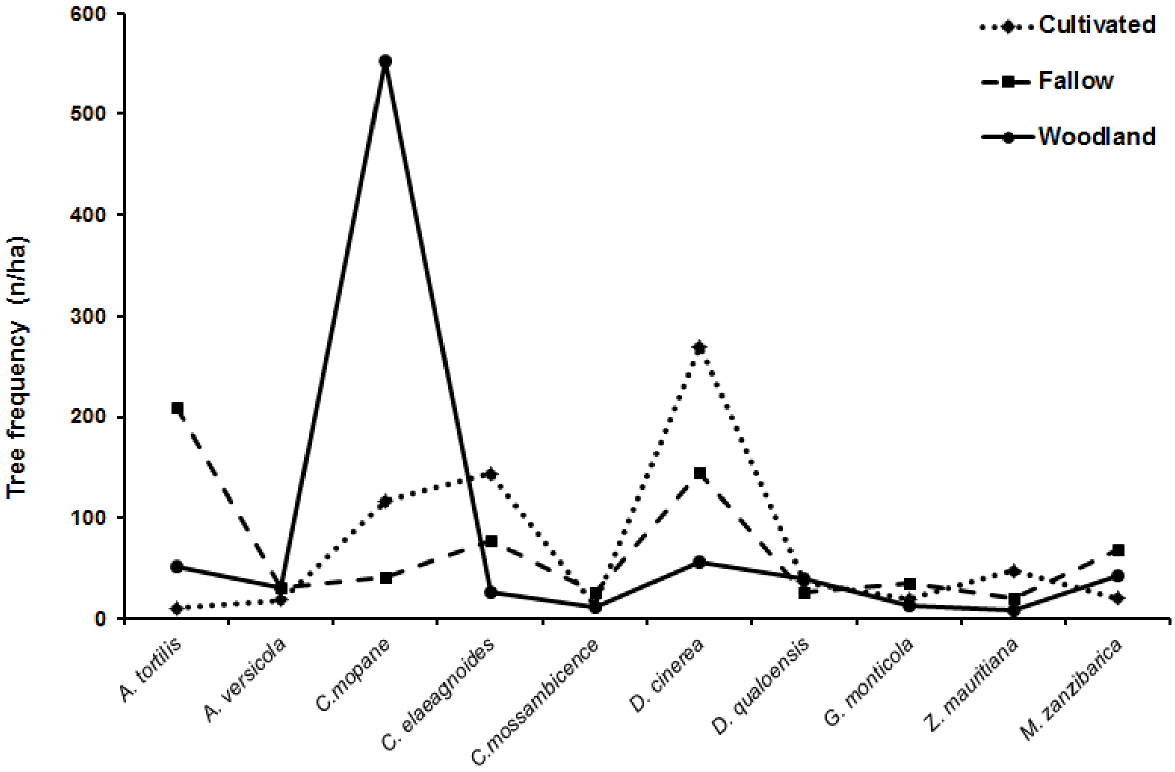


Figure 4.9. Frequency of individual trees of ten species in three land-cover types at site 1

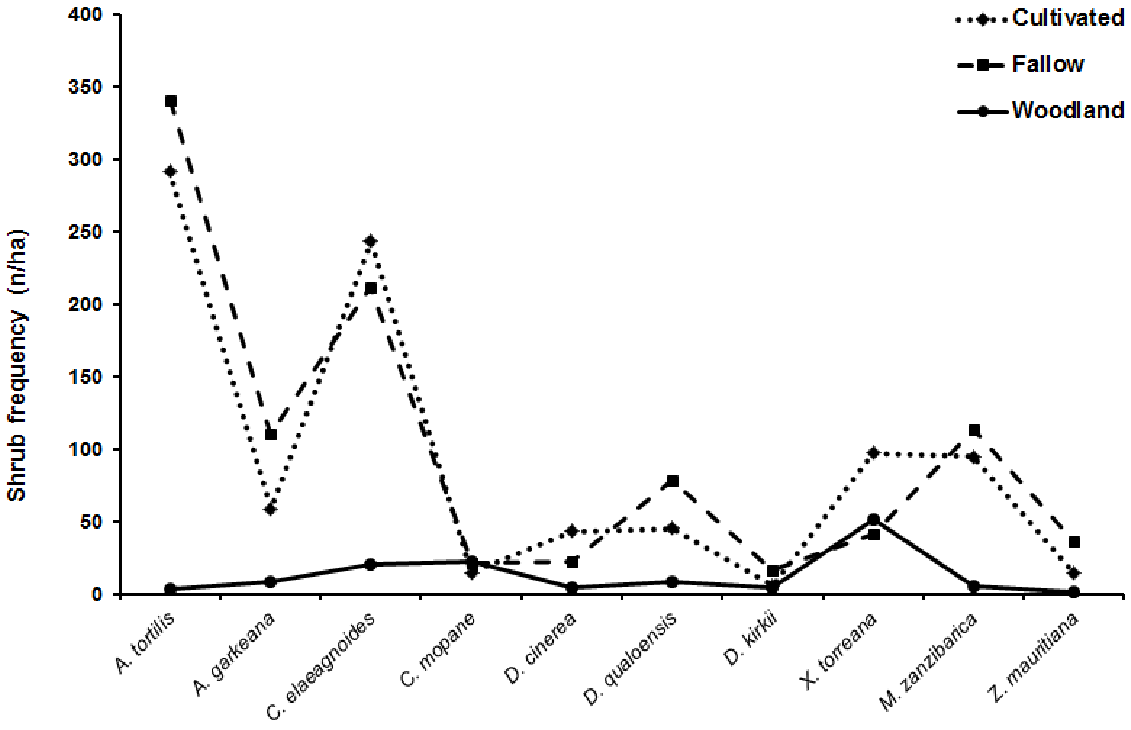


Figure 4.10. Frequency of shrubs of ten woody species in three land-cover types at site 2

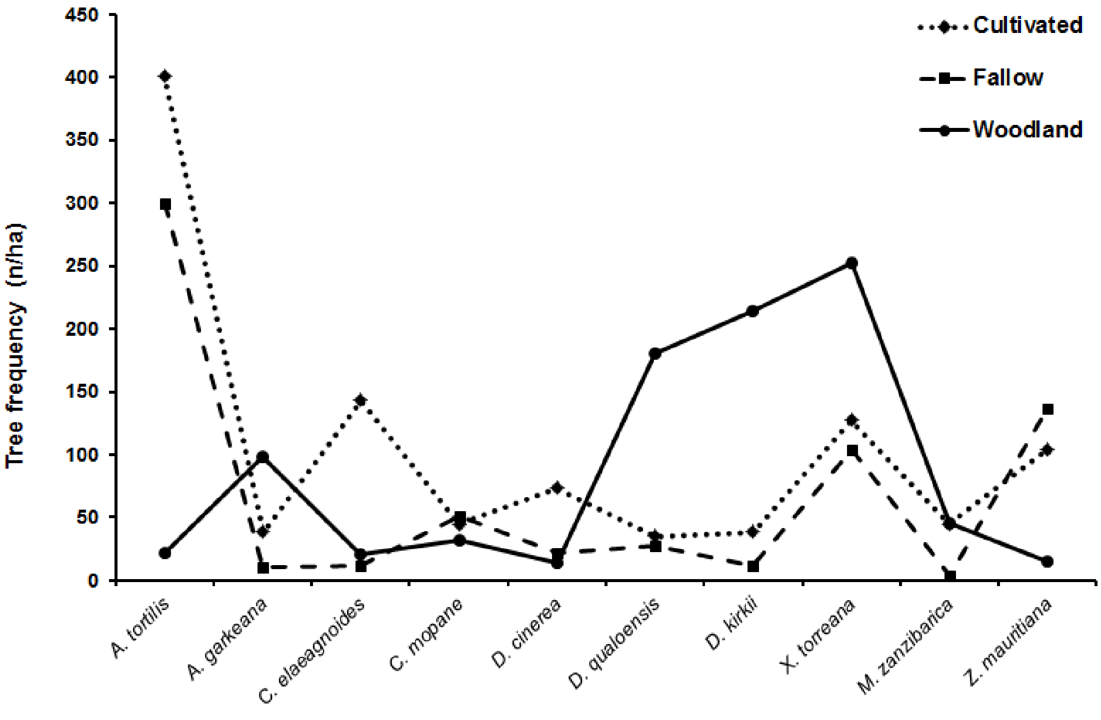


Figure 4.11. Frequency of individual trees of ten species in three land-cover types at site 2

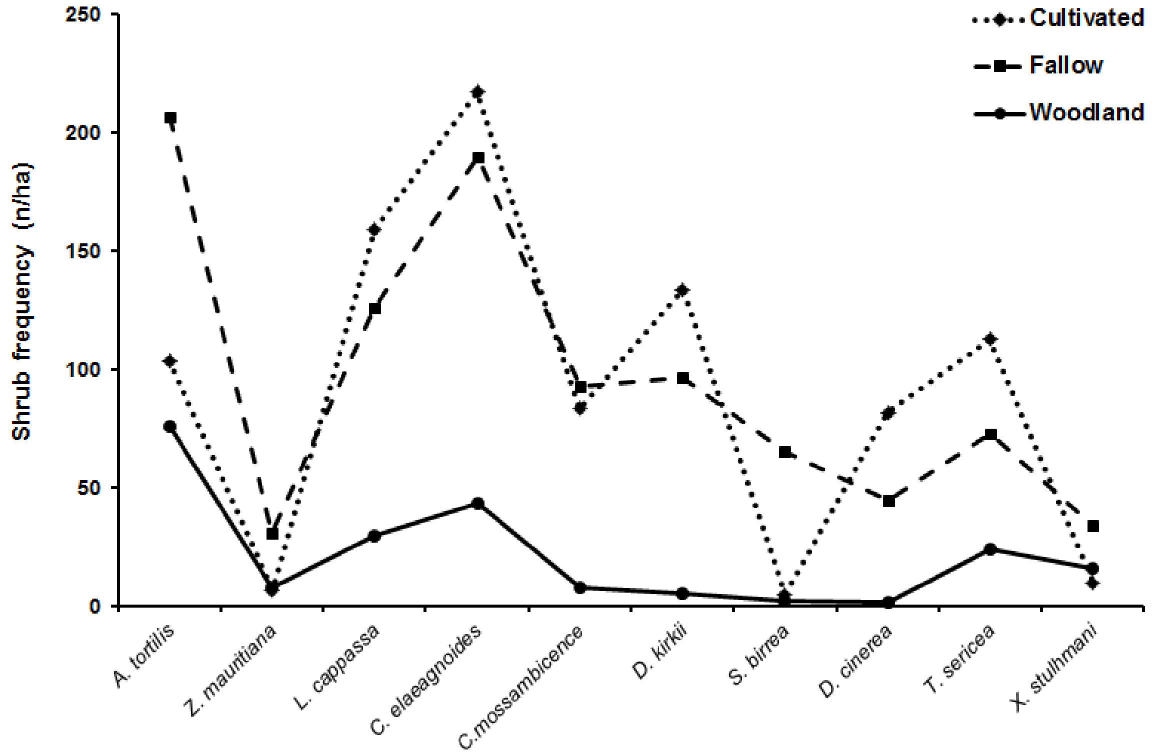


Figure 4.12. Frequency of shrubs of ten species in three land-cover types at site 3

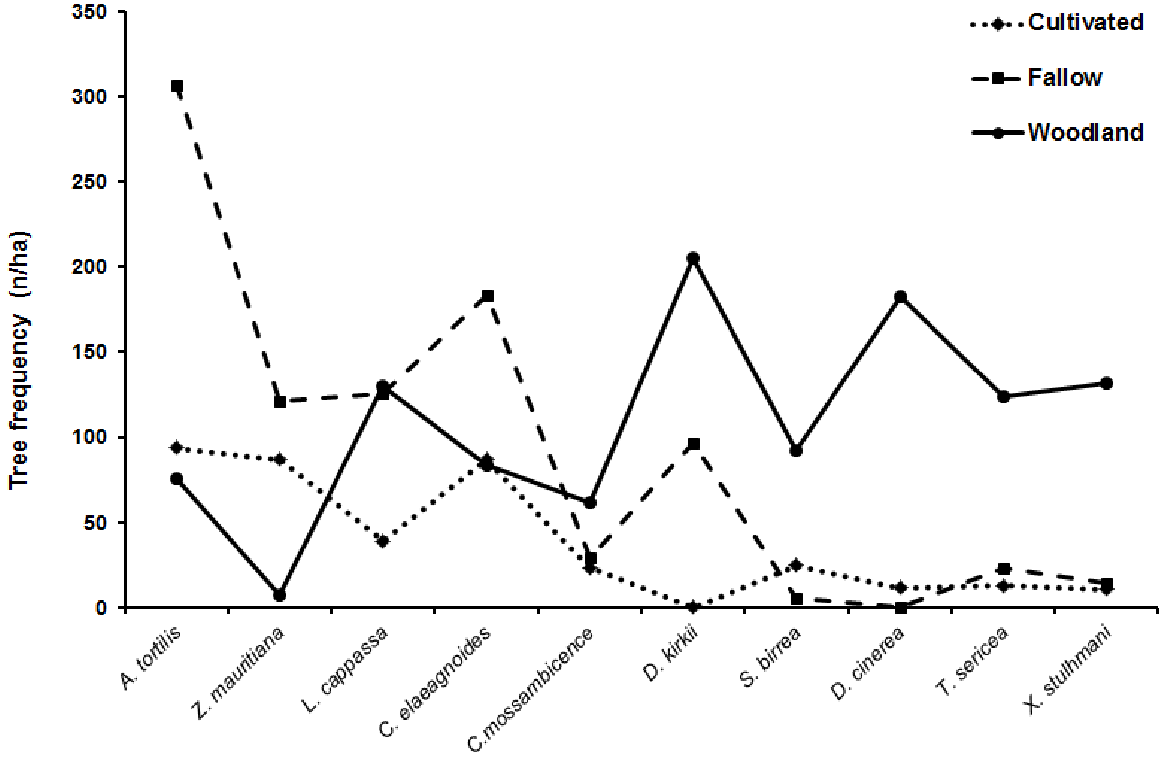


Figure 4.13. Frequency of individual trees of ten species in three land-cover types at site 3

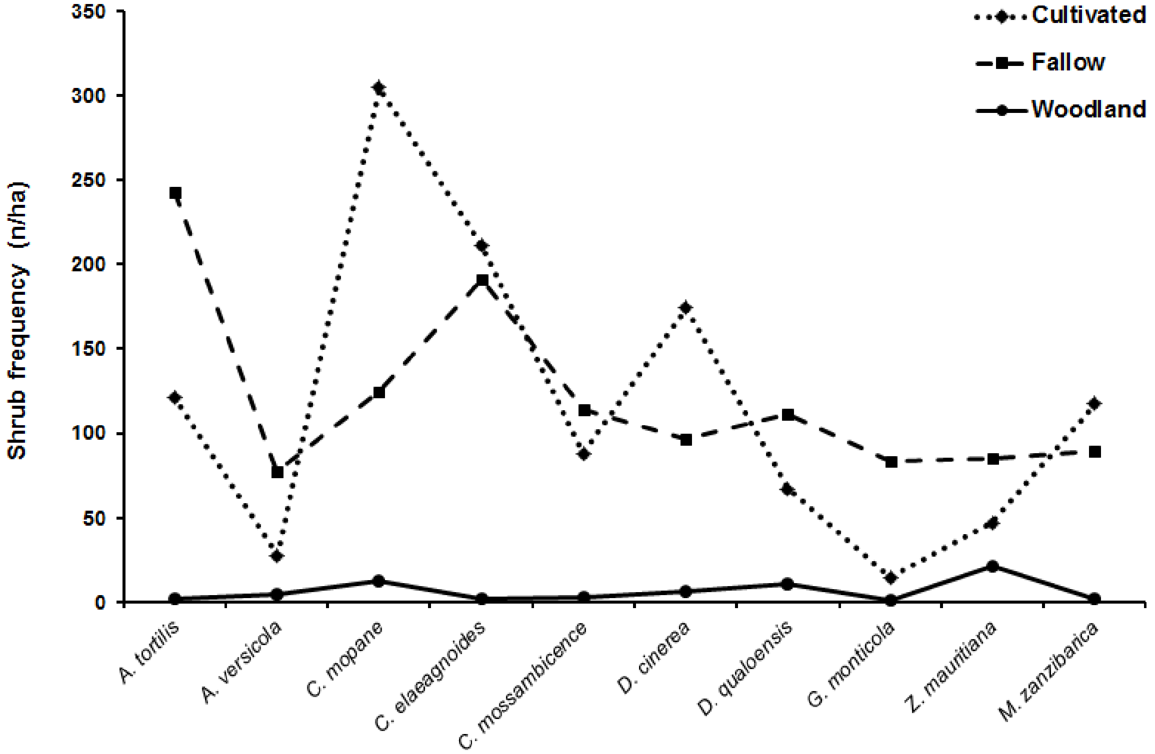


Figure 4.14. Frequency of shrubs of ten species in three land-cover types at site 4

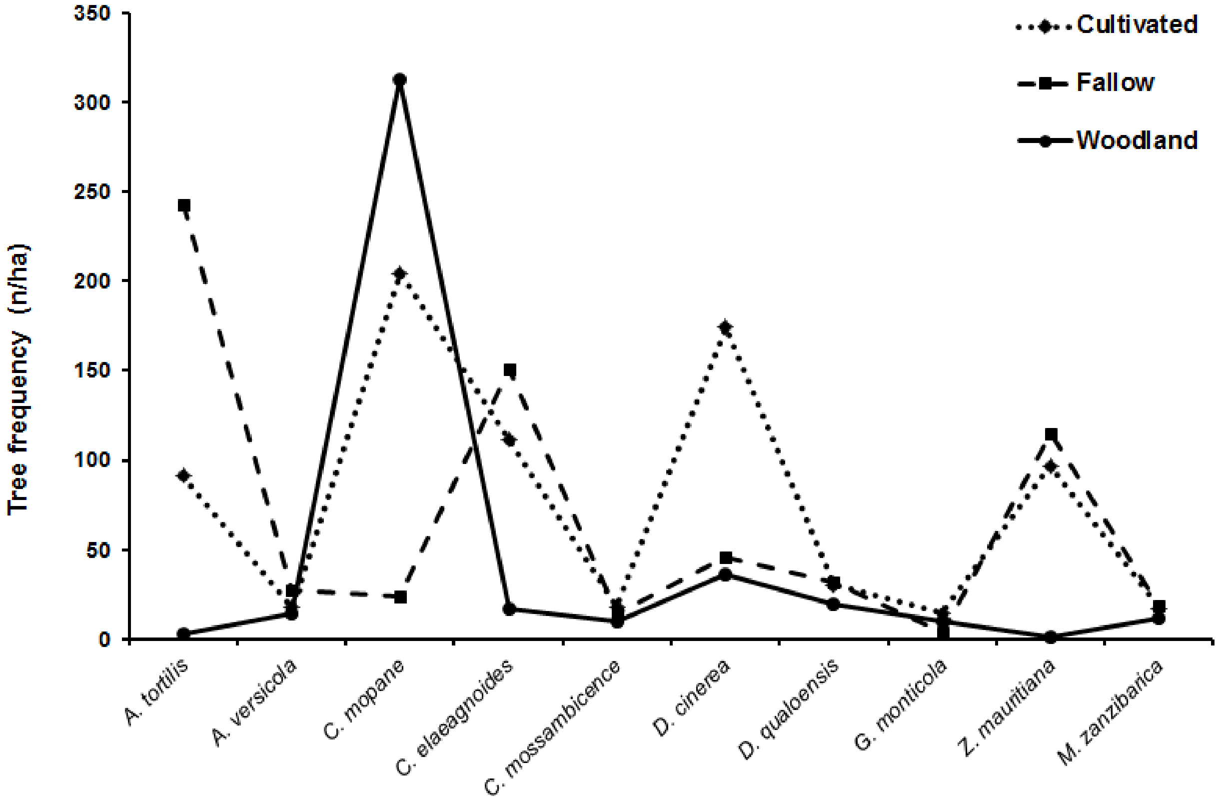


Figure 4.15. Frequency of individual trees of ten species in three land-cover types at site 4

4.1.5. Tree physiognomy

The lowest height class (< 0.50 m) recorded the highest frequency in cultivated areas in all sites, and frequency decreased with increasing size class (Figure 4.16 to 4.19). This is largely due to farmer’s practices of vegetation clearing and burning prior to planting each season. There was a distinct change in vegetation physiognomy from cultivated areas to fallow areas. In the fallow land, maximum frequency was recorded in the > 3 m height class. High frequency in this class (> 3 m) was more pronounced in Site 1 (Figure 4.16). The least frequency in the fallow zone was recorded in the 1.1–3 m size class. In the woodland category, a very high frequency was recorded for the > 3 m height class and very low frequencies were recorded for each of the remaining size classes (Figure 4.16 to 4.19). For size classes < 0.50 m, 0.51–1 m and > 3 m, there were significant differences ($p < 0.05$ for all classes) in frequency among the land use categories in all sites. In contrast, size class 1.1–3 m ($p > 0.05$) showed no significant differences in frequency among land use categories.

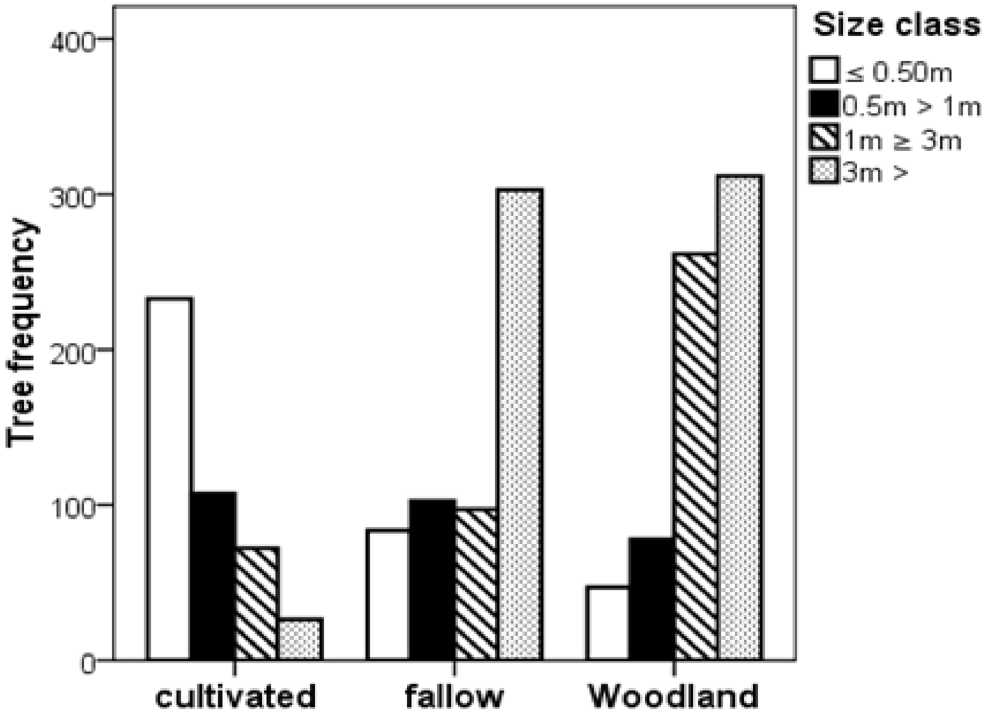


Figure 4.16. Woody tree frequency within height classes by land-use category at Site 1

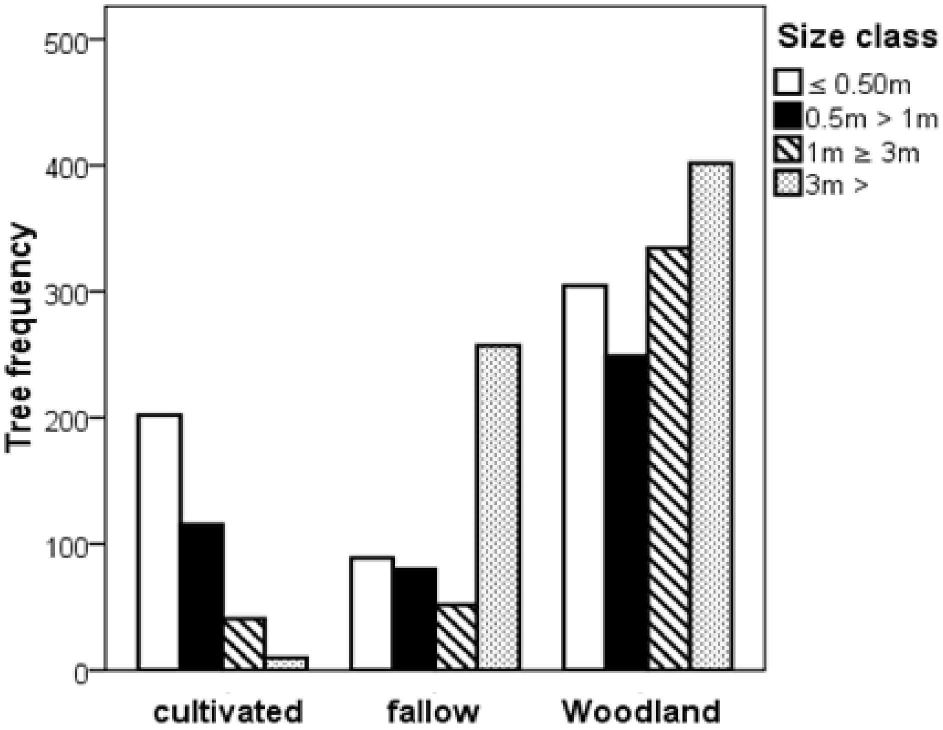


Figure 4.17. Woody tree frequency within height classes by land-use category at Site 2

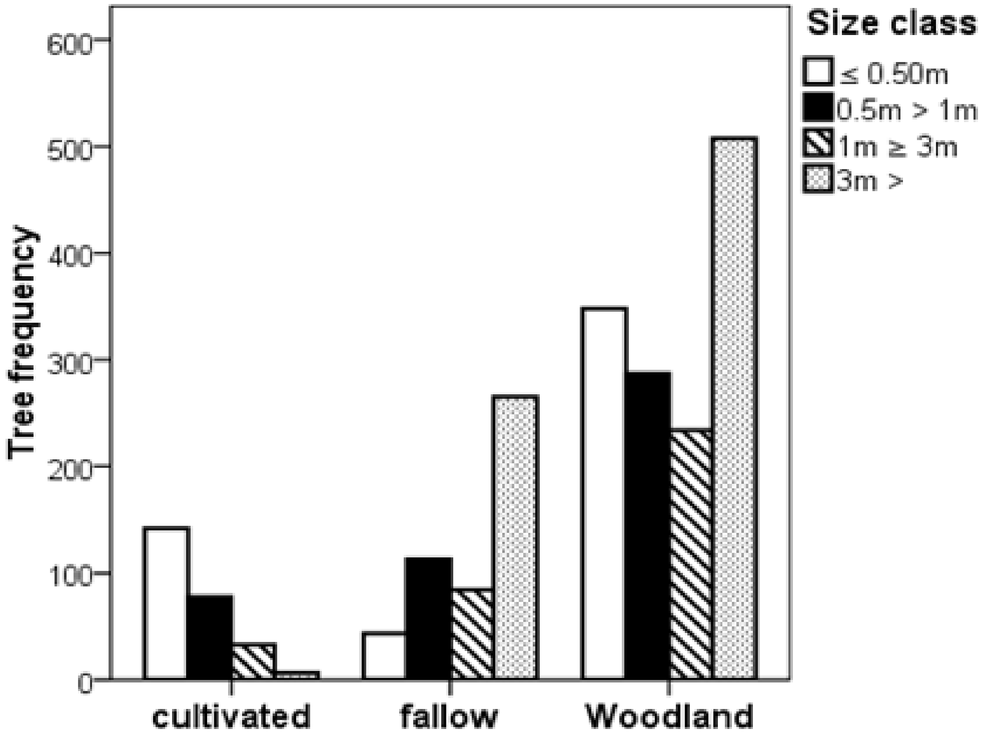


Figure 4.18. Woody tree frequency within height classes by land-use category at Site 3

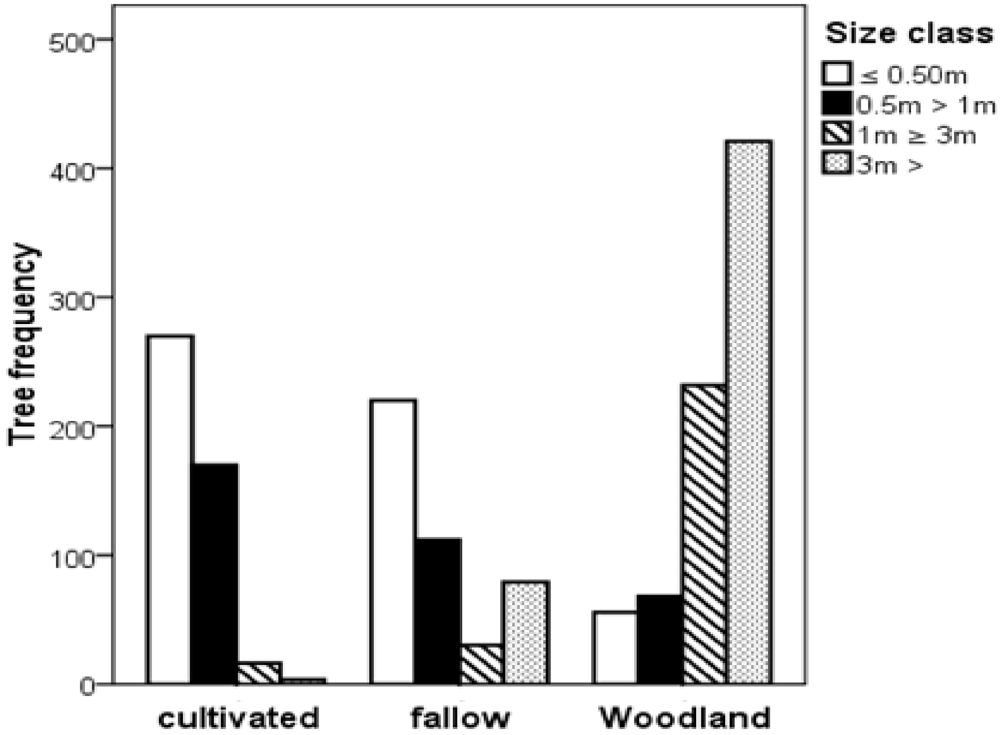


Figure 4.19. Woody tree frequency within height classes by land-use category at Site 4

4.2. Discussion

The study was designed to explain the effects of three underlying factors (soil type, cultivation intensity, and land-use) on aspects of woody species diversity. Results indicate that soil type significantly explains differences in species diversity in the landscape. The two dominant soils in the study area (*mutapo and bandate*) are known to have different physicochemical properties. *Mutapo* soils are eutrophic, grey-white clays which are associated with sodic/saline conditions (Baudron *et al.*, 2009). Fertility rating in these clayey interfluvial soils is very low. On the other hand, *bandate* soils are moderately heavy sandy clay soils. Compared to the clayey interfluvial soils these alluvial soils have a high moisture retention capacity (Mvuriye *et al.*, 2001). Each soil type is associated with a set of woody species that have managed to adapt to the chemical and physical properties of the soil. These results are in line with observations by Timberlake *et al.* (1993); Biodiversity Project (2002) and Fritz (2003). These studies highlighted that vegetation composition and structure in the mid-Zambezi valley vary with the type of soils.

Results showed high woody species diversity on *bandate* soil than on *mutapo* soil. Keeping in mind the description of the two soil types highlighted above, *mutapo* soil supports a floristically distinct vegetation type that is predominantly composed of *C. mopane* which form mopane woodlands (Mapaure, 1994). The unique characteristic of mopane woodlands is their low number of associated species, low alpha diversity (Timberlake *et al.*, 1993; Poilecot and Gaidet, 2010). Alluvial soils, showed high diversity of woody species, chemical and physical conditions in these soils are not restrictive to tree growth therefore many tree species grow well in these soils. Thus, this study has demonstrated that soil type is an important aspect in determining woody species diversity in the mid-Zambezi valley.

Results indicate that within cultivated fields are characterised by a significantly decrease in tree diversity in the first ten years after clearing followed by a gradual decrease in subsequent years. This is similar to what has been reported elsewhere in studies by Bierregaard *et al*. (1992); Oba *et al*. (2002); Ogunleye*et al*. (2004) and Augusseau *et al*. (2006). In essence, older fields have repeatedly been subjected to clearing and ploughing before each planting season. Overtime, all potential sources for tree regeneration that include stumps, soil seed banks and live root stalks are destroyed. It is important to note that this effect of cultivation was observed on alpha diversity (field diversity) of the woody plants. This decrease in diversity at field level can be attributed to two factors, firstly, the physical removal of trees during land preparation for cultivation each year and secondly the depletion of soil nutrients with each subsequent year of cultivation. This then highlights the importance of alternating periods of cultivation with fallowing. More importantly, this result highlights the need for research focused on understanding soil nutrient dynamics within cultivated landscapes.

In contrast to the observations made within cultivated fields, results on woody species beta diversity (total woody species diversity within each land-cover category) indicated that, conversion of natural woodlands to cultivated landscapes did not result in decrease in tree diversity or complete removal of any species from the landscape. Thus agriculture activities in this area do not necessarily conflict with woody species diversity at landscape level. This is largely attributed to the presence of natural vegetation patches, trees left on field edges and the surrounding natural woodland which act as sources of seed for reintroduction of species within cleared areas. However, clearing of natural woodlands for cultivation and subsequent modification of the soil through cultivation allows other species to establish on the cleared areas and this raises questions on, what impacts these new species have on the soil nutrient dynamics?, and what are the invasive capabilities of the new species?

On both *mutapo* and *bandate* soil, the characteristic tree species to establish on the newly opened fields include *A. tortilis* subsp. *spirocarpa*, *C. elaeagnoides*, *M. zanzibarica*, *D.*

qualoensis, *D. cinerea*, *C. mossambicense*, *G. monticola* and *A. versicola*. It is interesting to note that most of these species are early succession species with strong invasive capacities. *Dichrostachys cinerea* in particular is known to cause losses in agricultural production and restoration of invaded lands is often costly. It is a thorny, fast-growing woody bush or shrub which invades fields, wasteland, road sides, pasture lands and other disturbed areas (van Wyk and van Wyk, 1997; Coates-Palgrave, 2002). The spread of *D. cinerea* is a potential bottleneck for the survival of other species in the area.

Several factors trigger the incursion of bush encroachment in savanna, these include anthropogenic factors (Angassa and Oba, 2008), deforestation, suppression of fire (Oba *et al.*, 2000) and heavy grazing pressure (Westbrooks, 1998; Stohlgren, 2007; Terefe *et al.*, 2011). Given the on-going extensive clearing and disturbance of natural vegetation, it should be highlighted to farmers, extension officers, landscapes managers and policy makers how the current farmer practices raise the likelihood of invasive species such as *A. tortilis* subsp. *spirocarpa* and *D. cinerea* invading the whole landscape. Thus, there is an urgent need for systematic surveys to rapidly assess the vulnerability of these natural landscapes and specific habitats to invasion and eventual dominance of certain woody species in order to guide research, control and restoration efforts.

Results also indicate that woodland clearing for cultivation purposes is effecting removal and replacement of tree species previously dominant in natural woodland areas. Thus, clearing of woodlands is changing the pattern of species composition. In essence, cultivation is simultaneously effecting selective removal of dominant species (*C. mopane*, *D. kirkii*, and *X. torreana*), thus facilitating growth and dominance of *A. tortilis* subsp. s*pirocarpa* and other species such as *D. cinerea*. Woody species from the original vegetation were only observed as scattered individual trees within cultivated fields or on field edges of which the majority of these trees belonged to sets of tolerated species in the area, i.e. protected and left within fields during vegetation clearing. As highlighted by Leakey *et al*. (2004), tolerated trees within agricultural landscapes, have undergone a long traditional selection process and usually have a social value or provide a certain commodity to the community. In this area among tolerated species are species that provide edible fruits (e.g. *A. garckeana*, *Z. mauritiana*, *S. birrea*, *A. digitata*, *T. indica* L. and *D. kirkii*, medicinal plants (e.g. *Cassia abreviata* Oliv. and *Kigelia Africana* (Lam.) Benth.) and other locally important species (e.g. *L. cappassa*).

Results show that woodland clearing has resulted in isolated small patches of mature vegetation within extensive cultivated matrices characterized by vegetation in the form of saplings, regenerating stumps, shrubs and mature trees on field edges. This indicates a significant negative change in vegetation physiognomy. This change in vegetation physiognomy is severe in intensively cultivated landscapes. A shift in vegetation physiognomy at large spatial magnitudes has immediate implications on habitat diversity and quality for some wild species (Mac Nally *et al*., 2004). Thus, the simplification of vegetation physiognomy occurring in intensively cultivated agricultural landscapes could negatively affect some wild species which prefer complex vegetation structure. For example, avian species distribution has been shown to be associated with vegetation structure (physiognomy) (e.g. MacArthur *et al*., 1966; Rotenberry, 1985).

CHAPTER 5: Impact of agriculture on arthropod diversity and abundance

5.1. Results

5.1.1. *Charaxes* and Cetoniinae species diversity and abundance

Representatives of bio-indicator species were present in all sites and within all land-use zones along the agricultural intensification gradient. Sampling yielded some 1576 butterflies from eight species and two subspecies of the genus *Charaxes*. Butterfly abundance was highest in Site 1 and Site 2, with greater variation among trapping locations than in Sites 3 and 4 (Figure 5.1). Site 1 and Site 2 had an average capture score of 34.5 and 26.9, respectively, while Site 3 and Site 4 had an average capture score of 7.6 and 9.2, respectively. Table 5.1 shows the number and names of species present at each site. In terms of similarity, Site 1 and Site 2 shared the highest number of species, with a similarity index at 98 % while Sites 2 and Site 3 shared the lowest number of species with similarity index at 77 % (Table 5.2).

Seven Cetoniinae species were encountered across the study area. Sampling yielded a total of 1497 beetles. High numbers of beetles were recorded at Sites 1 and Site 2 than at Sites 3 and Site 4 (Figure 5.1). The average capture score per site was highest at Site 2 (30.4), followed by Site 1 (27.9) and Site 3 (9.7), and the lowest capture was at Site 4 (7.1). The number of individual beetles per trapping location was significantly different among the four sites (Kruskal-Wallis ANOVA with 80 cases, $H = 0.89_3$, p = 0.02). The number of species at each site is shown in Table 5.1). The Morisita-Horn sample similarity index showed high similarity among all sites (above 50 %). The highest number of shared species was recorded between Sites 1 and Site 3, with a 96 % similarity index while Site 1 and Site 4 shared the lowest number of species with a similarity index of 77 %. (Table 5.2).

Table 5.1. *Charaxes* and Cetoniinae species recorded at each site in the study area

Group	Species	Site 1	Site 2	Site 3	Site 4
Charaxes	*C. achamenes achamenes* Felder & Felder	+	+	+	+
	C. brutus natalensis Staudinger	+	+	+	+
	C. candiope Godart	+	+	-	-
	C. ethalion binghami Henning	+	+	+	+
	C. jahlusa argynnides Westwood	+	+	+	+
	C. jasius saturnus Butler	+	+	+	+
	C. bohemani Felder	+	+	+	+
	C. varanes vologeses Mabille	+	+	+	-
	C. zoolina climecki Westwood*	-	-	+	-
	C. zoolina zoolina Westwood*	+	-	-	-
	Number of species	**9**	**8**	**8**	**6**
Cetoniinae	*Chelorrhina savagei* Harris	+	+	+	+
	Chondrorrhina trivittata Schaum	+	+	+	-
	Cyrtothyrea marginalis Swartz	+	+	+	+
	Dyspilophora trivittata Schaum	+	+	+	+
	Pachnoda marginata Gray	+	-	+	-
	Plaesiorrhinella plana	+	+	+	-
	Spp 004	-	-	+	+
	Number of species	**6**	**5**	**7**	**4**

The symbol * represents subspecies of *Charaxes zoolina* and the positive sign (+) means present and negative sign means (-) absent

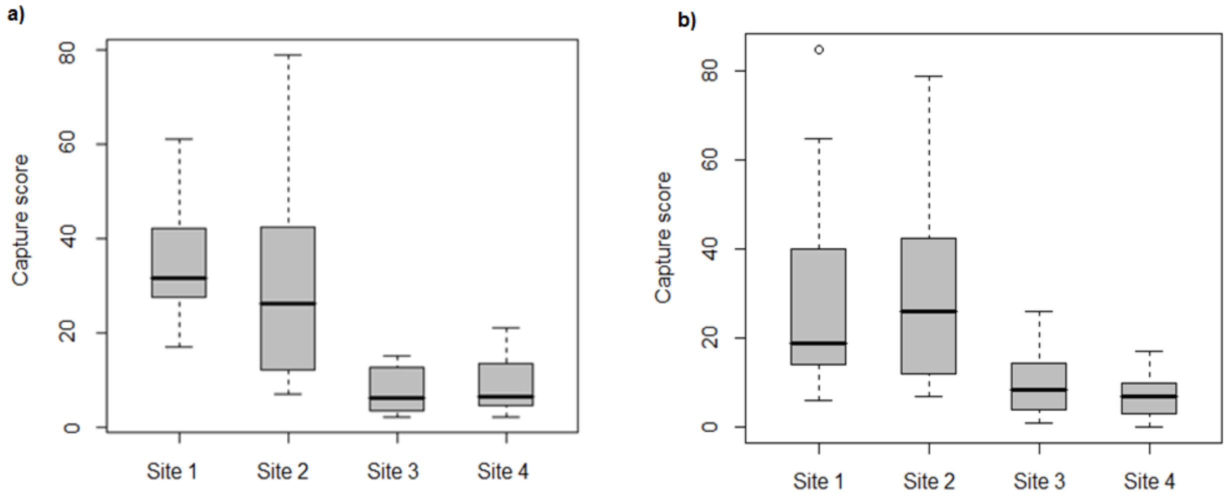


Figure 5.1. *Charaxes* (a) and Cetoniinae (b) abundance in each site

Table 5.2. Shared *Charaxes* and Cetoniinae species and similarity statistics for the four sites

	First Sample	Second Sample	Shared Species Observed	Morisita-Horn
Charaxes	Site 1	Site 2	8	0.98
	Site 1	Site 3	7	0.85
	Site 1	Site 4	6	0.91
	Site 2	Site 3	6	0.77
	Site 2	Site 4	6	0.87
	Site 3	Site 4	5	0.79
Cetoniinae	Site 1	Site 2	5	0.96
	Site 1	Site 3	6	0.80
	Site 1	Site 4	3	0.62
	Site 2	Site 3	5	0.90
	Site 2	Site 4	3	0.77
	Site 3	Site 4	4	0.93

Morisita-Horn sample similarity index calculated using EstimateS (Colwell 2006)

Table 5.3 provides a summary of abundances of each species. Among butterflies, *Charaxes jasius saturnus* was the most abundant species, making up to 37.0 % of the total butterfly species, with *C. brutus natalensis* and *C. achamenes achamenes* being the second most abundant, together making up to 33.6 % of the butterfly community. The two subspecies of *Charaxes zoolina* were the least abundant, with *C. zoolina climecki* making up only 0.8 % and *C. zoolina zoolina* making up 1.6 % of the encountered butterflies. Among species encountered in the study area, the following species were present at all four sites *C. achamenes achamenes*, *C. brutus natalensis*, *C. ethalion binghami*, *C. jahlusa argynnides*, *C. jasius saturnus* and *C. bohemani*. Some species were confined to specific sites within a specific land-use zone.

Among beetles (Table 5.4), *Dyspilophora trivittata* was the most abundant species, making up to 40 % of the total Cetoniinae abundance. *Chondrorrhina trivittata* was the second most abundant species, making up to 16 % of the total beetle abundance. *P. marginata* and *Spp 004*, were the least abundant species. Three species (*D. trivittata*, *Chelorrhina savagei* and *C. marginalis*) were present at all the sites, and in both natural woodlands and cultivated areas. *C. trivittata* and *P. plana* were present at three sites, except Site 4. *P. marginata* was confined to Site 1 and Site 3, while *Spp 004* was confined to Site 3 and Site 4.

Table 5.3. Total number of individual butterflies captured in each land-use zone (natural woodlands, W; Cultivated, C) in each study site

Species	Site 1		Site 2		Site 3		Site 4		Total	% Total
	W	**C**	**W**	**C**	**W**	**C**	**W**	**C**		
C. achamenes achamenes	60	54	41	12	30	18	22	10	247	15.7
C. brutus natalensis	83	42	44	55	19	5	14	10	272	17.9
C. candiope	72	18	65	26	0	0	0	0	181	11.5
C. ethalion binghami	14	6	10	2	0	0	10	1	43	2.7
C. jahlusa argynnide	10	10	22	10	4	1	4	1	62	3.9
C. jasius saturnus	131	123	110	81	25	16	74	23	583	37.0
C. bohemani	17	6	9	22	15	0	13	3	85	4.8
C. varanes vologeses	25	11	14	4	10	1	0	0	65	4.1
C. zoolina climecki	0	0	0	0	13	0	0	0	13	0.8
C. zoolina zoolina	12	0	0	0	12	1	0	0	25	1.6
Total	**419**	**270**	**325**	**212**	**123**	**42**	**137**	**48**	**1576**	

Table 5.4. Total number of individual beetles captured in each land-use zone (natural woodlands, W; Cultivated, C) at each study site

	Site 1		**Site 2**		**Site 3**		**Site 4**		**Total**	**% Total**
Species	**W**	**C**	**W**	**C**	**W**	**C**	**W**	**C**		
Chelorrhina savagei	28	26	49	50	19	11	20	8	211	14.1
Chondrorrhina trivittata	97	22	72	31	14	0	0	0	236	15.8
Cyrtothyrea marginalis	5	13	48	34	37	7	29	14	187	12.5
Dyspilophora trivittata	200	57	143	98	38	24	40	4	604	40.3
Pachnoda marginata	13	17	0	0	12	3	0	0	45	3.0
Plaesiorrhinella plana	61	18	51	29	7	4	0	0	170	11.4
Spp 4	0	0	0	0	18	0	20	6	44	2.9
Total	**404**	**153**	**363**	**242**	**145**	**49**	**109**	**32**	**1497**	

On average, more individuals of *Charaxes* were caught per sample in natural woodland than in cultivated areas (Figure 5.2). At the 5 % significance level, in three sites (Sites 1, 3 and 4) the Welch two sample t-test showed that the differences in *Charaxes* abundance were significantly higher in natural woodlands than in cultivated areas, while at Site 2, the Welch two sample t-test showed no significant differences ($t = 1.31_{15.4}$, $p = 0.211$) (Table 5.5). Lower abundances and more pronounced differences in individual numbers between the two land-use zones were observed at sites considered to be highly intensified (Site 3, $p = 8.242e-05$; Site 4, $p = 5.772e-05$, Table 5.5). At each site, certain species were responsible for the differences observed between land-use zones. Table 5.6 shows p-values for each species. Similar to *Charaxes*, Cetoniinae abundance was higher in natural woodlands than in cultivated areas (Figure 5.3). The Welch two sample t-test showed significant differences ($p < 0.05$ in all sites, Table 5.7) in beetle abundance between natural woodlands and cultivated areas. The Welch two sample t-test revealed maximal differences between the two land-use zones in intensively exploited sites (Site 3, $p = 0.00$ and Site 4, $p = 2.165e-05$). Table 5.8 shows p-values from the Kruskal-Wallis test on individual species.

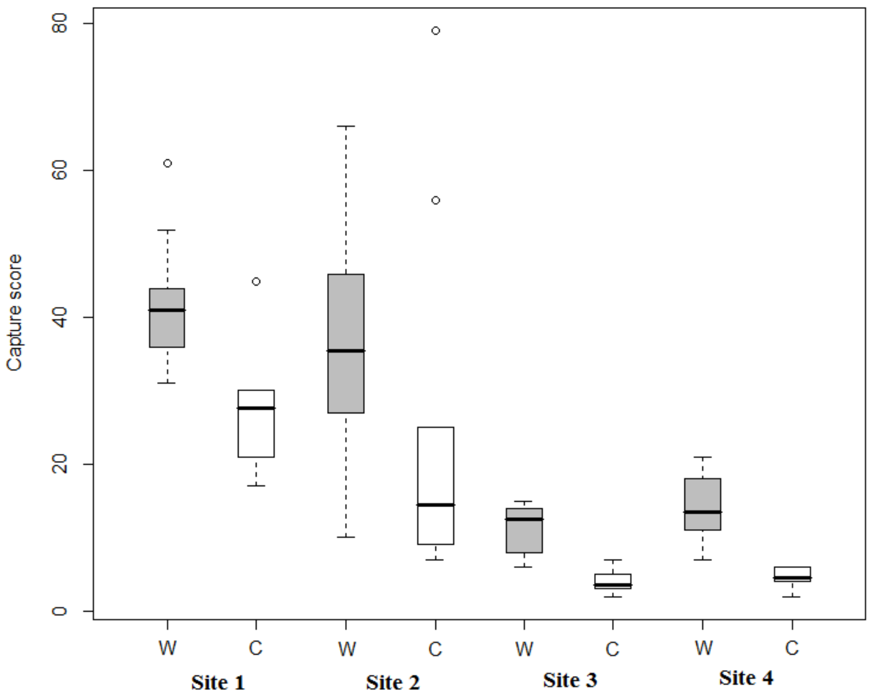


Figure 5.2. *Charaxes* abundance in natural woodland (W) and cultivated areas (C) at each site

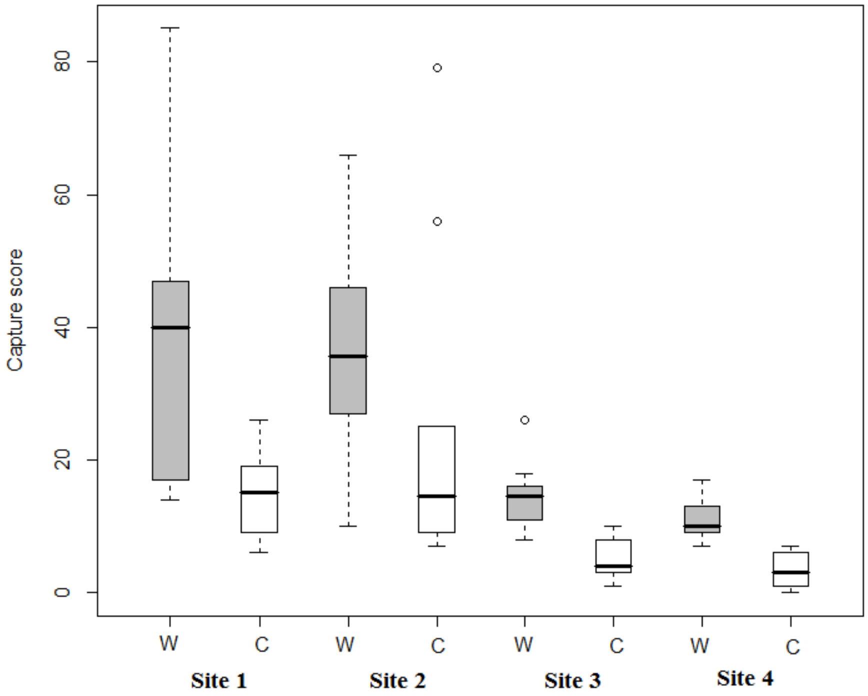


Figure 5.3. Cetoniinae abundance in natural woodland (W) and cultivated areas (C) at each site

Table 5.5. Comparison of butterfly abundance and species diversity between natural woodlands and cultivated areas at each site using the Welch two sample t-test at95 % confidence interval

Measure	Site	Land-use	N	Mean	Welch two sample t-test		
					t-value	d.f	p-value
Abundance	1	Woodland	20	41.9	3.9	17.7	0.04*
		Cultivated	20	27.0			
	2	Woodland	20	36.3	1.3	15.4	0.21
		Cultivated	20	24.5			
	3	Woodland	20	11.2	5.8	12.0	0.00*
		Cultivated	20	4.0			
	4	Woodland	20	13.9	6.3	10.9	0.00*
		Cultivated	20	4.6			
Shannon-Weiner	1	Woodland	20	0.1	4.3	9.7	0.00*
		Cultivated	20	0.0			
	2	Woodland	20	0.1	3.7	9.5	0.00*
		Cultivated	20	0.0			
	3	Woodland	20	0.2	4.6	10.6	0.00*
		Cultivated	20	0.1			
	4	Woodland	20	0.1	3.3	9.1	0.00*
		Cultivated	20	0.0			

The symbol * represents a statistically significant difference at $p < 0.05$

Table 5.6. Comparison of beetle abundance and species diversity between two habitat types at each site using the Welch two sample t-test at 95 % confidence interval.

Measure	Site	Land-use	N	Mean	Welch two sample t-test		
					t-value	d.f	p-value
Abundance	1	Woodland	20	40.4	3.3	10.0	0.00*
		Cultivated	20	15.3			
	2	Woodland	20	36.3	1.0	15.0	0.02*
		Cultivated	20	24.5			
	3	Woodland	20	14.5	5.1	14.0	0.00*
		Cultivated	20	4.9			
	4	Woodland	20	10.9	5.8	16.0	0.00*
		Cultivated	20	3.2			
Shannon-Weiner	1	Woodland	20	0.3	3.8	12.0	0.00*
		Cultivated	20	0.2			
	2	Woodland	20	0.2	6.1	18.0	0.00*
		Cultivated	20	0.1			
	3	Woodland	20	0.1	6.0	13.0	0.00*
		Cultivated	20	0.1			
	4	Woodland	20	0.1	4.0	10.0	0.00*
		Woodland	20	0.1	3.7	13.0	0.00*

The symbol * represents a statistically significant difference at $p < 0.05$

Table 5.7. Comparison of individual *Charaxes* species abundance between natural woodlands and cultivated areas at each site using the Kruskal-Wallis test at 95 % confidence interval

Species	**Site 1**		**Site 2**		**Site 3**		**Site 4**	
	t	**p**	**t**	**p**	**t**	**p**	**t**	**p**
C. ethalion binghami	1.6	0.21	9.7	0.00*	0.7	0.40	9.8	0.00*
C. achamenes achamenes	0.2	0.65	7.9	0.01*	7.9	0.01*	3.3	0.01*
C. bohemani	1.5	0.23	1.9	0.16	6.8	0.01*	5.1	0.02*
C. brutus natalensis	4.3	0.04*	0.2	0.68	0.5	0.47	0.7	0.40
C. jahlusa argynnide	0.4	0.52	2.7	0.10	1.1	0.30	3.3	0.07
C. jasius saturnus	0.2	0.65	3.5	0.06	4.7	0.03*	12.6	0.00*
C. varanes vologeses	4.3	0.04*	3.2	0.07				
C. candiope	11.0	0.00*	6.6	0.01*				
C. zoolina zoolina	3.3	0.07						

The symbol * represents a statistically significant difference at $p < 0.05$

Table 5.8. Comparison of individual Cetoniinae species abundance between natural woodlands and cultivated areas in each site using the Kruskal-Wallis test at 95 % confidence interval.

Species	**Site 1**		**Site 2**		**Site 3**		**Site 4**	
	t	**p**	**t**	**p**	**t**	**p**	**t**	**p**
Chelorrhina savagei	0.0	0.98	0.0	0.93	1.0	0.32	3.4	0.07
Chondrorrhina trivittata	5.8	0.02*	4.8	0.03*	10.9	0.00*		
Cyrtothyrea marginalis	3.6	0.06	0.7	0.42	11.2	0.00*	2.7	0.10
Dyspilophora trivittata	7.4	0.01*	1.3	0.23	0.8	0.36	13.9	0.00*
Pachnoda marginata	0.4	0.55			3.1	0.08		
Plaesiorrhinella plana	7.0	0.01*	2.9	0.09	0.5	0.47		
Spp 4							3.3	0.07

The symbol * represents a statistically significant difference at $p < 0.05$

5.1.2. Effect of habitat type and edge on butterfly *Charaxes* and Cetoniinae abundance and species diversity

Generally, mean *Charaxes* abundance was higher in natural woodland than in cultivated areas. A comparison of the four transects revealed contrasting trends in butterfly abundance among the four sites. Sites 1 and 3, recorded lower abundances for Transect 1 (natural woodland interior) than Transect 2 (natural woodland edge), with much lower abundances for Transects 3 (cultivated edge) and 4 (cultivated interior). In contrast Sites 2 and 4, showed a progressive decrease in abundance from Transect 1 to Transect 4 (Figure 5.4). There was a decreasing trend in *Charaxes* species diversity from natural woodland towards cultivated areas at all sites (Figure 5.5). Two-Way ANOVA test showed a significant ($p < 0.05$) effect of habitat on *Charaxes* abundance with the interaction effect also present (Table 5.9). This showed higher abundance within woodland habitats and the significant ($p < 0.05$) interaction also showed higher diversity and abundance in woodland interior and cultivated edge than woodland edge and cultivated interior (Figure 5.5). Both habitat and distance showed significant ($p < 0.05$) effects on diversity (Shannon-Wiener index).

Natural woodlands harbored higher levels of Cetoniinae abundance than cultivated areas at all four sites. At all sites, there was a decrease in Cetoniinae abundance from natural woodlands to cultivated areas (from Transect 1 to Transect 4). At Sites 3 and 4, there was no difference in Cetoniinae abundance between Transect 1 and Transects 2 in natural woodlands (Figure 5.6). At Site 1, Site 2 and Site 4, beetle species diversity decreased from natural woodland areas towards cultivated areas (Figure 5.7). Two-Way ANOVA test showed only the significant ($p < 0.05$) effect of habitat type on Cetoniinae abundance and diversity (Table 5.8) and no interaction effect was present for both measures (Figure 5.8). Cetoniinae diversity and abundance was significantly higher within woodland habitats, but no edge effect was detected in both habitat types.

Table 5.9. Two-Way ANOVA test on the effect of habitat type (woodland versus cultivated) and distance (interior versus edge transect) on *Charaxes* and Cetoniinae species diversity (Shannon-Weiner index) and abundance.

Group	**Measure**	**Variable**	**F-value**	**p - value**
Charaxes	abundance	Habitat	12.5	0.00**
		Distance	0.5	0.45
		Habitat: Distance	0.2	0.04*
	diversity	Habitat	70.2	0.00**
		Distance	29.0	0.00**
		Habitat: Distance	17.3	0.00**
Cetoniinae	abundance	Habitat	12.6	0.00**
		Distance	1.1	0.29
		Habitat: distance	0.2	0.62
	diversity	Habitat	4.6	0.03*
		Distance	1.4	0.25
		Habitat: distance	0.1	0.83

The asterisks * represents a statistically significant variable effect at * $p < 0.05$; ** $p < 0.01$

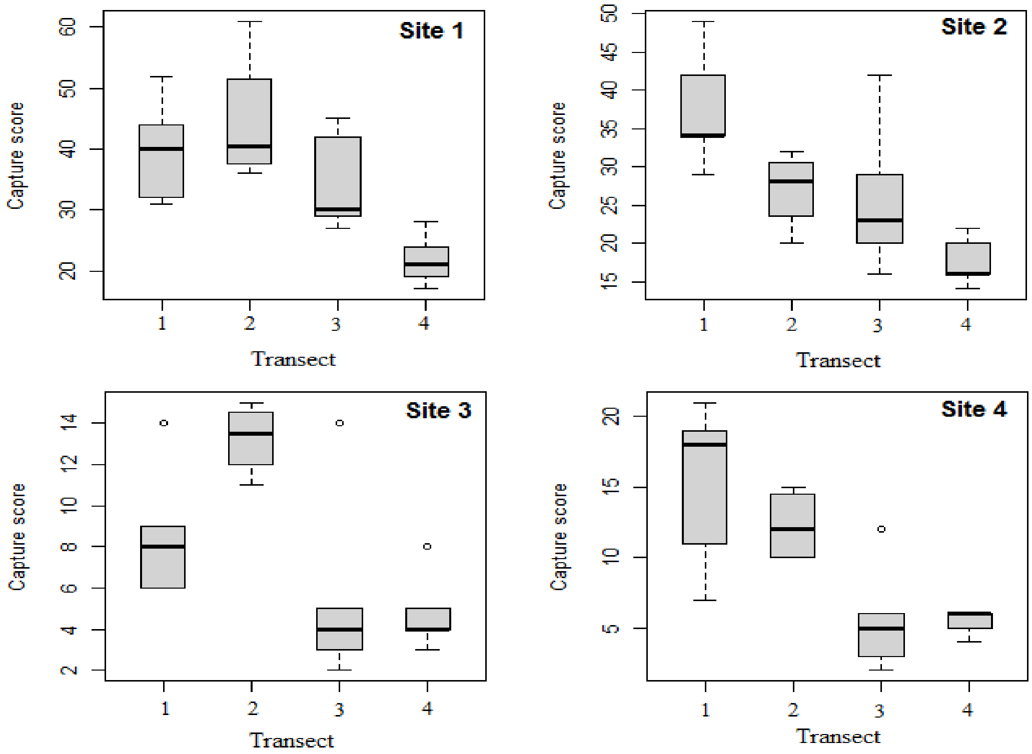


Figure 5.4. A comparison of *Charaxes* abundance between two transects in each habitat

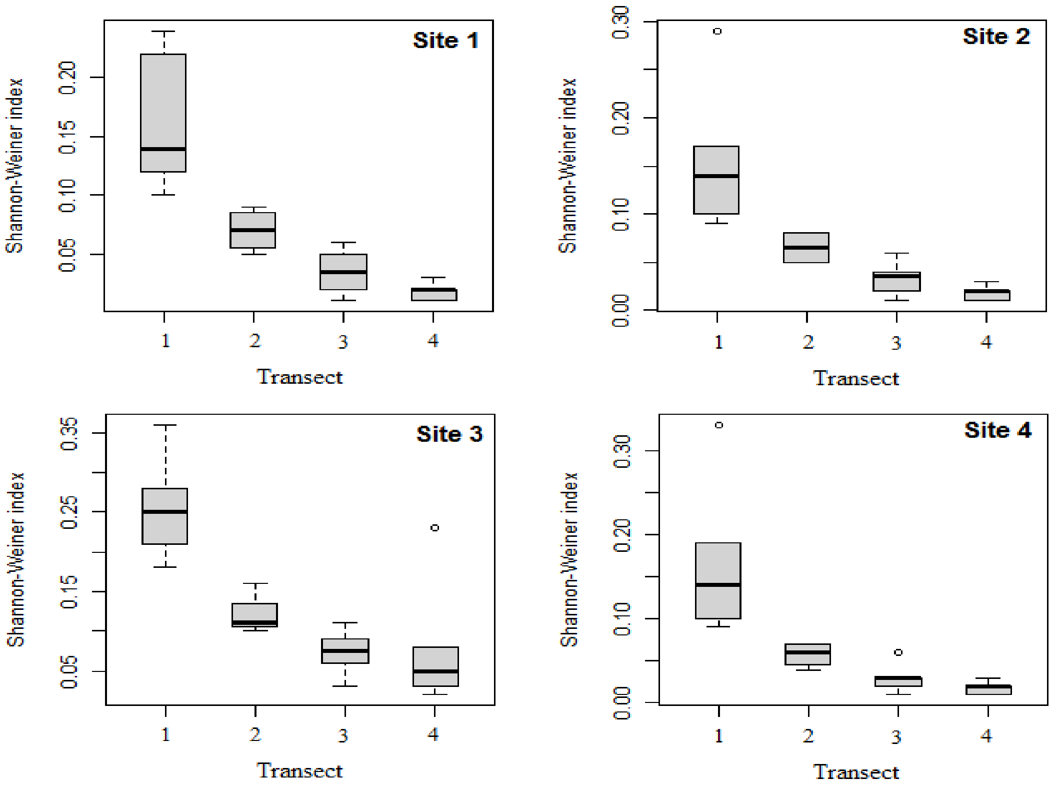


Figure 5.5. A comparison of *Charaxes* diversity between two transects in each habitat type

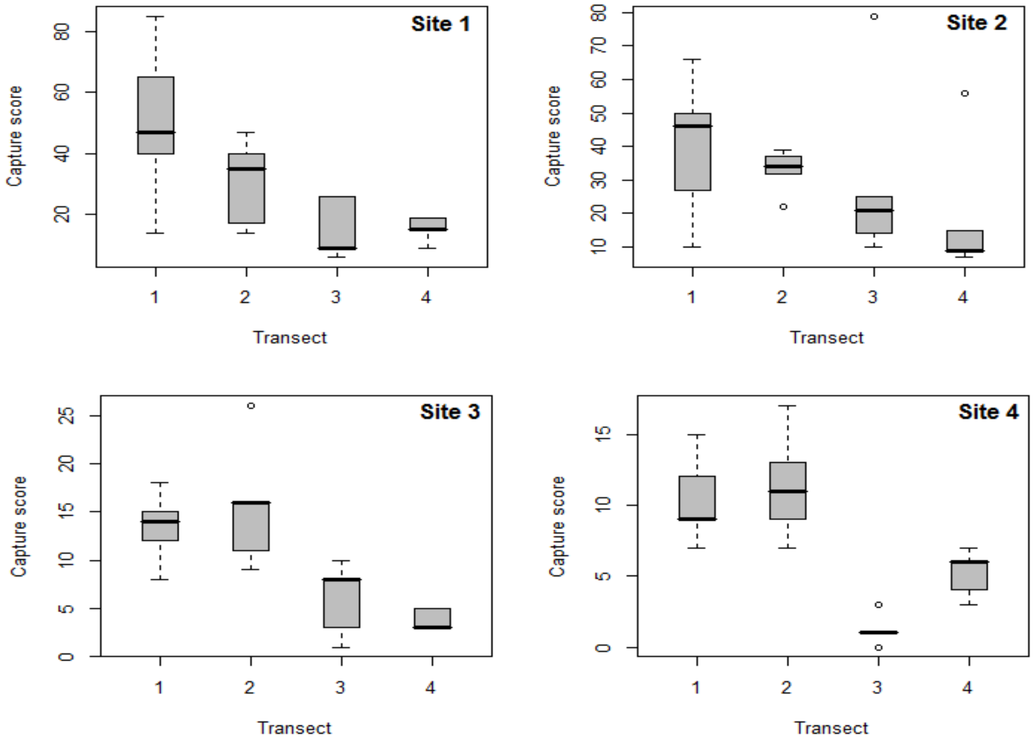


Figure 5.6. A comparison of Cetoniinae abundance between two transects in each habitat type in four sites

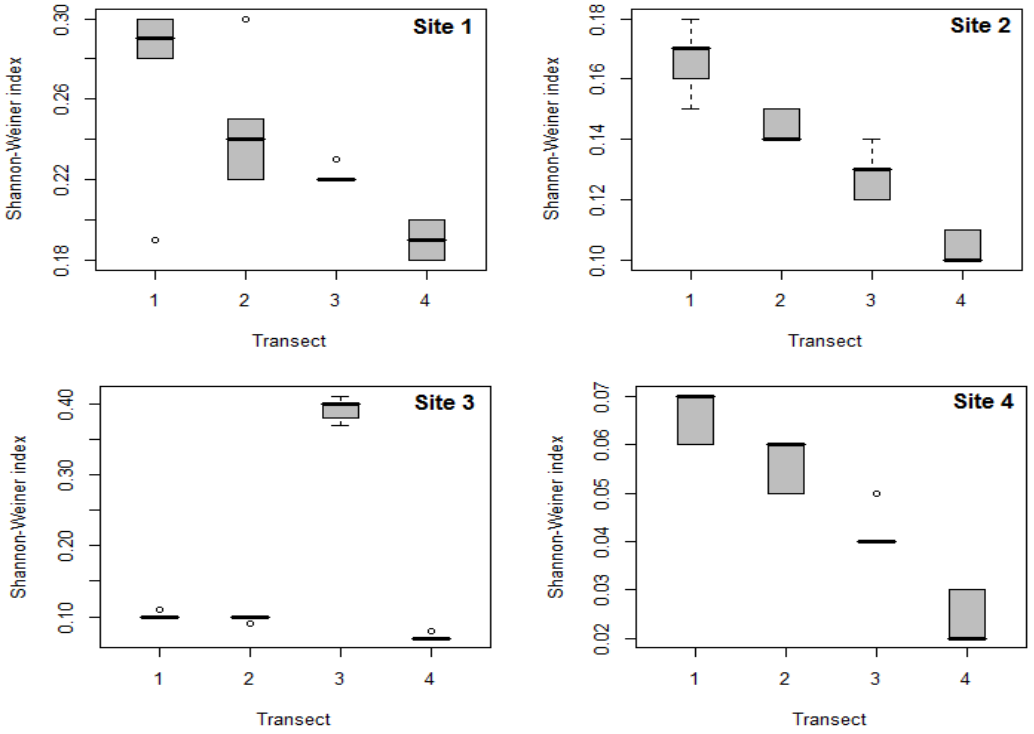


Figure 5.7. A comparison of Cetoniinae species diversity between two transects in each habitat type in all sites

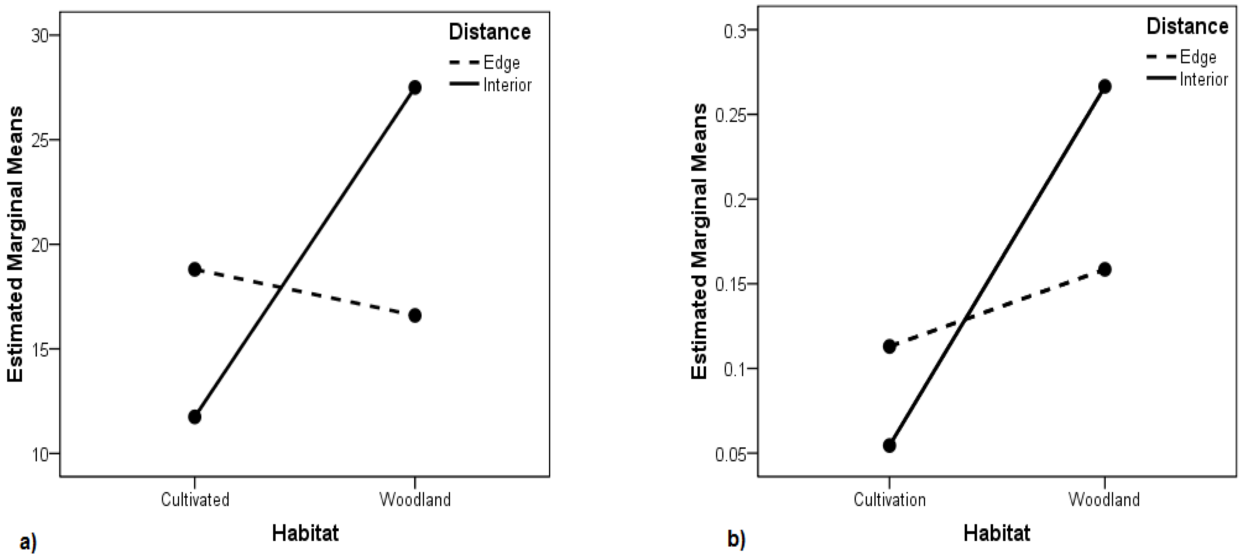


Figure 5.8. Interaction plot showing the effect of habitat and distance on *Charaxes* (a) Estimated Marginal Means of *Charaxes* abundance and (b) Estimated Marginal Means of *Charaxes* diversity.

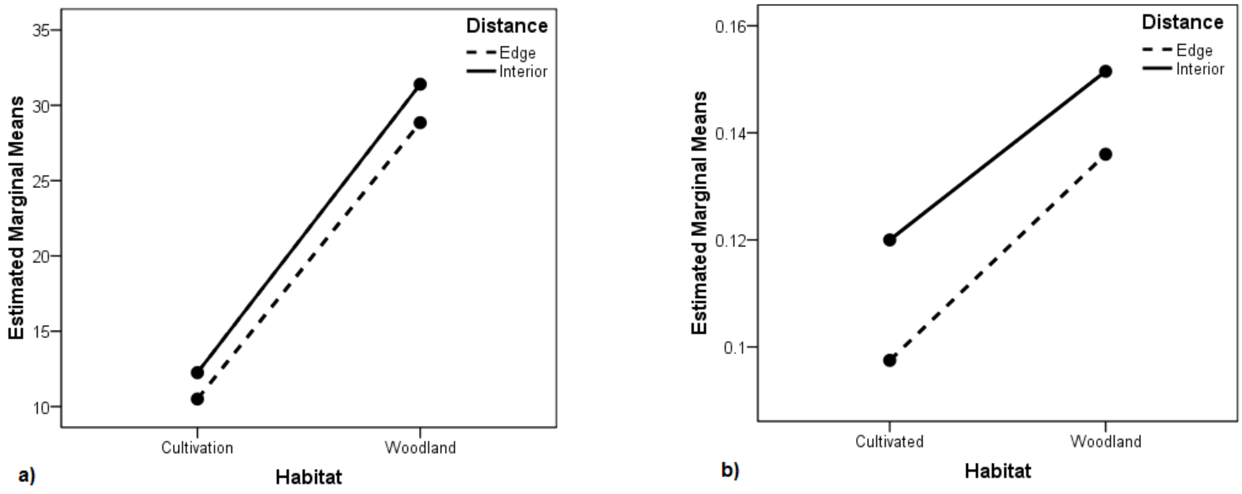


Figure 5.9. Interaction plot showing the effect of habitat and distance on Cetoniinae (a) Estimated Marginal Means of Cetoniinae abundance and (b) Estimated Marginal Means of Cetoniinae diversity.

5.2. Discussion

Results suggest that clearing of natural woodland for cultivation purposes accounts for the differences in butterfly and beetle communities at landscape and local scale. Thus, the type and quality of habitat largely accounts for the abundance and diversity of arthropods in a given landscape. This was evidenced by the high diversity and abundance of butterfly and beetle taxa within natural woodland areas compared to cultivated areas. Apart from cultivation, it is acknowledged in this study that, other site differences such as vegetation composition may explain the observed trends in diversity.

Arthropod groups used in this study are phytophagous insects that draw their food supply from a restricted number of plant species, both at larval and adult stages (Larsen 1996). Some 400,000 species of known insects are plant feeders (New, 1988). Thus, phytophagous insects make up approximately 25% of all living species on earth (Strong *et al*., 1984). Plants and phytophagous insects have close and evident ecologically intimate relationships. In this regard, it has been observed that many adult butterflies and other insects lay their eggs on certain food plants with great precision (Larsen 1996). Compared to *Charaxes*, Cetoniinae larval stages that feed on dead wood and leaf material are less resource specific (Bouyer *et al*., 2007). Thus, the presence of phytophagous insect groups is expected to be widely affected by the availability of host plants.

The presence of undisturbed habitats is essential for some specialist species, for example *C. zoolina zoolina* and *C. zoolina climecki* are confined to unique undisturbed woodlands. These could be resource specific or specialist species that require a well-defined habitat with specific environmental and floral characteristics (Larsen, 1996). Specialist species that require pristine habitats are usually threatened by slight disturbances. Contrary to the requirements of specialist species are generalist species. These species indicated an ability to thrive in a wide variety of conditions making use of a variety of different resources and habitats. These species responded positively to woodland clearing at some sites with an increase in abundance within cultivated areas. Some species, however, did not show responses consistent with the two broad groups described above. These species occurred in both disturbed and undisturbed areas, but their lower abundances revealed a limited tolerance to disturbance. These species tolerate intermediate levels of disturbance and these conditions are typical of the edge zone which is influenced by the interface between a transformed agricultural area and natural woodland.

The study also indicates that effects of woodland conversion on insect taxa studied are more pronounced in areas with high agricultural intensification. Intensively cultivated landscapes can be devoid of food, larval host plants and nectar resources for most arthropod groups, butterflies and beetle included (Rundlof and Smith, 2006). As shown by Giller *et al.* (1997), one characteristic of intensified agriculture is the reduction in planned above-ground diversity of plants and animals due to crop/animal specialization. Rigorous weeding through herbicide application and low crop diversity are the main factors resulting in reduced plant diversity within intensively farmed landscapes in the mid-Zambezi valley. Weeds, in fact, may have some functional importance within agricultural landscapes. They can support a high diversity of insects through increased vegetation heterogeneity and primary productivity (Benton *et al.*, 2003; Ruby *et al.*, 2011).

Apart from herbicide use, intensively farmed landscapes in the mid-Zambezi valley are also characterised by intensive pesticide use. Intensive pesticide use is more pronounced in cotton fields in particular (Baudron *et al.*, 2011a). More often than not, however, they have adverse effects on non-targeted species (Croft, 1990). Moreover, the effect of pesticides is not limited to treated areas, but can extend to adjacent areas housing beneficial species. Thus, loss of insect diversity in agricultural landscapes can have adverse effects on pollination, decomposition and other animal groups, such as insect eating birds (Chamberlain *et al.*, 2000; Benton *et al.*, 2002). Herbicides and pesticides are usually targeted at specific weed and pest species respectively. The concern here is that, both herbicides and pesticides have unpredictable effects on non-targeted flora and fauna. In this regard, there is need for future research to focus on empirically determining the effects of herbicides and pesticides on above-ground and below-ground flora and fauna within intensively farmed landscapes and adjacent areas in the mid-Zambezi valley.

6.1. Results

6.1.1. The dynamics in vegetation structure and composition with increasing fallow age

A total of seventy four woody species were recorded within fallow areas. Fallows on *bandate* soil recorded a marginally higher number of woody species (46 species) compared to fallows on *mutapo* soil (45 species). Table 6.1 summarises tree dbh and density, and shrub density and number of primary shoots per plant on *mutapo* and *bandate* soils. Generally, all parameters measured, for each fallow age, were considerably higher in fallows on *bandate* soil than in fallows on *mutapo* soil (Table 6.1). The early years (1-3 years) of fallowing, were characterised by high proportion of shrubs, with a few young trees (< 8 cm dbh) present, on both soil types. Figure 6.1 and Figure 6.2 show changes in fallow vegetation with time.

Table 6.1. Summary of changes in tree density and DBH with increasing fallow age on *mutapo* and *bandate* soil.

Year	**Trees**				**Shrubs**			
	Mean dbh (cm)		Density (ha^{-1})		Density (ha^{-1})		shoots/plant)	
	M	***b***	***m***	***b***	***m***	***b***	***m***	***b***
1	5.3	5.6	3.3	4.6	1383.0	2422.6	15.7	17.4
2	5.4	5.8	33.4	74.1	1419.4	1593.8	13.1	13.3
3	7.3	7.7	57.7	289.3	1096.3	1202.8	8.7	9.4
4	7.3	7.6	226.8	389.7	1250.4	311.1	7.9	8.1
5	8.7		333.5		441.1		7.1	
6	8.0	9.8	402.6	404.7	381.8	312.7	6.4	4.3
7	8.7	9.7	423.4	439.7	333.3	370.0	3.7	4.1
8	10.0		437.2		285.1		4.2	
9	10.7	11.7	530.6	407.5	237.9	248.5	3.6	4.0
10		12.8		533.2		201.1	4.4	3.3
12		14.5		518.1		153.9	3.1	4.1

m = *mutapo*, ***b*** = *bandate* soil

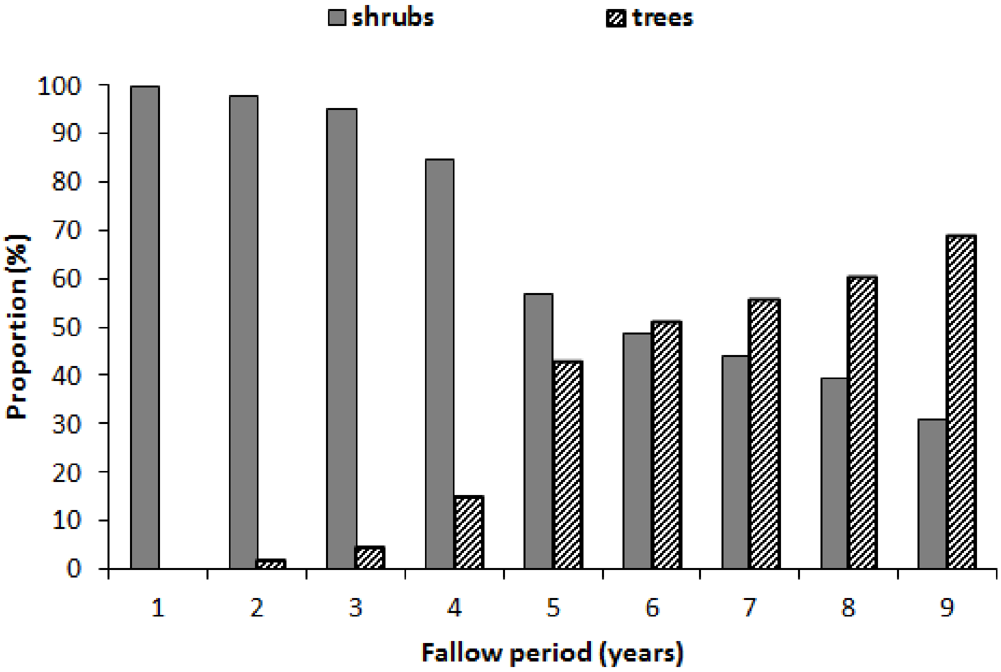


Figure 6.1. Proportion of mature trees and shrubs in change of fallow vegetation with time on *mutapo* soil.

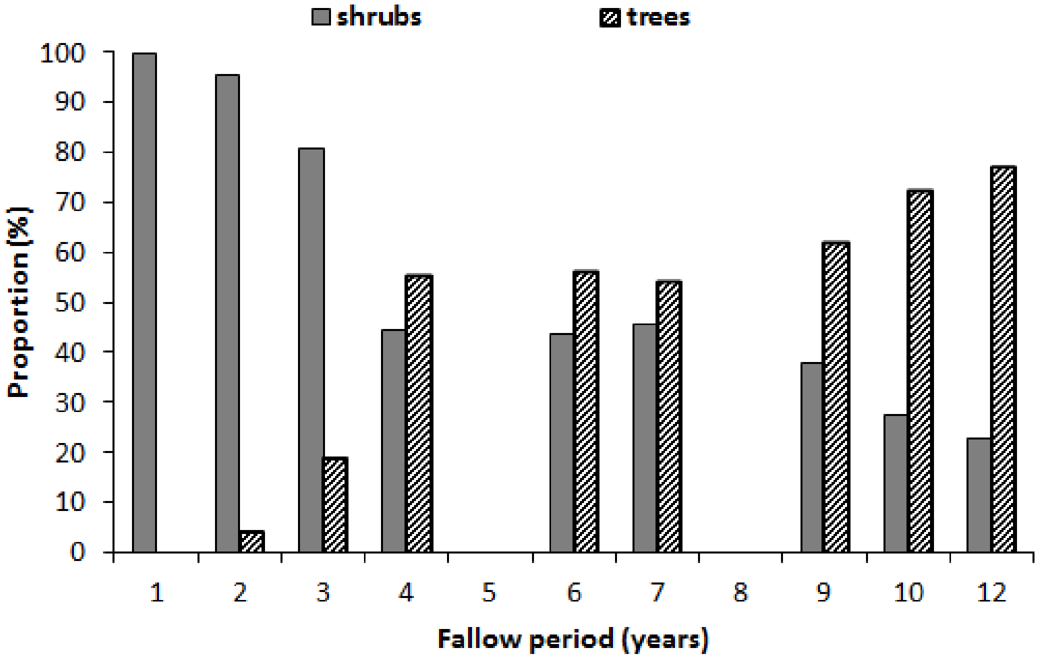


Figure 6.2. Proportion of mature trees and shrubs in change of fallow vegetation with time on *bandate* soil.

Woody species composition within fallow areas changed with period of fallowing (Table 6.2). Young fallows (class 1, 1 – 4 years) were dominated by shrubs and small trees of *Combretum eleaegnoides, Acacia tortilis* subsp. *spirocarpa* and *C. mopane* with *L. cappassa* completing the list of dominant species on *bandate* soil. Fruit trees (*A. garkeana, D. kirkii* and *Z. mauritiana*) were also present in high numbers. Older fallows (5-8 years) were dominated by medium sized *A. tortilis* subsp. *spirocarpa* trees. The dominance of *A. tortilis* subsp. *spirocarpa* continued in much older fallows (9-12 years) with very limited presence of other tree species. On both soil types, the dominant species within fallow areas were completely different from those dominant in the woodland plots. Older fallows (9-12 years) and woodlands had two species in common on *mutapo* soil and one species in common on *bandate* soil (Table 6.2). The Morisita-Horn similarity measure ranged from a minimum of 0.19 between young fallows and woodland plots, to a maximum of 0.38 between oldest fallows and woodland plots (Table 6.2).

Woody species diversity measures quantified within fallow areas were higher on bandate soil than mutapo soil. On both soil types, the number of woody species was significantly higher (*mutapo* soil: $F = 1.26_{2,1}$, $p = 0.01$; *bandate* soil: $F = 2.46_{2,1}$, $p = 0.02$) in the early years of fallowing (1 to 2 years).The relationship between fallow age and the number of woody species was more distinct on *mutapo* than on *bandate* soil (Figure 6.3). There was a strong variability in the number of species among fallows of the same age. Although the number of species decreased with increasing fallow age, the linear relationship between the number of woody species and fallow age was not significant with R^2 less than 50 % and $p > 0.05$ on both soil type (Figure 6.3). There was a significant ($p < 0.01$) exponential decrease in tree diversity with increasing fallow age on both soil types (Figure 6.4). In fact, on both soils higher R^2 values ($R^2 > 0.8$) were obtained when fallow age was used to predict woody tree diversity (Shannon-Weiner index).

Table 6.2. The five dominant woody species, ranked by stocking density, tree height and age (1 – 5) in each fallow age class and in woodland on *mutapo* and *bandati* soil.

Order	1 – 4 years	5 – 8 years	9 – 12 years	Woodland
Mutapo				
1	*Combretum eleaegnoides*	*Acacia tortilis subsp. spirocarpa*	*Acacia tortilis subsp. spirocarpa*	*Colophospermum mopane*
2	*Acacia tortilis subsp. spirocarpa*	*Markhamia zanzibarica*	*Markhamia zanzibarica*	*Diospyros qualoensis*
3	*Colophospermum mopane*	*Combretum eleaegnoides*	*Ziziphus mauritiana*	*Combretum mossambicense*
4	*Markhamia zanzibarica*	*Colophospermum mopane*	*Diospyros qualoensis*	*Combretum eleaegnoides*
5	*Dichrostachys cinerea*	*Ziziphus mauritiana*	*Combretum eleaegnoides*	*Grewia monticola*
H'	2.6	1.8	1.7	1.45
I_M	0.26	0.33	0.38	N/A
Bandati				
1	*Combretum eleaegnoides*	*Acacia tortilis subsp. spirocarpa*	*Acacia tortilis subsp. spirocarpa*	*Diospyros kirkii*
2	*Lonchocarpus cappassa*	*Lonchocarpus cappassa*	*Lonchocarpus cappassa*	*Lonchocarpus cappassa*
3	*Acacia tortilis subsp. spirocarpa*	*Combretum eleaegnoides*	*Combretum eleaegnoides*	*Terminalia sericea*
4	*Terminalia sericea*	*Combretum mossambicense*	*Ziziphus mauritiana*	*Sclerocarya birrea*
5	*Diospyros kirkii*	*Ziziphus mauritiana*	*Azanza garkeana*	*Xylia torreana*
H'	3.1	2.3	2.1	3.6
I_M	0.19	0.23	0.28	N/A

Also shown is the number of sampled plots per age class (*n*), mean species richness, Shannon-Wiener index (*H'*) and Morisita-Horn similarity index (I_M).

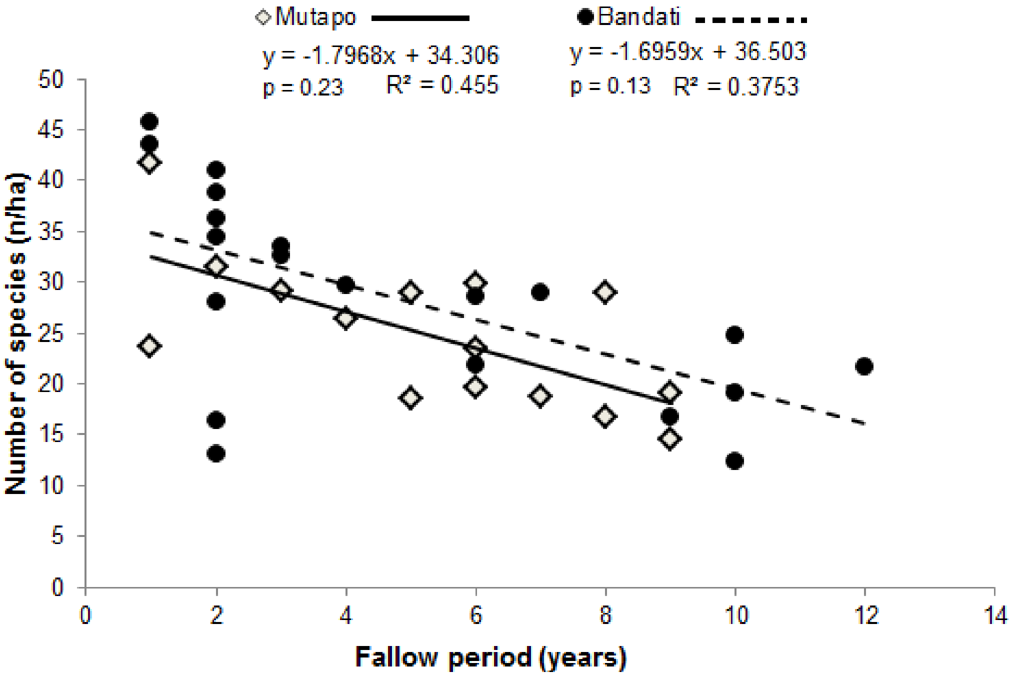


Figure 6.3. The relationship between number of woody species and fallow age on *mutapo* and *bandate* soil.

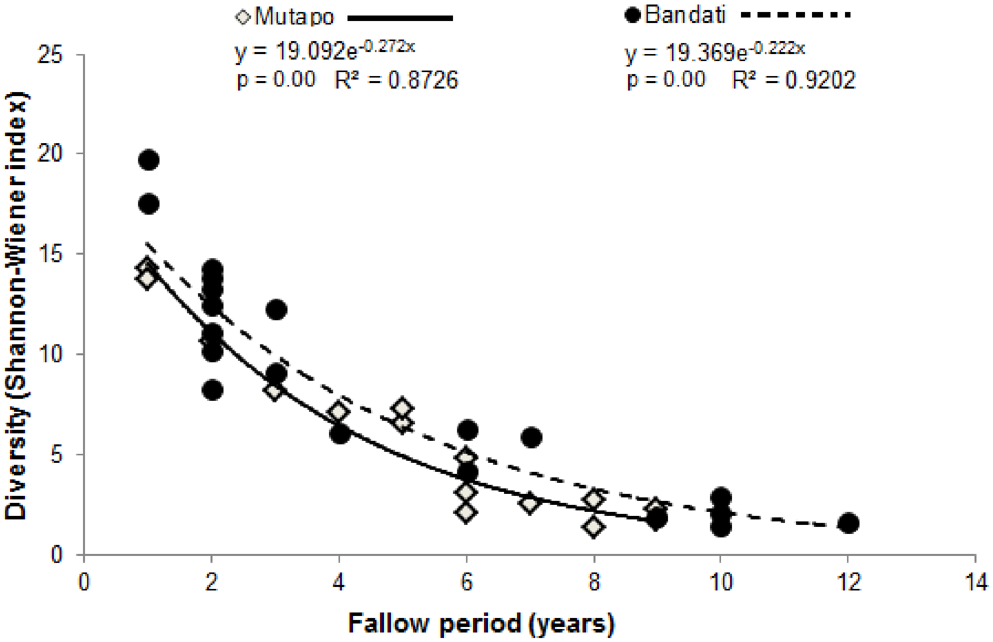


Figure 6.4. The relationship between woody species diversity and fallow age on *mutapo* and *bandate* soil.

6.1.2. Above-ground biomass accumulation as a function of fallow age

Regression analysis on tree biomass data (Appendix 2) from fallows showed a significant ($p < 0.05$) linear increase in tree above-ground biomass with increasing fallow age. This linear relationship was obtained for each of the four allometric equations with high R^2 values ranging from 70 % to 94 % (Figure 6.5). The mean biomass estimates from the four allometric equations showed a higher R^2 value on *mutapo* soil ($R^2 = 0.\ 95$) than on *bandate* soil ($R^2 = 0.85$) (Figure 6.6). The same regressions on the mean biomass estimates showed that biomass accumulation rate decreased on both soils, from 1.12 t ha^{-1} yr^{-1} in fallows aged 1-5 to 0.82 t ha^{-1} yr^{-1}in fallows aged 6-9 years on *mutapo* soil and from 1.75 t ha^{-1} yr^{-1} in fallows aged 1-5 to 1.32 t ha^{-1} yr^{-1} in fallows aged 6-12 years on *bandate*. Biomass accumulation rate was higher on *bandate* soil with 0.91 t ha^{-1} $year^{-1}$ than on *mutapo* soil with 0.84 t ha^{-1} $year^{-1}$. Regression analysis also revealed negative linear relationships between accumulated tree biomass and number of woody species (Figure 6.7) and between accumulated tree biomass and tree diversity (Shannon-Wiener index) with increasing fallow age (Figure 6.8). Linear trends observed were not significant, $p > 0.05$ in both cases with low R^2 values of 0.41 (number of species) and 0.44 (tree density).There was a positive and significant ($R^2 = 0.65$, $p = 0.03$) linear relationship between biomass and tree DBH (Figure 5.7b).

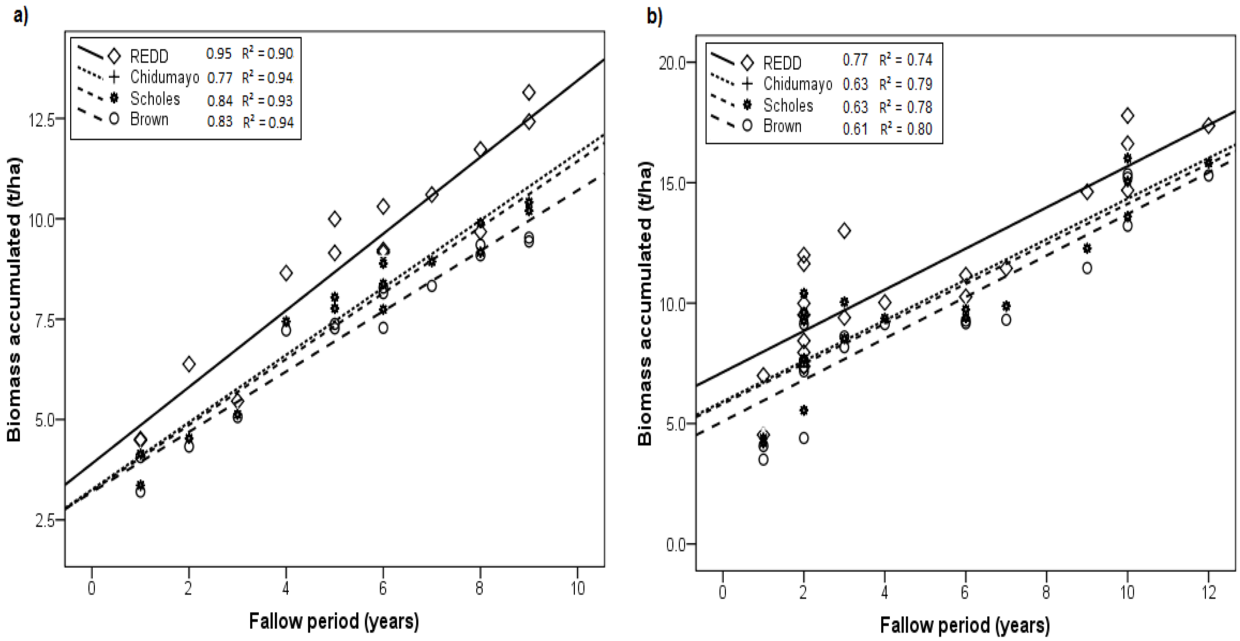


Figure 6.5. The relationship between above-ground biomass and fallow age, biomass estimates from four allometric equations a) *mutapo* soil and b) *bandate* soils.

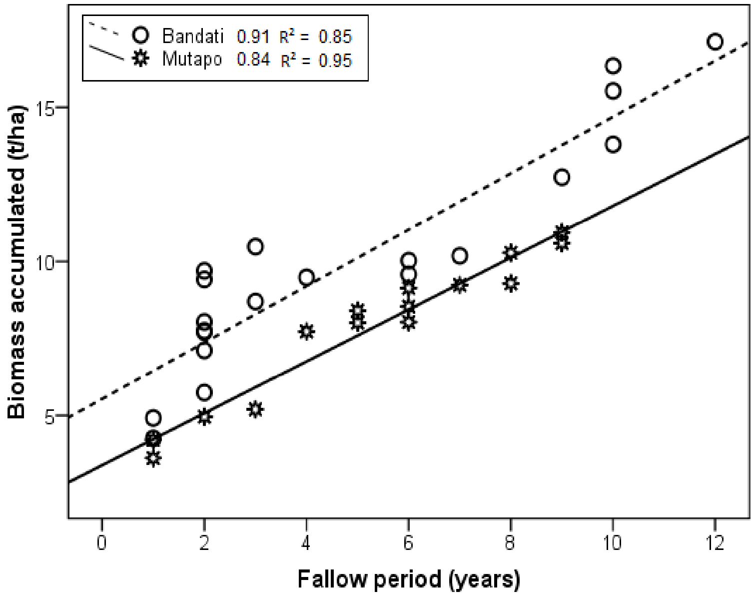


Figure 6.6. The relationship between above-ground biomass and fallow age using the mean biomass estimate of the four different allometric equations for *mutapo* and *bandate* soils.

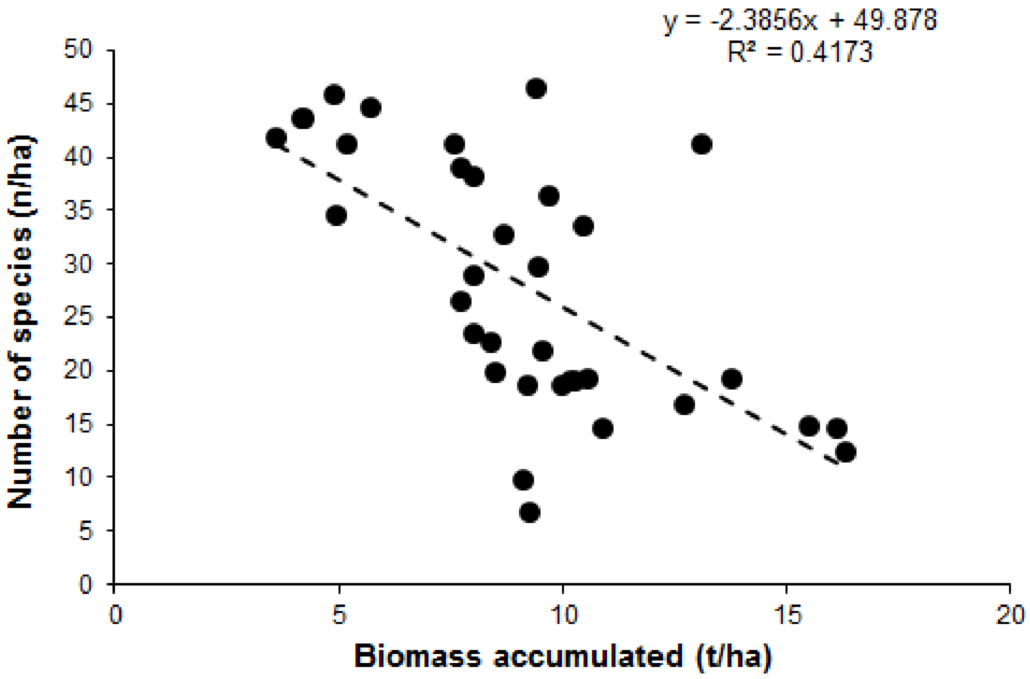


Figure 6.7. The relationship between the number of species and tree above-ground biomass.

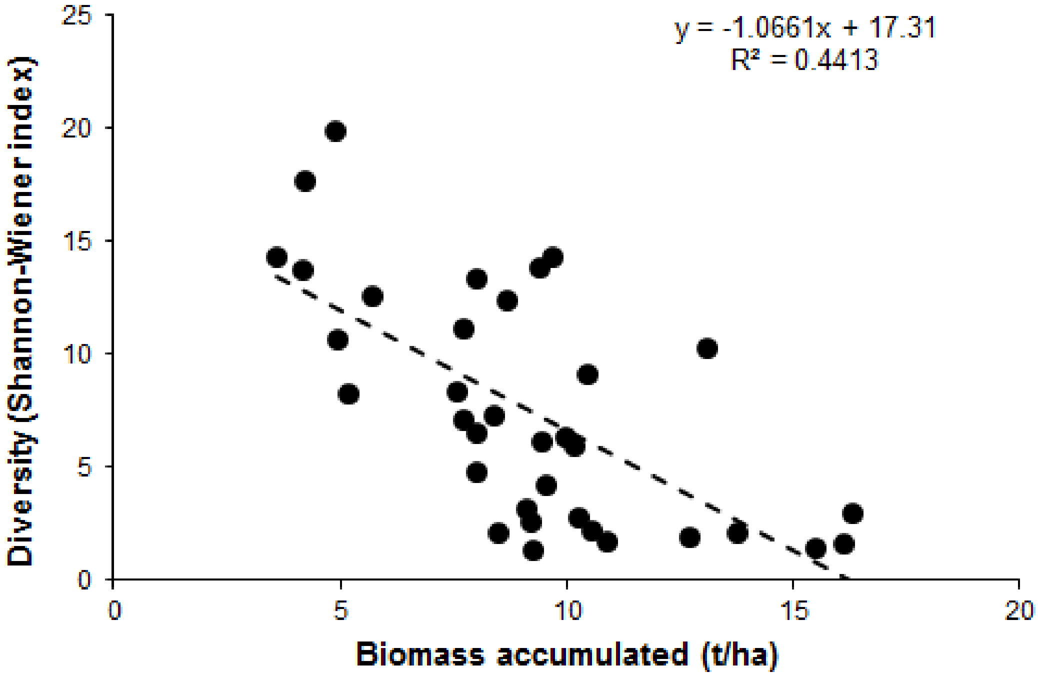


Figure 6.8. The relationship between the woody species diversity and tree above-ground biomass.

On both soil types a few species dominated the accumulated above-ground biomass (Appendix 3). In one year old fallows on *mutapo* soil, *C. mopane* and *C. elaeagnoides* had the highest amount (2.9 t ha^{-1} and 2.2 t ha^{-1} , respectively) of above-ground biomass followed by *A. tortilis subsp. spirocarpa* (1.9 t ha^{-1}). At two years, *A. tortilis subsp. spirocarpa* had the highest above-ground biomass, which increased rapidly in subsequent years until about the fifth year, after which the rate of increase slowed significantly (Figure 6.9). The early years of fallowing (1-3 years) on *bandate* soil had higher above-ground biomass of *C. eleaegnoides*, *L. cappassa* and *A. tortilis* subsp. *spirocarpa* shrubs. Above-ground biomass stocks only changed after the fourth year of fallowing. At this stage of fallowing *A. tortilis* subsp. *spirocarpa* had the highest amount of standing biomass. There was a rapid increase in *A. tortilis* subsp. *spirocarpa* above-ground biomass between the fifth and the seventh year, after which it continued to increase but at a slower rate (Figure 6.10). *L. cappassa* had the second highest total biomass, its biomass was constant between the first and the seventh year of fallowing, after which it increased but remained below that of *A. tortilis* subsp. *spirocarpa*.

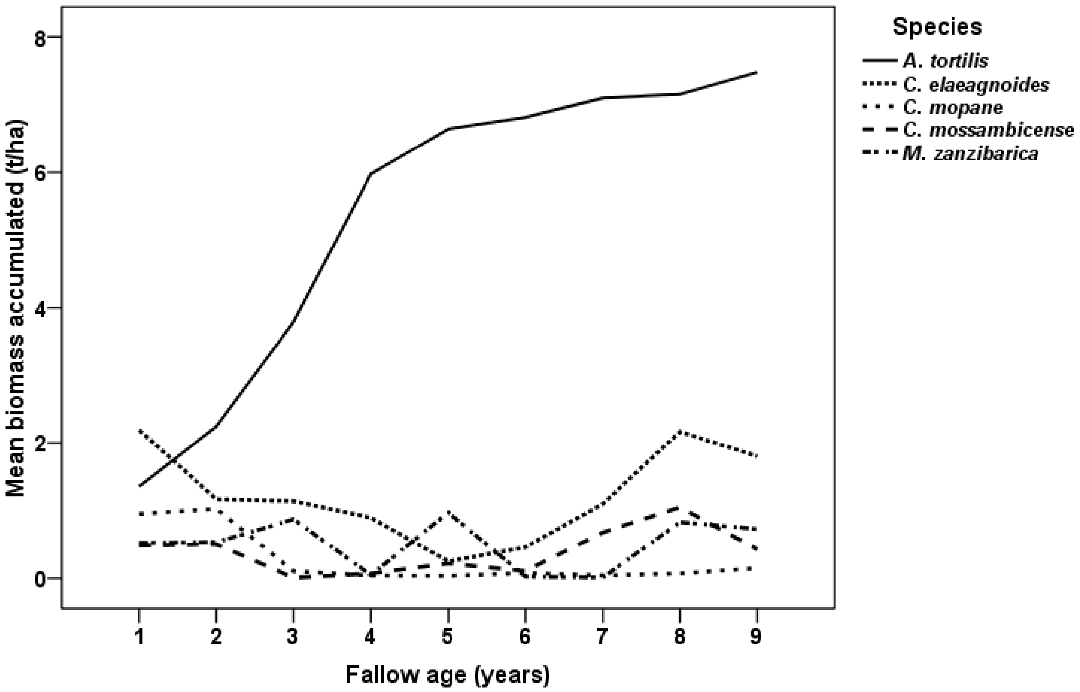


Figure 6.9. Above-ground biomass accumulation of five dominant woody species within fallows on *mutapo* soil.

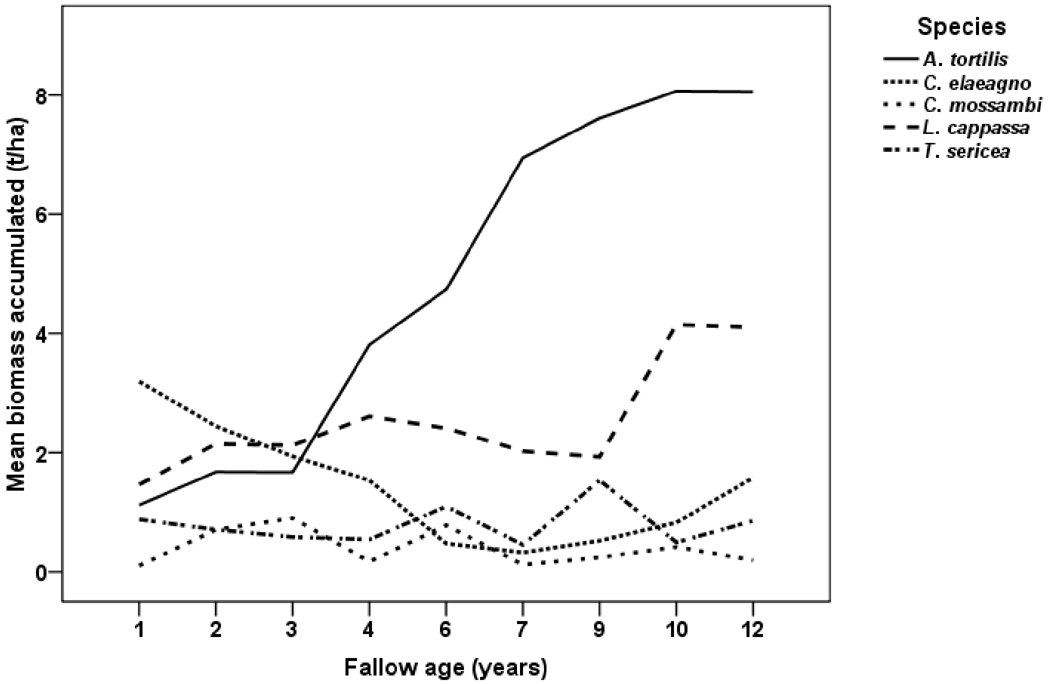


Figure 6.10. Above-ground biomass accumulations of five dominant species within fallows on *bandate* soil.

6.2. Discussion

The dynamics in diversity and structure of the vegetation arising through post-agricultural vegetation dynamics within fallows is an important indicator of ecosystem status after disturbance (Donfack *et al*., 1995). Results from this study indicate that early stages of the fallowing phase are characterized by high tree diversity, however, which is gradually lost with increasing fallow age. However, it is important at this stage to acknowledge the effect of other factors such as fire and grazing on fallow vegetation recovery. Fire and grazing are common phenomenon within agricultural landscapes in the study area particularly during the dry season. Observations confirmed by farmers in the area indicated that, the frequency and intensity of fire and grazing are different on the two soil types with both factors more pronounced on *bandate* soil than on *mutapo* soil.

These observations could be explained by the fact that on *bandate* soil, recovering fallow vegetation is characterised by a well-developed herbaceous stratum that is grazed upon by animals when green and also provides high fuel load for the fire when dry, whereas on *mutapo* soil the herbaceous stratum is very poor or absent in some areas (Timberlake *et al*. 1993; Tambara *et al*. 2012a). It is important to note that, the lack of information on the dynamics and effects of fire and grazing on recovering fallow vegetation is a major, but unavoidable, cause of uncertainty. In spite of the fire, grazing and other stresses that maybe more pronounced on *bandati* soil, results from this study indicate that, tree establishment and recovery occurs earlier and at a faster rate in fallows on *bandate* soil (4th year) compared to fallows on *mutapo* soil (6th year) and this result can be attributed to the high fertility levels of *bandate* soil and to the different properties of the two soil types as discussed by Mvuriye (2001).

Results from this study indicate that woody species diversity is high in the early years of the fallowing (1-2 years) but gradually decreases with increasing fallow age. This is in contrast to observations that were made elsewhere, Shagamu-Ijebu-Ode, South-western Nigeria by Aweto (1981); N'hambita community in Sofala Province, Mozambique by Williams *et al*. (2007), were woody species diversity within fallows areas would increase with age since abandonment of fallow and then saturate at older ages. The high species diversity observed within young fallows in this study can largely be attributed to re-sprout and reproduce from root suckers (Frost, 1996) and mature seed trees in field edges, natural vegetation patches and soil seeds banks act as a source of seeds for germination within recently abandoned areas (Peters and Neuenschwander, 1988; Augusseau *et al*., 2006). The agricultural landscape is characterised by a mosaic of different land units, each in a different phase of the clearance–cultivation–abandonment–recovery–clearance cycle, thus, abandoned areas are never more than fifty meters from other land-use units such as natural vegetation patches.

What is pertinent in this study is that, in spite of the transient returns of some secondary dominant woody species within the first year of fallowing, there is rapid penetration by *A. tortilis* subsp. s*pirocarpa* which was associated with higher rates of biomass accumulation particularly during the early years of fallow (1–5 years). Biomass accumulation rates were slightly higher on *bandate* soil than *mutapo* soil indicating high fertility of the sandy-clay alluvial *bandate* soils, an observation that was also highlighted by Mvuriye *et al*. (2001). The annual biomass accumulation rates recorded in this study (0.8 t ha^{-1} yr^{-1} on *mutapo* and 0.9 t ha^{-1} yr^{-1} on *bandate* soil) are higher than that recorded by Williams *et al*. (2007), 0.7 t ha^{-1} yr^{-1} in N'hambita, Sofala Province, Mozambique. These differences could be attributed to differences in soil type, climate, dominance of invasive species in the current study area and the limitations of using allometric equations (Pearson *et al*., 2005). Henry *et al*. (2011) highlighted that only 1% of tree species in the sub-Saharan Africa region have country specific allometric models for biomass estimation, thus, applicability of these allometric models outside the area from which they were developed remains to be tested (Frost, 1996). Therefore, results on biomass estimation have to be interpreted with caution.

Dominance of *A. tortilis* subsp. *spirocarpa* has also been observed in old (> 25 years) abandoned fields in the Sengwa Wildlife Research Area, South-west Zimbabwe (David Cumming, personal communication, November 6, 2012). *Acacia spp.*, as other leguminous trees (e.g. *L. cappass*a, *D. cinerea*), is known to form symbiotic relationships with nitrogen-fixing *Rhizobium* bacteria (Coates-Palgrave, 1995; Moustafa and Mansour, 2003). Thus, *A. tortilis* subsp. *spirocarpa* is able to improve soil fertility lost during the cultivation phase. This in turn allows *A. tortilis* subsp. *spirocarpa* to quickly adapt and grow rapidly with limited competition from other species. Added to the ability to fix nitrogen are characteristics which give *A. tortilis subsp*. *spirocarpa* a more advantages over other species, naturally it is a fast growing species, very hardy, tolerates salinity, water logging and extremely drought resistant (Vetaas, 1992; Moustafa and Mansour, 2003; Cramer *et al*. 2007).

This study expected that the woody species dynamics of recovering fallow vegetation will show a return towards the composition of the local natural woodland. Contrary to this expectation, results indicate that although there is higher similarity between older fallows and the natural woodland compared to any other stage in the fallowing period, the primary dominant species within woodlands such as *C. mopane* and *D. kirkii* were not the defining species of the older fallow vegetation. A similar result was also found by Williams *et al*. (2007). However, this result contradicts the findings of Aweto (1981); Brown and Lugo (1990); Guariguata and Ostertag (2000, 2001) which show that the characteristics of fallow vegetation become increasingly similar to that of mature forest. The study by Aweto (1981) was carried out in a humid region therefore, the contrasting results found in this study may confirm the differences in successional pathways between humid forest fallows and those in sub-humid regions. However, the differences noted here maybe due to the fact that 12 years of recovery observed in this study is not adequate to fully represent succession within these fallows, the time scale of the defining original vegetation of the area is unclear.

Based on results of this study, we deduce that vegetation dynamics within fallows do not follow the succession model as proposed by Clements (1916), which supposes that following disturbance a vegetation community will regenerate to a resemblance of itself by an orderly and predictable series of species replacements. In this system this model has proved to have limited applicability because of frequent human induced disturbances, such as fire and vegetation clearing for cultivation. Thus, vegetation dynamics in the mid-Zambezi valley are not characterised by single deterministic pathway of succession that result in a climax community with sets of species similar to the original vegetation. Instead, vegetation dynamics within fallows show long periods of *A. tortilis* subsp. *spirocarpa* dominance indicating a stable state. Therefore, vegetation dynamics in this area can best be explained by mechanism followed by the state transition model (Westoby *et al.*, 1989) which supposes that vegetation dynamics can be described by a set of discrete “states” of the vegetation on one piece of ground and a set of discrete “transitions” between states.

In this study the aspect of limited fallow age (12 years) is a major but unavoidable, cause of uncertainty on the prospect of vegetation dynamics after cultivation. This is a known weakness of the chronosequence method (Williams *et al.*, 2007). Another important aspect that, fire plays a significant role in the mid-Zambezi systems and the vegetation dynamics in these landscapes can only be fully understood and predicted within the context of fire disturbance. Therefore some caution is required in interpreting the results. Despite these limitations this study underlines the importance of further research to determine whether the *A. tortilis* subsp.

spirocarpa dominated vegetation within older fallows can be considered to have crossed a threshold to a new or different state, which could either be transient or stable?

SYNTHESIS

CHAPTER 7: Agriculture and biodiversity persistence within communally managed landscapes

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7.1. Introduction

Global deteriorating environmental conditions since the 1950s (Brown *et al.*, 2008) has led to scientific work aiming at testing whether there are optimal landscape conditions that enable the co-existence of biodiversity and agricultural production. Scientific work aimed at addressing this issue has been lacking in developing countries and the case of the mid-Zambezi valley, northern Zimbabwe presents a good example. In this area, programs such as the Mid-Zambezi Resettlement Development Project (MZRDP) were implemented based on information and research gathered on social-templates which only focus on addressing societal needs achieved through agriculture and utilisation of wildlife resources in the area. While, this provided people with livelihoods no attention was given to scientific research that would provide an ecological template which would enable the co-existence of biodiversity and agricultural production. An ecological-template would inform on, what landscape conditions are optimal for persistence of wildlife species in these communally managed landscapes, were agriculture is the dominant land-use.

In light of this knowledge gap, this thesis, focused on the ecological template by aiming to understand the dynamics of tree species diversity and vegetation productivity and recovering patterns in fallow areas. In addition the thesis focused on indicators of ecological health by testing the diversity and abundance of arthropods in these landscapes. The thesis was premised on the assertion that, ecosystems may move from one stable domain to another and persist in an altered configuration (Huggett, 2005). To this end, this study investigates some ecologically relevant questions, which will help in implementing an agricultural paradigm that can satisfactorily balance production outputs and biodiversity conservation within communally managed landscapes, i.e. ecoagriculture. If properly implemented the ecoagriculture paradigm can address several challenges faced in harmonising agricultural production and biodiversity conservation within tropical regions (Scherr and McNeely, 2008; Scherr *et al.*, 2012).

In this thesis, the main objective was to examine how the key features of agriculture in the mid-Zambezi valley affect biodiversity at two trophic levels, primary producers (woody species) and primary consumers (arthropods). The current study, therefore, aimed to provide base-line information on how biodiversity respond to agricultural ecosystem disturbance. This thesis, thus, fits within the context of a broader scientific debate on how to reach a balanced compromise between the ever-growing demand for agricultural products and biodiversity conservation.

7.1.1. Crop farming does not result in tree diversity loss in the mid-Zambezi valley

A key finding in this thesis is that, conversion of natural woodland areas to cropped landscapes does not affect tree beta and gamma diversity in the mid-Zambezi valley, at least in the short term. We observed this in both intensified and less intensified agricultural areas, a situation attributed to the presence of potential sources for tree regeneration that include stumps, live root stalks, soil seed banks, seed from trees left within field edges and from natural vegetation patches left within the agricultural landscapes. At field level tree diversity decreases with increasing cultivation period with a characteristic rapid decrease within the first few years of cultivation. Thus alpha diversity is affected in field cultivated for long periods which have repeatedly been subjected to vegetation clearing and ploughing before each planting season. This could potentially then affect tree gamma diversity in the area if long periods of cultivation are not interspaced with periods of fallow.

In this thesis, we also found that, the negative impact of woodland conversion and subsequent crop cultivation is on tree physiognomy and tree dominance. Woodland clearing has created patches of natural vegetation within an agricultural matrix dominated by shrubby vegetation. In a way, cultivation is turning woodland areas into agricultural field matrices characterized by trees in the form of saplings, regenerating stumps and mature trees on field edges and small patches of undisturbed natural vegetation. Given the importance of vegetation physiognomy and tree dominance in determining the type and availability of, food resources, foraging and nesting substrates, shelter from predators and physiological stress for other species (Walker, 2008), this result reflects negatively on the impact of agriculture on biodiversity at higher trophic levels. In light of these findings, there is need to create land units that promote tree and growth within cultivated landscapes.

Another important finding in this study is that there are sets of tolerated species within agricultural landscapes in the mid-Zambezi valley. As suggested by Augusseau *et al*. (2006), these tree species have the potential to contribute to the development of ‘‘agroforestry parklands’’ within agricultural landscapes. In the mid-Zambezi valley, tolerated species within cropped areas, can be used to as examples to farmers on how they can conserve the broader suite of tree species making up the natural biodiversity within agricultural landscapes.

7.1.2. Impact of woodland clearing and crop farming on arthropod diversity

In this study, findings show that diversity and abundance of fruit-feeding butterflies of the genus *Charaxes* and beetles of the subfamily Cetoniinae decreases due to natural woodland clearing and subsequent crop cultivation, therefore agricultural activities in the mid-Zambezi valley have a negative impact on arthropods. Moreover, results show that, negative impacts on arthropod taxa increase from areas with a mosaic of cultivated landscapes to homogeneous areas with high levels of cultivation intensity. We also found that, based on habitat preference, species within taxa under study, formed two broad groups: (1) generalist species, and (2) specialist species. Generalist species showed the ability to thrive in different habitats, from undisturbed woodlands to highly intensified cultivated areas while specialist species were confined to undisturbed woodlands with unique vegetation composition. Despite their extensive distribution, generalist species showed lower abundances in highly intensified agricultural areas.

7.1.3. Vegetation dynamics and above-ground biomass accumulation in fallows

The study found that dynamics in woody vegetation within fallow areas are characterized by recovery in vegetation features such as diversity, tree density and above-ground biomass, nevertheless, tree diversity only recovers during the early years of fallow then decrease in the latter years. This study also found that, recovery in tree diversity and above-ground biomass of vegetation in fallow areas is characterized by dominance of leguminous species such as *A. tortilis* subsp. s*pirocarpa* and they are no immediate indications that this invasive, easily raised from seed, very hardy and drought resistant species will be outcompeted by other species in the succession process. Being a leguminous tree, *A. tortilis subsp. spirocarpa* provides ecosystem services such as soil stabilization, rapid biomass accumulation and nitrogen fixation. Thus, given the current fallowing practices were fallows are cleared within a period of twelve years successional pathways will always be interrupted ensuring the continued dominance of *A. tortilis* subsp. *Spirocarpa* within fallow areas. Therefore, proper control and management of this species can bring new functions and services to the ecosystem which cannot be provided successfully by other native species (Moustafa and Mansour, 2003).

7.2. A summary of the findings

In this thesis the study investigated how biodiversity at two trophic levels, i.e., primary producers and primary consumers, respond to woodland clearing and crop cultivation in landscapes with varying cultivation intensity. The study also analyzed vegetation dynamics within fallow areas. Conclusions from this study are that, woodland clearing and subsequent crop cultivation does not result in decrease in tree beta and gamma diversity in the mid-Zambezi valley. Instead negative impacts of woodland clearing and subsequent cultivation occur at field level were diversity decrease with increasing cultivation period. Cultivation is also affecting vegetation physiognomy and changing tree dominance. Another conclusion from this study was that, conversion of natural woodlands to cropland is resulting in a decrease in arthropod diversity and abundance and this is more pronounced in intensively cultivated areas. The last conclusion from this study was that, vegetation recovery within fallows is characterised by dominance of *A. tortilis* subsp. *spirocarpa* with no indication on whether this species will be replaced by other species that dominate mature woodlands in the area.

7.3. Biodiversity within agricultural landscapes

What emanates from this thesis is the concept that there are ecological thresholds at which, the impact of agriculture on biodiversity may become obvious and after which marked changes in the pattern of occurrence of species occur. This was shown by noticeable changes in biodiversity within intensively cultivated areas compared to less intensified areas. In this context, any efforts to conserve biodiversity within agricultural landscapes in the mid-Zambezi valley may have to focus on implementing ecoagriculture, a paradigm that can satisfactorily balance production outputs and biodiversity conservation. There is need for further research to understand the specific impacts of different inorganic inputs such as pesticides, herbicides and fertilizers used by farmers on biodiversity. There is also need to establish the relationship between biomass accumulation rates and carbon sequestration rates within fallow areas and also assess the role of *A. tortilis* subsp. *spirocarpa* in soil organic carbon recovery. These could be the appropriate questions to hypothesize on for future research.

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APPENDIX 1

Woody species list in the study area (The positive sign (+) means present at that site)

Botanical name	**Family**	**Vernacular name**	**S_1**	**S_2**	**S_3**	**S_4**
Acacia nigrescens Oliv.	Fabaceae	Muungu, chinanga		+	+	
Acacia nilotica (L.) Willd. Ex Delile	Fabaceae	Mubayamhondoro		+	+	
Acacia robusta Burch.	Fabaceae	Munhunga	+		+	+
Acacia tortilis Engl.	Fabaceae	Muzunga	+	+	+	+
Adansonia digitata L.	Bombacaceae	Muuyu	+	+	+	+
Albizia brevifolia Schinz	Fabaceae	Musveva	+			
Albizia versicolor Welw. ex Oliver	Fabaceae	Mudzanga	+			+
Annona senegalensis Pers.	Annonaceae	Muroro				+
Azanza garkeana (Hoffm) Excell & Hillc	Malvaceae	Mutohwe		+	+	
Balanites aegyptiaca (L.) Delile	Balanitaceae	Mungezi	+			
Bauhinia petersiana Bolle	Fabaceae	Mupondo				+
Berchemia discolor (Klotzsch) Hemsley	Rhamnaceae	Munhacha		+	+	
Bolusanthus speciosus (Bolus) Harms	Fabaceae	Mubacha, munyati	+			+
Boscia mossambicensis Klotzsch	Capparaceae	Mugarahangaiwa		+		
Bridelia mollis Hutch.	Euphorbiaceae			+		
Burkea africana Hook.	Fabaceae			+	+	
Capparis tomentosa Lam.	Capparaceae	Karango	+			
Cassia abreviata Oliv.	Fabaceae	Mutemberembe		+	+	
Catunaregam spinosa (Hochst.) Keay	Rubiaceae	Chirobadunguru	+			
Citropsis daweana Swingle & Kellerman	Rutaceae	Mubayamhondoro	+			+
Colophospermum mopane Kirk ex Benth.	Fabaceae	Mupani	+	+	+	+
Combretum apiculatum Sonder	Combretaceae	Kagoro, kogoro				+
Combretum elaeagnoides Klotzsch	Combretaceae	Mubikanyemba	+	+	+	+
Combretum imberbe Wawra	Combretaceae	Muchenarota	+			+
Combretum molle R. Br. Ex G. Don	Combretaceae			+	+	
Combretum mossambicense (Klotzsch) Engl.	Combretaceae	Mubakwa	+	+	+	+
Combretum obovatum F. Hoffm	Combretaceae		+			
Combretum psidioides Welw	Combretaceae	Mupembere-kono			+	+
Combretum zeyheri Sonder	Combretaceae	Mugwakwa		+		
Commiphora ugogensis Engl.	Burseraceae			+		
Cordyla africana Lour.	Fabaceae	Mutondo	+			
Crossopteryx febrifuga (Afzel. Ex G. Don) Benth	Rubiaceae	Muteyo	+		+	+

Woody species list in the study area, Conti.

Botanical name	**Family**	**Vernacular name**	**S_1**	**S_2**	**S_3**	**S_4**
Dalbergia melanoxylon Guillemin & Perrottet	Fabaceae	Murwiti, muhati	+			+
Dichrostachys cineria (L.) Wight & Arn	Fabaceae	Mupangara		+	+	
Diospyros kirkii Hiern.	Ebenaceae	Muchenje	+			+
Diospyros qualoensis (Hiern) F. White	Ebenaceae	Kasvisva	+			+
Englerophytum magalismontanum (Sonder) Heine & J. H. Hemsley	Sapotaceae	Muhorongwa	+			
Erythrina livingstoniana Baker	Fabaceae	Muchirara	+			
Ficus sycomorus L.	Moraceae	Mukuyu		+	+	
Flacourtia indica (N. L. Burm.) Merr.	Flacourtiaceae	Mududwe, mutudza			+	
Friesodielsia obovata (Benth.) Verdc.	Annonaceae	Muchinga, munyani	+			+
Garcinia livingstonei T. Anders.	Clusiaceae	Munhinzwa		+	+	
Grewia flavescens Juss.	Tiliaceae	Chibubunu				+
Grewia monticola Sonder	Tiliaceae	Mubura	+			+
Karomia tettensis (Klotzsch) Vatke	Verbenaceae	Muparadzamusha	+			+
Kigelia africana (Lam.) Benth.	Bignoniaceae	Mumvee, mubveve	+	+	+	+
Kirkii acuminata Oliver	Simaroubaceae	Mutuva			+	
Lannea discolor (Sonder) Engl.	Anacardiaceae	Mushamba		+	+	
Leconiodiscus fraxinifolius Baker	Sapindaceae	Munharara, Mutarara	+			+
Lonchocarpus bussei Harms	Fabaceae			+	+	
Lonchocarpus capassa Rolfe	Fabaceae	Mupakasa	+		+	+
Maerua edulis Gilg & Gilg-Ben.	Capparaceae	Katunguru		+	+	+
Maerua prittwitzii Gilg & Benedict	Capparaceae	Maporoatsindi		+	+	
Markhamia zanzibarica (Klotzsch) K. Schum.	Bignoniaceae	Mubikanyemba	+	+	+	+
Gymnosporia senegalensis (Lam.) Excell	Celastraceae	Mugaranjiva				+
Ormocarpum kirkii S. Moore	Fabaceae	Kapurupuru				
Ozoroa reticulata (E. G. Baker) R. & A. Fernandes	Anacardiaceae	Chibvukati	+	+		
Piliostigma thonningii (Schumacher.) Milne-Redh.	Fabaceae	Musekesa		+		
Pseudolachnostylis maprouneifolia (T. R. Sim) Verdc.	Phyllanthaceae	Mutsonzowa	+		+	+
Pterocarpus brenanii Barbosa & Torre	Fabaceae	Muomba			+	
Pterocarpus rotundifolius (Sonder) Druce	Fabaceae	Muchirara		+	+	
Schrebera trichoclada Welw.	Oleaceae	Muhakata	+		+	+

Woody species list in the study area, Conti.

Botanical name	**Family**	**Vernacular name**	**S_1**	**S_2**	**S_3**	**S_4**
Sclerocarya birrea (A. Rich.) Hochst.	Anacardiaceae	Mutsomo	+	+	+	
Securidaca longipedunculata (Fresen.)	Polygalaceae	Mufufu, Kasakwe,		+	+	
Senna singueana (Delile) Lock.	Fabaceae	Munzungu		+	+	
Sterculia africana (Lour.) Fiori	Sterculiaceae	Murere		+	+	
Strychnos spinosa (Lam.)	Loganiaceae	Mutamba	+			
Tamarindus indica L.	Fabaceae	Musinga	+		+	+
Terminalia brachystemma Welw	Combretaceae	Mukononomukuru		+	+	
Terminalia prunioides C. Lawson	Combretaceae	Nyakasanza			+	
Terminalia sambesiaca Engl. & Diels	Combretaceae				+	
Terminalia sericea Burch. Ex DC.	Combretaceae	Mukonono-mudiki			+	
Terminalia stulhmannii Engl.	Combretaceae	Mukonono-mudiki		+		
Vangueria infausta Burch	Rubiaceae	Munzviru			+	
Xanthocercis zambesiaca (Baker) Dumaz-le Grand	Fabaceae	Mutsvanzva	+			+
Xeroderris stulhmannii (Taub.) Mendonca & E. P. Sousa	Fabaceae	Mupururu	+	+		
Ximenia caffra Sonder	Olacaceae	Munonde	+	+		
Xylia torreana Brenan	Fabaceae	Mupondo jesse			+	
Ziziphus mauritiana Lam.	Rhamnaceae	Musawu	+	+	+	+
Ziziphus mucronata Willd.	Rhamnaceae	Muchecheni		+		
Total	**32**		**39**	**38**	**43**	**42**

APPENDIX 2

Above-ground biomass estimates from four allometric equations

Soil	Age	Chidumayo (1997)	Brown (1989)	Chidumayo (1997)	REDD (2012)	Mean biomass (t/ha)	Standard deviation
Mutapo	1	3.42	3.36	3.21	4.51	3.62	0.60
	1	4.17	4.14	4.06	4.48	4.21	0.19
	2	4.60	4.53	4.33	6.38	4.96	0.95
	2	4.55	4.23	3.99	5.20	4.49	0.52
	3	5.15	5.13	5.06	5.46	5.20	0.18
	3	4.91	4.71	4.03	5.10	4.69	0.47
	4	6.11	5.71	5.03	7.01	5.97	0.83
	4	7.62	7.44	7.22	8.65	7.73	0.63
	5	7.88	7.77	7.27	9.15	8.02	0.80
	5	8.21	8.05	7.38	9.99	8.41	1.12
	6	7.90	7.74	7.28	9.22	8.04	0.83
	6	9.08	8.89	8.29	10.30	9.14	0.85
	6	8.46	8.38	8.15	9.16	8.54	0.44
	7	9.08	8.93	8.33	10.60	9.24	0.97
	8	10.15	9.90	9.35	11.73	10.28	1.02
	8	9.22	9.17	9.09	9.67	9.29	0.26
	9	10.32	10.20	9.43	12.42	10.59	1.28
	9	10.67	10.41	9.53	13.15	10.94	1.55
Bandate	1	4.77	4.39	3.51	6.99	4.92	1.48
	1	4.24	4.19	4.07	4.53	4.26	0.20
	2	5.66	5.55	4.40	7.38	5.75	1.23
	2	9.28	9.27	9.13	9.99	9.42	0.39
	2	7.61	7.61	7.61	7.61	7.61	0.00
	2	9.60	9.61	7.57	12.00	9.70	1.81
	2	7.74	7.67	7.17	8.44	7.75	0.52
	2	7.64	7.67	7.34	9.51	8.04	0.99
	3	10.90	10.41	9.53	11.64	10.62	0.88
	3	8.71	8.53	8.17	9.40	8.70	0.51
	3	10.24	10.05	8.61	13.01	10.48	1.84
	4	10.40	9.71	8.30	10.58	9.75	1.04
	4	9.02	8.71	7.95	10.26	8.98	0.96
	4	9.40	9.37	8.13	10.03	9.23	0.79
	6	9.50	9.39	8.92	10.26	9.52	0.56
	6	9.93	9.74	9.03	11.16	9.96	0.89

Above-ground biomass estimates from four allometric equations, Conti.

Soil	**Age**	**Chidumayo (1997)**	**Brown (1989)**	**Chidumayo (1997)**	**REDD (2012)**	**Mean biomass (t/ha)**	**Standard deviation**
Bandate	7	10.09	9.88	9.31	11.45	10.18	0.91
	9	12.60	12.28	11.46	14.62	12.74	1.34
	10	16.26	16.01	15.35	17.79	16.35	1.03
	10	13.72	13.59	13.21	14.69	13.80	0.63
	10	15.25	15.05	15.20	16.62	15.53	0.73
	12	16.07	15.82	15.30	17.37	16.14	0.88

APPENDIX 3

Above-ground biomass accumulation for ten species on mutapo and bandati soil

Soil type	**Age**	***A. tortilis***	***A. versicola***	***C. mopane***	***C. elaeagnoides***	***C. mossambicense***	***D. cinerea***	***D. qualoensis***	***G. monticola***	***Z. mauritiana***	***M. zanzibarica***
Mutapo	1	1.36	0.06	0.95	2.19	0.16	0.08	0.59	0.49	0.21	0.52
	2	2.24	0.08	1.03	1.17	0.19	0.04	0.61	0.50	0.09	0.53
	3	3.79	0.00	0.10	1.14	0.01	0.05	0.03	0.01	0.07	0.87
	4	5.97	0.11	0.04	0.90	0.14	0.03	0.14	0.07	0.33	0.05
	5	6.99	0.03	0.04	0.25	0.48	0.06	0.09	0.22	0.05	0.97
	6	7.45	0.28	0.08	0.46	0.15	0.04	0.04	0.11	0.03	0.02
	7	7.10	0.08	0.04	1.10	0.18	0.01	0.02	0.68	0.01	0.01
	8	5.15	0.09	0.07	2.17	0.94	0.06	0.11	1.05	0.13	0.83
	9	7.47	0.07	0.15	1.80	0.30	0.02	0.12	0.43	0.41	0.73
Bandate		***A. tortilis***	***A. garkeana***	***L. cappassa***	***C. elaeagnoides***	***C. mossambicense***	***T. sericea***	***S. birrea***	***D. cinerea***	***D. kirkia***	***X. torreana***
	1	1.12	0.88	1.47	3.19	0.11	0.10	0.51	0.08	0.13	0.27
	2	1.68	0.70	2.15	2.44	0.71	0.41	0.52	0.30	0.87	0.73
	3	1.67	1.17	2.13	1.94	0.90	0.59	0.90	0.21	0.09	0.21
	4	1.81	0.09	2.61	1.54	0.17	0.55	0.35	0.10	0.06	0.63
	6	4.74	0.10	2.41	0.48	0.78	1.10	0.09	0.06	0.04	0.04
	7	6.95	0.45	2.03	0.33	0.12	0.00	0.07	0.10	0.10	0.56
	9	8.61	0.17	1.93	0.52	0.24	1.54	0.41	0.07	0.13	0.52
	10	8.07	0.35	4.15	0.83	0.42	0.20	0.05	0.13	0.03	0.65
	12	8.05	0.11	4.10	1.58	0.19	0.86	1.12	0.11	0.02	0.07