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Assessing the resilience of West African parklands using a trait-based approach

Samuel Morel
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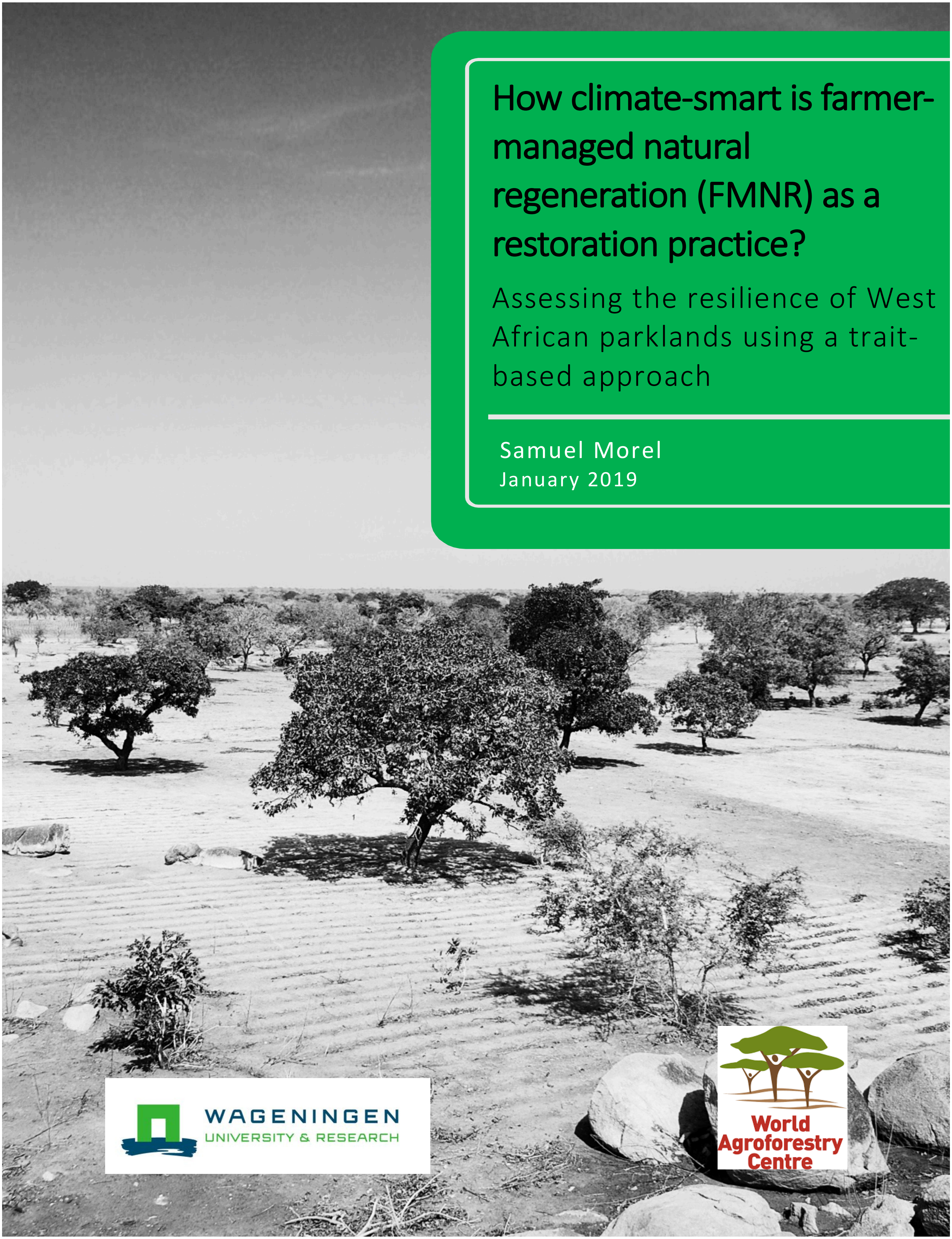

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Abstract

Recently, widespread regreening of some parts of the Sahel took place because farmers have allowed the regeneration of young trees in their fields, a practice known as farmer managed natural regeneration (FMNR). FMNR is seen as a promising method to restore degraded landscapes while improving rural livelihoods, yet little is known about the factors influencing the regeneration dynamics and diversity of woody species on which FMNR practices depend. Insight from the field of functional ecology could help to guide the restoration of agroecosystems that are resilient in the face of erratic climate conditions. In an agricultural landscape of Burkina Faso, I evaluated changes in species composition, diversity and functional properties (weighted-mean trait values and functional diversity) between the established and the regenerating tree community. Using the same parameters, I also explored the response of the regenerating vegetation to different land use types and farming practices. My results show that regeneration is dominated mainly by drought tolerant shrub species at the expanse of large tree species that characterize the established community. Higher land use intensity had a negative effect on regeneration diversity and favored drought tolerant vegetation. Diversity of the regenerating vegetation was affected positively by fire and selective tree cutting but negatively by livestock. My findings highlight that current land use management in west African parklands hampers diversification of the regenerating vegetation and by relying solely on the natural regeneration of trees, FMNR alone might not be able to restore multifunctional and resilient ecosystems. The maintenance of land units that promote the regeneration of multiple species, in particular those of socio-economic value for local communities, should be a priority. When natural regeneration is failing, enrichment planting and domestication of key species should be considered.

Keywords: *land degradation, restoration, FMNR, agroforestry, West Africa, functional traits, climate resilience*

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1. Introduction

1.1. Parklands in Sub-Saharan West Africa

Agroforestry parklands are a widespread land-use system of sub-tropical savannahs where farmers grow crops in combination with valuable native trees and shrubs retained when establishing crop fields (Pullan 1974). Parklands systems fulfil major functions that support over 40 million people in the semi-arid lands of West Africa (Boffa, 1999). They cover a range of the rural needs, from wood for fuel and building material to medicine and various food commodities, including cooking oil and fodder for livestock (Boffa, 2000). Parkland trees are also an important source of income for poor rural population, especially for women. Lamien and Vognan (2001) have shown for example that the African shea tree *Vitellaria paradoxa* contributed to 27% of the annual income of rural women in Burkina Faso. Agroforestry parklands also deliver regulatory environmental services that are essential for sustaining food production such as soil nutrient cycling, water regulation as well as preserving biodiversity (Garrity et al., 2010). Despite their role in providing important environmental goods and services, West African parklands have been suffering severe degradation over the last fifty years resulting in a significant decline in tree density and diversity, as a consequence, parklands are now dominated by old trees while young trees and regeneration are scarce or absent (Kindt et al., 2008; Wezel & Lykke, 2006). This decline is attributed to various reasons. Frequent droughts and increasing temperatures associated with climate change in the second half of the 20th century could be causing tree cover change in the Sahel (Gonzalez at al., 2012). In addition, the growing population has led to an increased demand for tree products and to agricultural intensification, characterized by a reduction of the fallow period and longer continuous cultivation on the same land, thus hampering the regeneration of woody species (Nikiema, 2005). Additionally, unfavourable agricultural and forest policies might also be a cause of forest cover loss in parklands systems (Boffa, 2000). For instance, farmers need a permit from the forestry service in most Sahelian countries if they want to prune or harvest trees that are naturally regenerating on their field (Reij & Garrity, 2016). The loss of ecosystem goods and services associated with deforestation and land degradation is worrying in a region where millions of farming households practice rain-fed agriculture and rely on local natural resources for their livelihoods (Funk et al., 2009).

1.2. Farmer managed natural regeneration as a restoration practice

In order to stop land deterioration and enhance livelihood conditions in the region, improved agroforestry systems like farmer-managed natural regeneration (FMNR) have been promoted by environmental and developmental NGOs in the last three decades (Birch et al., 2015). First developed in southern Niger in 1983, FMNR is the practice of pro-actively nurturing new stems regenerating from previously felled trees' stump (Rinaudo, 2007). Farmers select and prune the strongest and straightest stems to promote their growth while culling competing shoots. The techniques are flexible as farmers can adapt them to their own context and purpose (Garrity et al., 2010). The success of FMNR also resides in its low cost since it requires no special input, in contrast to enrichment planting where seedling nurseries and transport are needed (Weston et al., 2015). Recent studies have shown that improved agroforestry parklands have emerged in Sahelian countries like Niger and Burkina Faso, where tree cover recovery has taken place on 5 million and 300'000 hectares respectively, thanks to the adoption of soil and water conservation measures and FMNR practices, combined with an enabling policy environment (Reij, 2009). Although increasing density of woody vegetation recorded in these regions is seen as the result of protecting and nurturing seedlings, data on the abundance and diversity of the regenerating plant communities and their drivers are scarce. Studies have reported that the recent increase in tree cover in the Sahel originate mainly from a few regenerating species, in the *Combreataceae* family and the *Caesalpiniaceae* family in particular (Gonzalez et al., 2012). This shift in species composition has considerable consequences for the success of restoration initiatives that rely on the natural regeneration of trees like FMNR and further exploration on the drivers of natural regeneration is necessary.

On-farm tree diversity dynamics depend on a multitude of natural and anthropological factors (Bayala et al., 2011). The presence of seed dispersal vectors as well as the landscape matrix will determine the composition, density and spatial arrangement of the regenerating tree community by directly affecting the replenishment of the seed bank (Ordonez et al., 2014). The presence and abundance of root stock from which species can resprout is equally important, especially in the context of FMNR. After they have germinated, seedlings and saplings survival is a strong bottleneck that can limit natural regeneration in agricultural areas (Vieira & Scariot, 2006). Changes in human activities (e.g. conversion between land use types and changes in disturbance regimes, such as wood cutting, fire or grazing) may also directly or indirectly affect the regeneration of species. Fire is a common practice to clear lands in West African parklands, and while its effect on vegetation dynamics depends on the intensity and the frequency as well as the response of plant species to burning, studies have found that annual early fire significantly reduces species richness and density of saplings in savanna woodlands (Nikiema, 2005; Zida et al., 2007). Browsing is another major anthropogenic disturbance in sub-tropical savannas, where local people rely on livestock rearing for their livelihood (Savadogo, 2007). It is often assumed that heavy browsing and grazing by mammals can hamper the development of woody vegetation while no grazing improve regeneration of tree species (Luoga et al., 2002). However, livestock can play an important role in the dispersion and germination of seeds, therefore increasing species richness at a given site (Razanamandranto et al., 2004). Selective tree cutting for fuelwood and poles is also a common practice in West Africa and can influence vegetation structure in various ways. In arid and semi-arid areas, opening of the canopy can result in unfavorable thermal condition in the understory promoting the growth of drought-tolerant species (Savadogo, 2007). Studies have shown that removal of trees can also increase grass production, leading to more competition with woody regeneration (Gambiza et al., 2000). Additionally, most savanna tree species will produce sprouts and suckers after removal of above-ground biomass, potentially increasing seedling density but decreasing species diversity due to competition processes (Zida et al., 2007).

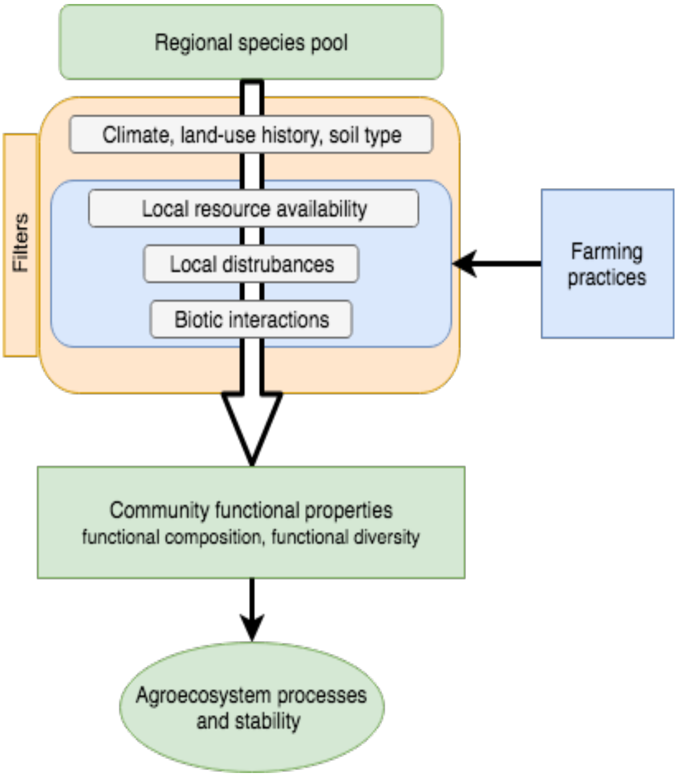


Figure 1. *Conceptual framework of this study (modified from Damour et al., 2017). Interconnected environmental and anthropogenic (including farming practices) filters determine the functional properties of the regenerating plant communities. The composition and* diversity of response and effect traits will in turn affect agroecosystem processes and its stability in the face of environmental fluctuations.

Through their different practices, farmers shape the tree community not only in terms of species composition, density and diversity but also in terms of functional composition (FC) and functional diversity (FD) (Damour et al., 2017). In the next paragraph, I explain how the resulting functional properties of the plant community modulate the agroecosystem processes (mainly through effect traits) and stability (mainly through response traits) (Figure 1.) (Garnier & Navas, 2012).

1.3. The role of functional diversity in ecosystem functioning and resilience, and its relevance for restoration

It is widely acknowledged that biodiversity plays an important role in ecosystem functioning, as much in natural systems as in agroecosystems (Altieri, 1999; Hooper et al., 2005). To date, most studies of biodiversity-ecosystem functioning have used taxonomic species richness as a diversity metric (Balvanera et al., 2006). However it is increasingly recognized that the diversity of functional traits has a greater influence on ecosystem functioning than does the taxonomic species richness and diversity (Díaz & Cabido, 2001). This is because functional traits allow to mechanistically link change in species composition with ecosystem processes (Cadotte et al., 2011). Traits are defined as 'any morphological, physiological or phenological feature measurable at the individual level, from the cell to the whole-organism level, without reference to the environment or any other level of organization'(Violle et al., 2007). An underlying principle of trait-based ecology is that (i) environmental and biotic filters determine which species from the regional species pool can survive by selecting 'response' traits that promote fitness in the given environmental conditions and (ii) in turn, the resulting community assemblage will influence ecosystem function according to the distribution of 'effect' traits within the community (Suding et al., 2008). From an applied perspective, trait-based approaches have gained a recent interest in many applied fields including conservation (Freitas & Mantovani, 2017), restoration ecology (Laughlin, 2014; Lohbeck et al., 2017) and agroecology (Garnier & Navas, 2012; Martin & Isaac, 2015). According to Martin & Isaac (2018) "the transition of trait- based ecology into resource management fields was facilitated by clear empirical evidence that plant functional traits, and functional trait diversity are strong mechanistic predictors of ecosystem functioning". In particular, the variation in responses to disturbances and environmental changes within a community, called 'response diversity' (Elmqvist et al., 2003), is thought to play a key role in ecosystem resilience (Mori et al., 2013). This can be explained using the concept of insurance hypothesis, suggesting that a great diversity of response traits among species contributing to the same ecosystem function allows the stabilization of this function in the face of environmental change, through compensatory dynamics (Loreau, 2000).

In light of the above, it is recommended that strategies to restore degraded land for the benefits of the environment and the people are informed by principals of functional ecology (Lohbeck et al., 2018). Forward-thinking restoration goals should seek to restore functional ecosystems that are adapted to predicted environmental conditions in order to ensure sustaining the ecological services on which humans depend (Laughlin, 2014). Recent synthesis raised concerns that tree mortality attributed to drought and heat stress could increase around the world as a consequence of climate change (Allen et al., 2010). This is especially relevant in West Africa, where model simulations predict that winter temperatures will increase by 2–3°C in the twenty-first century, accompanied by an increased frequency of heat waves (Dike et al., 2015). On the other hand, trends in rainfall patterns are not as uniform as the ones in temperatures (Sultan & Gaetani, 2016). Instead, extreme climatic events are forecasted in future decades, characterized by intense rainfalls concentrated in less rainy days and droughts that set in quicker and more intensely (Salack et al., 2016). In order to cope with the fluctuations of future climate and ensure stability, future tree communities should reflect species that are tolerant to higher temperatures and drought spells (Gonzalez et al., 2012). Based on the climatic predictions mentioned above, it is possible to make prediction on which functional traits will likely promote fitness in the given abiotic conditions using empirical traits-environment relationship (Laughlin et al., 2017). For instance, studies have shown that larger trees are more prone to drought-induced death than smaller ones in a variety of savanna ecosystems (O'Brien et al., 2017). Similarly, heat and water stress tend to select plants with relatively smaller leaves (Pérez-Harguindeguy et al., 2013). On the other hand, thick leaves lower the risk to extreme temperatures and can enhance long-term survival for perennial desert flora (Leigh et al., 2012). Leaf compoundness is another strategy to avoid high insolation and water loss since compound-leaved species have the ability to fold their leaflets during heat stress (e.g. at noon or during the dry season) (Lohbeck et al., 2015). Wood density is also a good indicator of resistance to drought as high wood density is usually associated with higher survival in dry environment (Cornelissen et al., 2003). In this study, the functional property of the vegetation was characterized by selecting nine response traits known to affect survival in dry environments (Table 1).

Table 1. *List of plant functional traits related to water and heat stress used in this study. Some traits were also selected because they are correlated with other traits harder to measure (e.g. plant height is in indicator of rooting depth and Specific Leaf Area a strong predictor Relative Growth Rate, see Appendix 1).*

Functional trait	Description	Related function	Reference
Maximum adult height; Hmax (m)	Distance (m) between the superior limit of the main photosynthetic tissues of a plant and the soil level	Trade-offs between maximum height and tolerance to environmental stress, proxy for root length	(Cornelissen et al., 2003; O'Brien et al., 2017)
Deciduousness; De (binary)	Ability to shed leaves seasonally	Drought avoidance	(Poorter & Markesteijn, 2007)
Leaf area; LA (cm^2)	One-sided area of an individual leaf	Trade-offs between leaf size and tolerance to heat and water stress	(Pérez-Harguindeguy et al., 2013)
Specific leaf area; SLA (m^2 kg^{-1})	One-sided area of a fresh leaf divided by its oven-dry mass	Correlated with potential relative growth rate (RGR)	(Lopez-Iglesias et al., 2014)
Leaf thickness; LT (mm)	Thickness of the leaf	Physical strength, leaf lifespan, resistance to heat	(Leigh et al., 2012; Pérez-Harguindeguy et al., 2013)
Leaf dry matter content; LDMC (g g^{-1})	Oven-dry mass of a leaf divided by its water-saturated fresh	Physical strength, tolerance against water limitation and heat	(Cornelissen et al., 2003; Poorter &

	mass		Markesteijn, 2007).
Leaf compoundness; LP (binary)	Leaf constituted of several or many distinct parts (leaflets) joined to a single stem	Thermoregulation and control of water loss	(Yate et al., 2010).
Wood density; WD (g cm^3)	Oven-dry mass of a section of the main stem of a plant divided by the volume of the same section, when still fresh	Drought survival, correlated with RGR	(Bucci et al., 2004; Hacke et al., 2001; Mitchel et al,. 2008)
Twig dry matter content; TDMC (g g^{-1})	Oven-dry mass of a terminal twig, divided by its water-saturated fresh mass	Correlated with wood density, drought resistance	(Cornelissen et al., 2003).

In an agricultural landscape of South-Central Burkina Faso, I evaluate to what extent FMNR can contribute to maintaining future tree biodiversity by measuring how the current regenerating trees (<5 cm DBH) differs from the established older trees (>5 cm DBH) in terms of species composition (SC), taxonomic diversity (TD) as well as functional composition (FC) and functional diversity (FD). I then look at how the regenerating vegetation responds to different land use types and regimes of farming practices by analyzing changes in TD, FD and FC. In the study site, I identified six different land-use types that differ in term of intensity of use and human pressure: home field (HF; 'champs de case'), bush fields (BF; 'champs de brousse'), short fallow (SF), long fallow (LF), protected area (PA) and the buffer zone (BZ) of the protected area (Table 2). I selected three different farming practices: burning, livestock browsing and wood cutting

I hypothesized that: (i) TD and FD will be lower in the regenerating community compared to the established community. I also expect the regenerating vegetation to be dominated species from the *Combreataceae* and Caesalpiniaceae family, and to be more represented by functional traits that confer resistance to heat and drought compared to the established community. (ii) TD and FD of the regenerating vegetation will be lower in land-use types where the human pressure is strong like BF, HF and SH but higher in less disturbed land-use type like LF, PA and BZ. Additionally, increasing land use intensity change local conditions to a more arid state and therefore the regenerating tree community in these land use type will have trait values that indicate higher resistance to drought (iii) farming practices will have a negative effect on the TD and FD of the regenerating community and increased intensity of farming practices will shift regenerating community towards more drought-resistance strategies.

Considering the factors that affect the vulnerability of tree communities to future climatic disturbances is essential to evaluate and optimize the long-term success of FMNR. Through this research, I hope to contribute evidence to support and scale up the adoption of FMNR as an effective technique to restore multifunctional and resilient ecosystems.

Table 2. *List of land use types identified in this study and their associated level intensity*

Land-use type	**Description**	**Intensity of use**
Home field (HF)	Field in the immediate vicinity of homesteads, continuously under cereals cultivation (millet, sorgho and maize), manure and compost as main source of amendment	HIgh
Bush field (BF)	Fields where cash crops like groundnut and cotton are usually grown, fertility restored mainly thanks to fallow periods	High

Short fallow (SF)	Two to five years old fallow, livestock often herded there during the cropping season	Intermediate
Long fallow (LF)	Ten to twenty years old fallow, livestock often herded there during the cropping season	Intermediate
Buffer zone (BZ)	Edge of the protected forest area where pastoralism and wood cutting is occurring	Low
Protected forest (PF)	Protected forest vegetation represented by the National Park Kaboré Tambi	Low

2. Methods

2.1. Field sites

This study took place in the province of Zoundweogo in the south-central region of Burkina Faso between latitude 11°25' and 11°45' North and longitudes 1°20' and 1°84' West (Figure 2). The site is located in the Sudanian ecological zone with a rainy season lasting for 4 to 5 months (May – September) while the dry season lasts from October to April. The annual rainfall varies between 800 and 1000 mm (Nikiema, 2005). The soil is characterized as ferruginous leached, overlaying clayey-sandy to sandy-clayey material (FAO, 2015). The natural vegetation in the area is typically of a Sudanian wooded savanna with species like *Detarium microcarpum*, *Pterocarpus erinaceus*, *Terminalia sp*, and *Burkea africana*, as represented in the National Park Kaboré Tambi (Nikiema, 2005). The anthropogenic landscape is characterized mainly by extensive farmlands with woody cover dominated by useful tree species retained by farmers, such as *Vitellaria paradoxa* (Shea tree), *Parkia biglobosa* (Néré tree), *Lannea microcarpa, Sclerocarya birrea,* and *Tamarindus indica*. The majority of the population is engaged in subsistence agriculture, pearl millet (*Pennisetum glaucum*) and sorghum (*Sorghum bicolor*) being the dominant crops. Some farmers grow cash crops as well, mainly groundnuts (*Arachis hypogaea*) and cotton (*Gossypium sp*). Other activities include pastoralism, sometime resulting in conflicts between herders and farmers due to the damage to crops and trees caused by the livestock (Petit, 2003). FMNR has been promoted widely amongst smallholders in the region by the West Africa Forest-Farm Interface program (WAFFI), a project led by the Center for International Forestry Research (CIFOR), in collaboration with the World Agroforestry Centre (ICRAF) and Tree Aid.

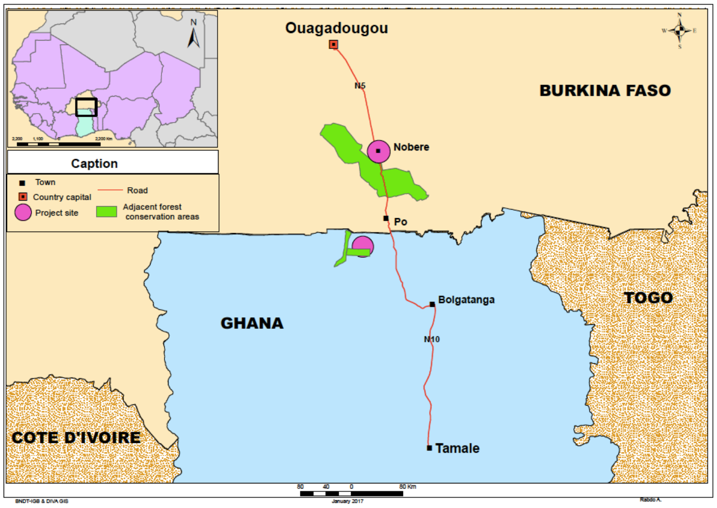


Figure 2. *Study site. Souce: WAFFI*

2.2. Data collection

2.2.1. Farming practices and vegetation survey

Data on land uses, farming practices and vegetation were retrieved from a field survey carried in March 2017. The survey used the Land Degradation Surveillance Framework (LDSF), designed to provide key indicators of ecosystem health like vegetation cover, structure and floristic composition, signs of disturbances, land-use history and soil characteristics (Vågen et al., 2013). The LDSF is built around a hierarchical sampling framework using sites that are 100 km^2 (10x10km). In each a site, there are sixteen basic sampling units called “clusters” which consists of 10 “plots” measuring 1000m^2, randomized around each cluster center-point (Figure 3). Each plots contains four 100m^2 “subplots”. Land-use categories were assessed visually and confirmed by the farmer who owned and managed the land. Because the LDSF site is selected at random across a region, the number of plots in each land-use type was unequal (PA=10, BZ=30, HF=60, BF=24, SF=13 and LF=23). The regime of fire, browsing, and tree cutting was also determined visually by the proportion of the plot impacted and ranked on a scale from 0 to 3. A vegetation inventory was carried out in each plot in order to sample every woody species whose diameter at breast height (DBH) was equal or larger than 5 cm. Natural regeneration was quantified by sampling individuals whose DBH was smaller than 5 cm in one sub-plot of 20m² located at the center of each plot.

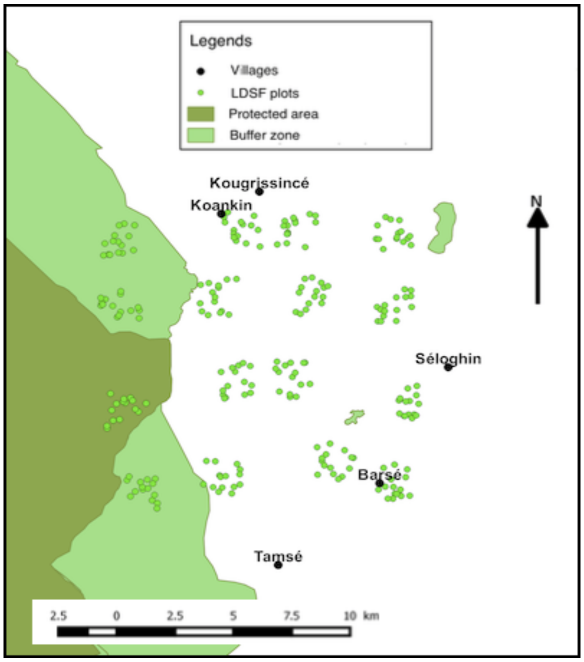


***Figure 3**: Distribution of the LDSF plots in the study site.*

2.2.2. Functional traits selection and measurement

Using the LSDF inventory, I calculated the relative abundance of seedlings species and the relative basal area of adult tree species in every plot of the two sites. Seedlings species that made up at least 80% of the abundance and adult trees species that made up at least 80% of the basal area for every plot were selected as ‘focal species’ for functional trait measurements. Secondly, 11 species that had an abundance lower than 4 (adult trees and regeneration) were further excluded, as well as Tectona grandis that was only present in monoculture plantations. The list of focal species is presented in Appendix 2. The threshold of 80% is used because it is not realistically possible to sample every single species and 80% has been shown to be representative for the whole community (Pakeman & Quested, 2007). I selected nine functional traits that are important for the response of plants to different environmental gradient such as temperature and water availability and can be used to predict performance under future climatic conditions. Except for wood density that was retrieved from a database (Zanne et al., 2009), traits were measured on at least five individuals of each species, as this is the minimum recommended sample size to account for intra-specific variation (Cornelissen et al., 2003). Additionally, individuals growing under complementary conditions were selected to get a good representation of the natural variation within the species. Traits were measured using standardized protocols (Cornelissen et al., 2003; Pérez-Harguindeguy et al., 2013). See Appendix 1 for a detailed description of trait measurements procedure and their relation with drought survival.

2.2.3. Taxonomic and functional properties of the woody community

To quantify TD, a dataset describing the abundance of all encountered species in each plot was analyzed with the Biodiversity.R statistical software (Kindt, 2018) that was developed for the R statistical language and environment (R Development Core Team, 2016). Since species richness does not take into account the relative abundance of each species, the Shannon-Wiener diversity index (*H’*) was also used to describe the heterogeneity of species diversity. The index is calculated with the following formula:

$$H = -\sum_{i=1}^{S} p_i \ln p_i$$

where S is the total number of species (richness) in the community and p_i is the proportion of individuals belonging to the ith species.

In addition, rarefied species richness (rarefaction) was used instead or absolute richness when testing for differences between the established and the regenerating tree community. This measurement allows to take into account sampling effort by "generating the expected number of species in a small collection of *n* samples drawn at random from the large pool of *N* samples" (Gotelli & Colwell, 2001). Here I used a subsample size of 5.
For each plot, richness and *H* were calculated with the *diversitycomp* function of the BiodiversityR package while rarefied richness was calculated with the *rarefy* function of the vegan package.

FD was based on Rao's quadratic entropy (Rao's Q), which incorporates both the relative abundances of species and a measure of the pairwise functional differences between species (Zoltan, 2005). It is calculated using the equation:

$$FD_Q = \sum_{i=1}^{S-1} \cdot \sum_{j=i+1}^{S} p_i \times p_j \times d_{ij}$$

where S is the number of species in a community, p_i and p_j are the relative abundances of species i and j, and d_{ij} is the trait distance between species i and j in a community.

FC was characterized by the community-weighted mean (CWM) of each single trait, which corresponds to the mean value of a trait in the community weighted by the species relative abundance and is calculated as:

$$\mathrm{CWM} = \sum_{i=1}^{n} p_i \times \mathrm{trait}_i$$

where p_i is the relative abundance of species i to the community, and trait_i is the trait value of species i.

It is assumed that CWM values reflect the trait value that optimizes fitness and performance of organisms in a given set of environmental conditions (CWM-optimality hypothesis) (Muscarella & Uriarte, 2016). This can be explained by the fact that species contributing strongly to the CWM are the most abundant ones, thus presumably the best adapted to specific abiotic conditions (Shipley et al., 2011).

Rao's Q and CWM were obtained by computing coupled datasets containing information on tree communities (tree relative abundance for a cross-tabulation of plots x focal species) and functional traits (functional traits values for a cross tabulation of focal species x traits) in R using the *dbFD* function of the package "FD" (Laliberté et al., 2015).

2.3. Data analysis

To investigate difference in species composition between the regenerating and the established tree community, I constructed a species-by-site matrix with abundance as input data and calculated the ecological distance between each pair of plots using the Bray-Curtis distance. I then tested whether species composition differs significantly between the established and the regenerating vegetation first in the entire study site and then in each land-use type using permutational multivariate analysis of variance (PERMANOVA). This was done using the function *vegdist* and *adonis* of the "vegan" package (Oksanen et al., 2018). Moreover, I ran a distance-based redundancy analysis (db-RDA) of the same matrix using the vegetation community age (established or regenerating) as a constraining variable with function *capscale* from the same R package. This is a constrained ordination technique based on metric multidimensional scaling, which can illustrate (dis)similarities in species composition according to some explanatory variables (Kindt & Coe, 2005), community age (established or regenerating) in this case. Next, to test differences in taxonomic and functional diversity between the established and the regenerating tree community I used a paired Wilcoxon signed-rank test.

The hypothesis that species richness, species diversity (i.e Shannon-Wiener index), functional diversity (i.e Rao's Q) and functional composition (i.e CWM) of the regenerating tree community is different between different land-use types was tested with a Kruskall-Wallis rank sum test. For the CWM of traits that follow a normal distribution, a simple ANOVA was used. Pairwise differences were further analyzed with a Dunn's Multi Comparison posthoc test using the package "dunn.test" in R (Dinno, 2017). Dunn's test is appropriate for groups with unequal numbers of observations, which was the case in this study (Zar, 2009). The response of diversity parameters along a gradient of different intensity of farming practices was tested with regression analysis in R using the *lm* and *glm* function of the "stats" package. Generalized linear models (GLM) (family=Poisson) were used for species richness while general linear models were used for taxonomic diversity after having log-transformed the data to meet assumption of normality. For functional diversity, no alternative statistical method was suitable to fit the nature of the data even after log-transformation. I used the Gaussian distribution even though it did not strictly meet the assumptions of normality. The CWM of five traits were transformed (log, ln or square-root transformation) in order to perform general linear model and test the effect of farming practices on the functional composition.

Additionally, the effect of both land-use and farming practices on diversity parameters of the regenerating vegetation were systematically tested in a series of alternative models. I started with the most complex model including every land use type and all the farming practices. Then a backward selection was employed where non-significant independent variables were dropped and the model with the best fit was selected based on Akaike information criterion (AIC). When models did not differ significantly (ΔAIC < 2), I chose the model which had the highest number of significant explanatory variables.

All analyses were conducted at alpha = 0.05 in R version 3.5.1 (R Development Core Team, 2016) using RStudio environment version 1.1.442 (RStudio Team, 2016).

3. Results

3.1. Difference between established and regenerating tree community

Taxonomic and functional diversity - A total of 43 regenerating tree species and 46 established tree species were counted in the study site. Difference in TD (rarefied richness and Shannon index) and FD (Rao's Q) between the established and regenerating tree communities in each land-use type was tested with paired Wilcoxon signed-rank test (Table 3). I found that in the buffer zone, the regenerating community has significantly higher values compared to the established community for all the diversity parameters except for Rao's Q. In the other land-use type, TD and FD were higher in the established community, except in bush fields where the difference was not significant.

Table 3 *Diversity parameters (mean ± SD) of the established and regenerating tree community. Taxonomic diversity is measured using Shannon-Wiener diversity index (*H'*) and functional diversity using Rao's quadratic entropy (Rao's Q). V correspond to the value of the signed rank statistic. Significant differences are indicated in bold.*

Land-use	**Established community**	**Regenerating community**	**Wilcoxon signed-rank test**	
			V	*P*
Buffer zone				
Rarefied richness	2.34 (±1.18)	3.27 (±0.83)	321	**<.01**
H'	0.72 (±0.57)	1.25 (±0.39)	405	**<.001**
Rao's Q	0.01485 (±0.013)	0.01954 (±0.008)	147	0.08
Protected area				
Rarefied richness	3.30 (±0.53)	2.92 (±1.21)	19	0.43
H'	1.50 (±0.34)	1.07 (±0.65)	12	0.13
Rao's Q	0.026 (+0.012)	0.016 (±0.009)	46	0.06
Long fallow				
Rarefied richness	2.17 (±1.19)	2.12 (±1.19)	106	0.98
H'	0.72 (±0.60)	0.65 (±0.54)	102	0.65
Rao's Q	0.01497 (±0.0146)	0.0165 (±0.0178)	61	0.73
Short fallow				
Rarefied richness	1.45 (±1.11)	1.68 (±1.27)	30	0.83
H'	0.50 (±0.45)	0.50 (±0.51)	33	1
Rao's Q	0.013 (±0.012)	0.008 (±0.009)	10	0.58
Bush field				
Rarefied richness	1.54 (±1.25)	1.65 (±1.20)	102	0.80
H'	0.38 (+0.58)	0.43 (±0.54)	76	0.70
Rao's Q	0.0078 (±0.012)	0.0056 (±0.001)	85	0.40
Home field				
Rarefied richness	0.95 (±0.10)	0.82 (±1.11)	479	0.50
H'	0.21 (±0.36)	0.20 (±0.43)	149	0.98
Rao's Q	0.0041 (±0.008)	0.0035 (±0.008)	128	0.77

Species composition - The results of the Adonis analysis indicated that species composition differs significantly between the established and the regenerating woody community across the 160 plots (F=21.11, P=0.001). The ordination plot indicates that species *Vitellaria paradoxa*, *Anogeissus leocarpus* and contribute particularly to the difference in SC between the two communities, due to their large abundance as established trees (Figure 4).

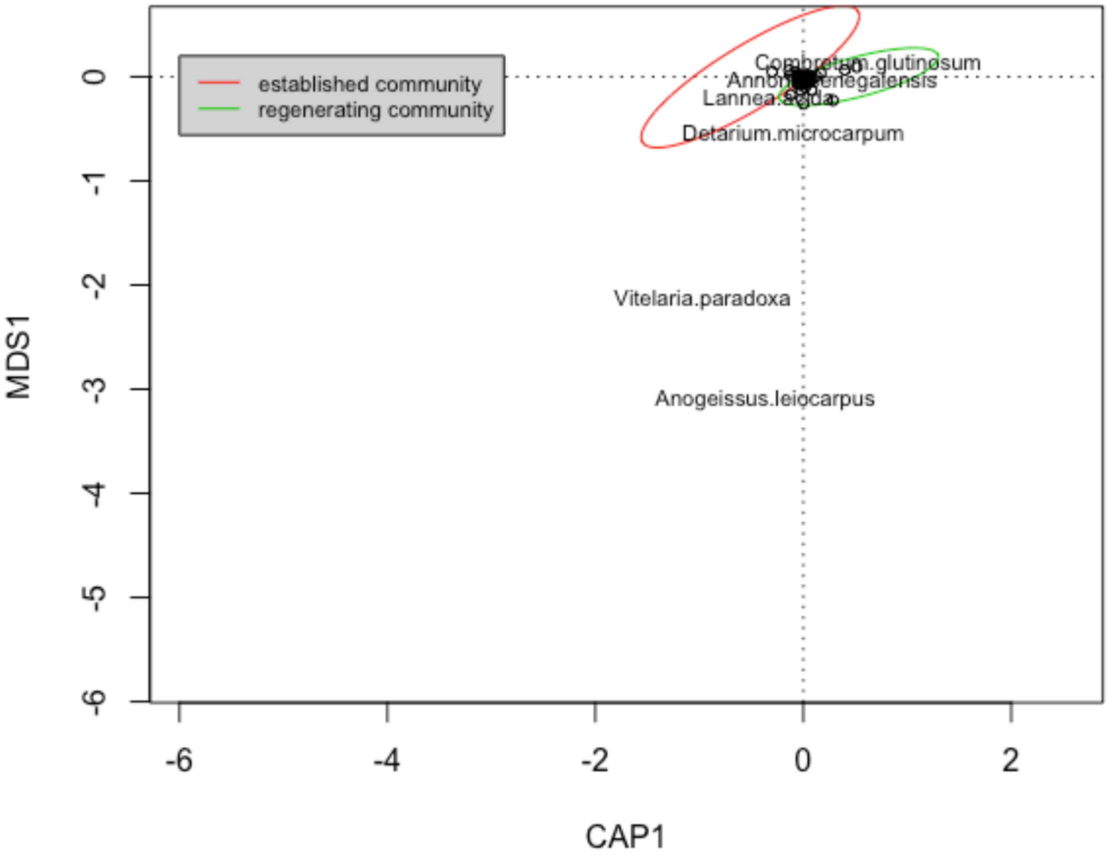


Figure 4 *Ordination plot of db-RDA (*distance-based redundancy analysis*). The ellipses indicate where 95% percent of site of the same category (established or regenerating tree community) are expected to occur. The low overlap of the two ellipses indicates that both communities differ in terms of species composition*

Further the Adonis analysis showed that SC is significantly different between the established and regenerating tree communities in all type of land-use: buffer zone (F=5.1195, P=0.001), bush field (F=8.2261, P=0.001), home field (F=10.368, P=0.001), long fallow (F=3.9248, P=0.001), protected area (F=2.4804, P<0.01) and short fallow (F=3.0943, P<0.01). A table detailing species abundance in each community and across the six land-use types is provided in Appendix 2.

Five species made up more than half of the abundance in the regenerating community, from which three are found in the the Combretacea family (*C. glutinosum*, C. nigricans and *G. senegalensis*) and one in the Caesalpiniaceae familiy (*P. thonningii*). Vitellaria paradoxa was also dominant in both the regenerating and established, with respectively 12% and 31% of the total number of individual (Figure 5a; 5b). *A. leiocarpus* was the second most abundant species in the established community but occurred mainly in the buffer zone and the protected area (Figure 5b; Table 3 Appendix 2). *Lannea spp*. were also abundant in the established tree community (Figure 5b).

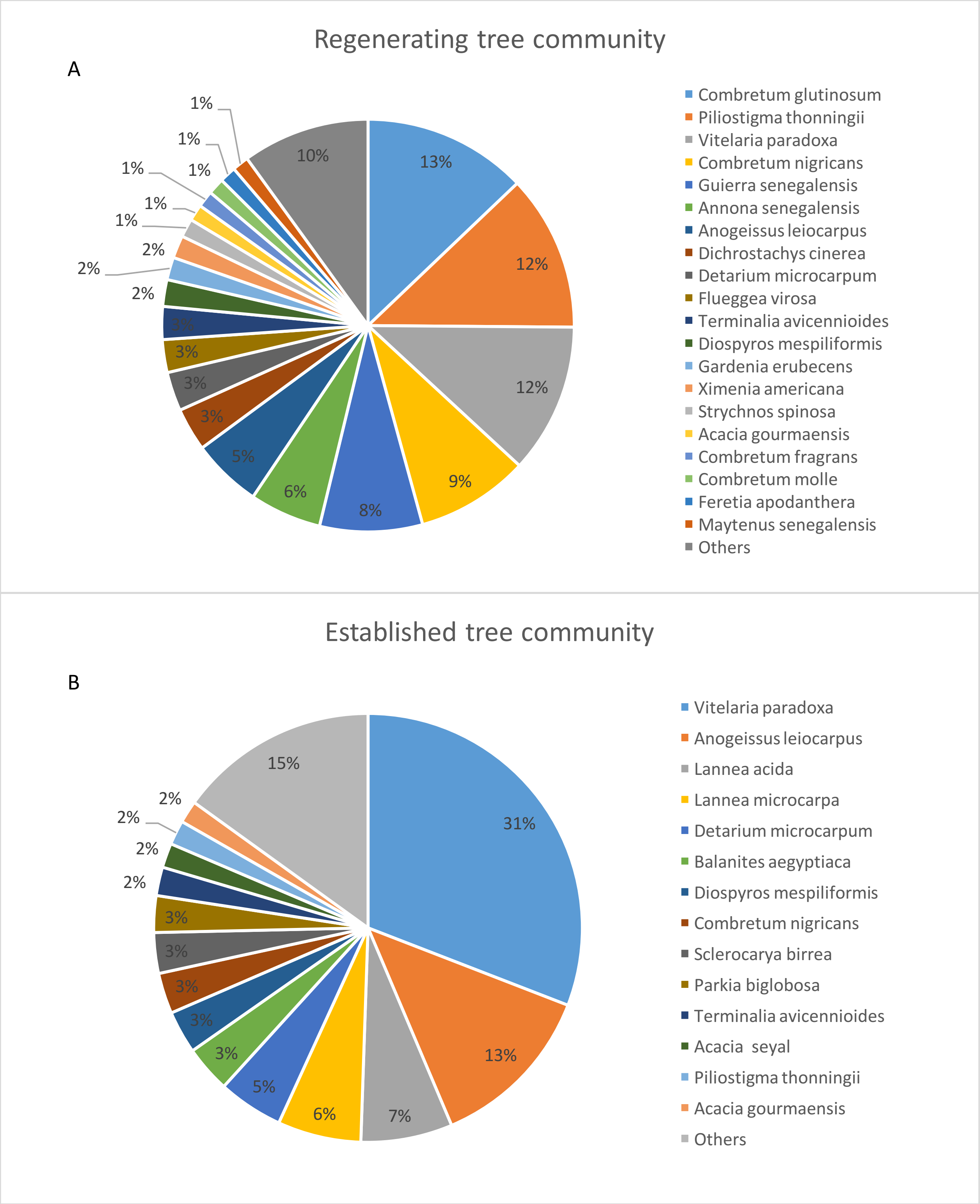


Figure 5 *Relative abundance of regenerating (A) and established (B) tree species in the entire study site*

Functional composition – All community-weighted mean trait values differed significantly between the regenerating and the established tree community when all the plots were pooled together (Wilcoxon signed-rank test, all $p < .05$). In short fallows, the CWM of the two communities did not differ significantly for any of the functional trait measured. In the other types of land-use, maximum height of the established trees was significantly higher. Additionally, the regenerating community had a higher LT, TDMC and WD but smaller LA than the established community, but the difference was not significant in the protected area. SLA of the regenerating community was higher in the buffer zone and protected area but only significantly in the former. CWM for compoundness was always higher in the established community but the difference was found to be significant only in the buffer zone, bush field and long fallow. Deciduousness was significantly higher in the established community only in the buffer zone and protected area. A table with the results from the paired Wilcoxon signed-rank test in each land-use is provided in Table 1 of Appendix 3.

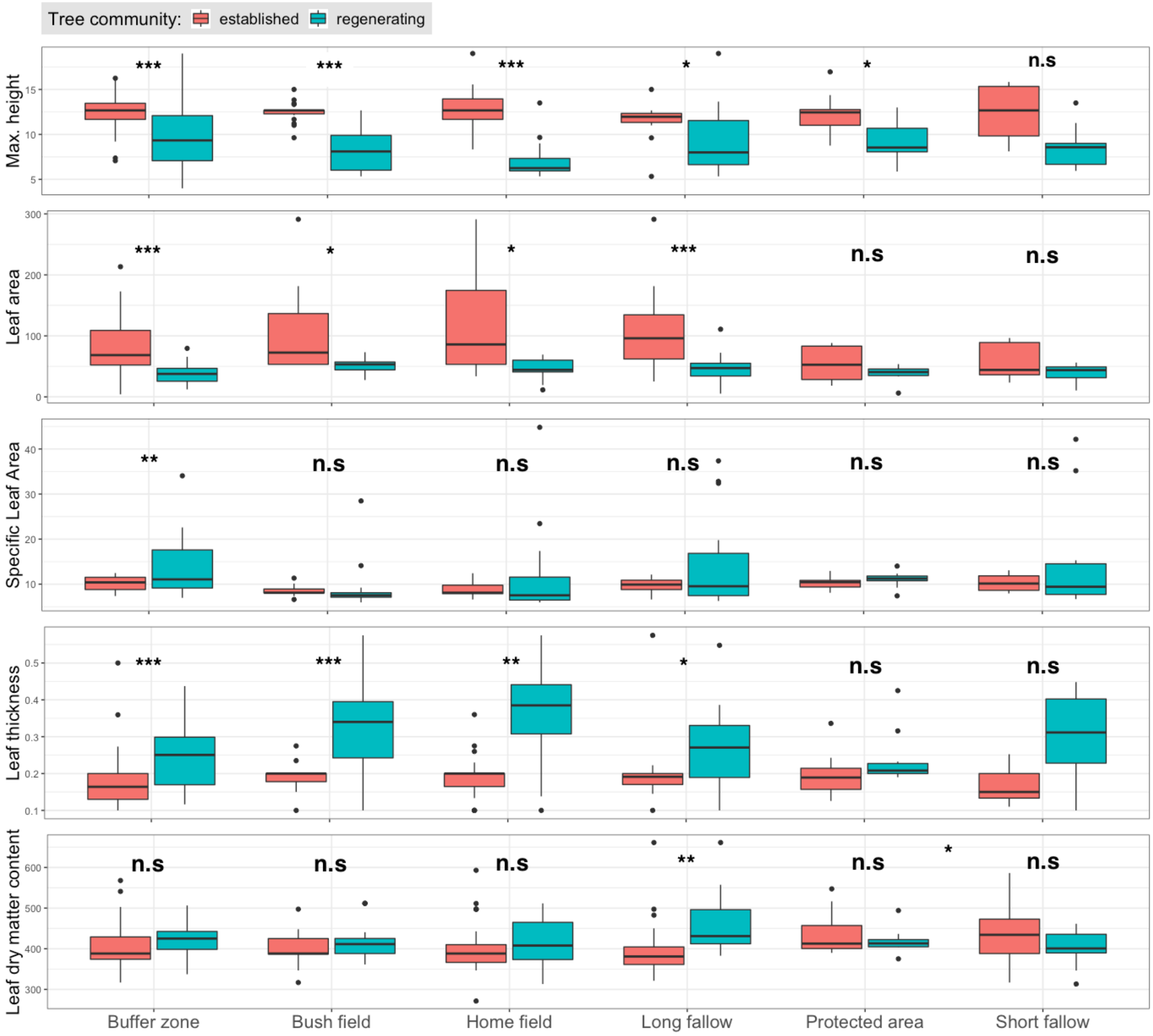
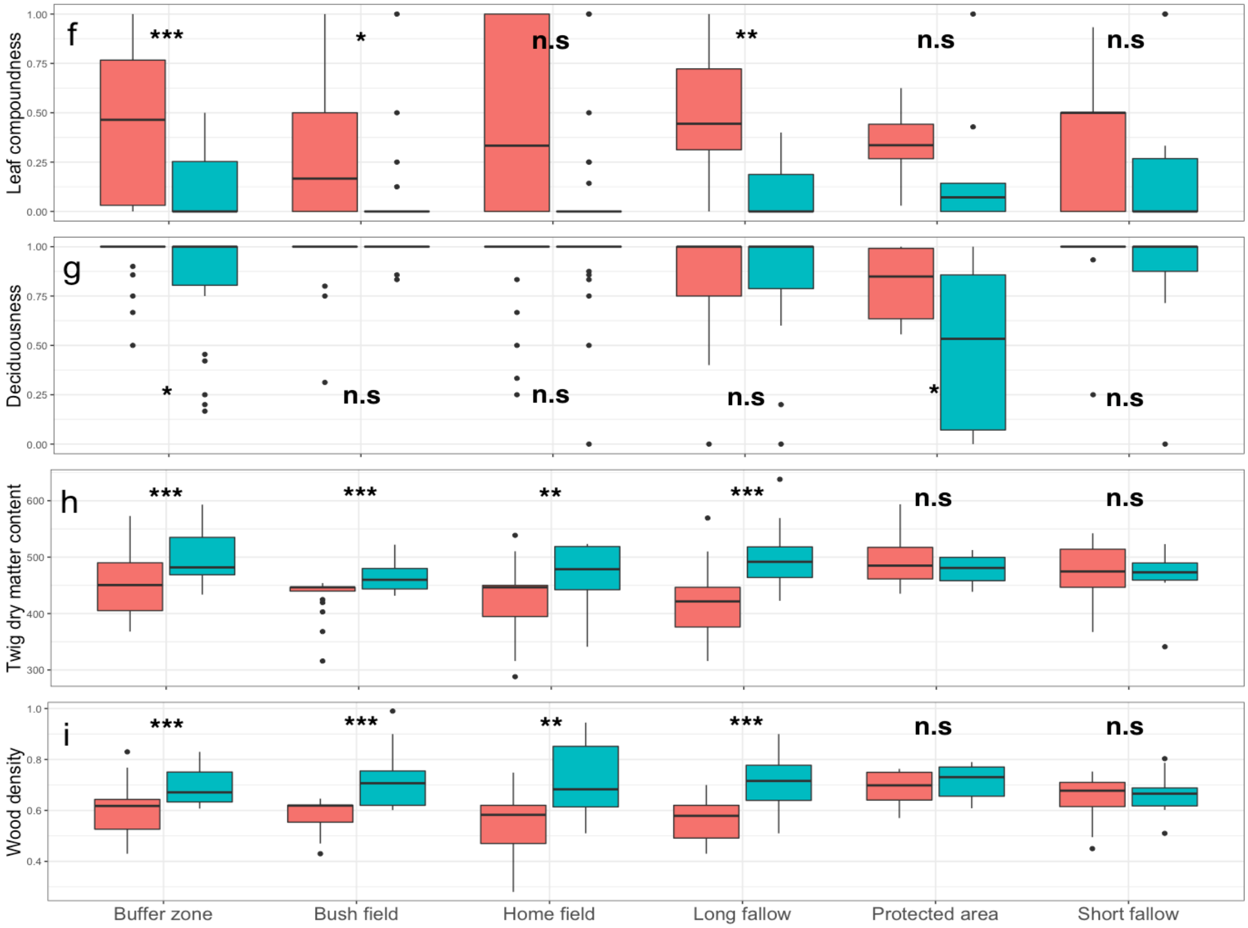


Figure 6 *Differences in functional composition between the regenerating and established tree community across different land-use type. Asterisks indicate the significance level of the difference (***: <0.001, **: <0.01, *: <0.05, n.s: not significant).*

Figure 6 *continued.*



3.2. Effect of land-use type on the regenerating tree community

Taxonomic and functional diversity - Most of the regenerating species were found in the buffer zone (BZ) (32 species in total across the plots) followed by the protected area (PA) and LF (20 species each). Surprisingly, home fields (HF) had a higher number of species (19 species) than bush fields (BF) and short fallow (SF) (15 and 12 species respectively) but this could be a result from the larger sample size of this category, which is indeed confirmed when looking at the average richness per plot (Table 4). Plots in HF and BF had the smallest average species richness (0.88 and 1.79 respectively) followed by SF (1.85). Average species richness was the highest in the BZ and PA (4.23 and 4 respectively) while intermediate in long fallow (LF) (2.39). Taxonomic and functional diversity follow the same pattern (Table 4).

Table 4 *Diversity parameters (mean ± SD) of the regenerating tree community in the different land-use type. Mean values with different superscript in a column differ significantly at P < 0.05.*

	Buffer zone (n=30)	**Bushfield (n=24)**	**Home field (n=60)**	**Long fallow (n=23)**	**Protected area (n=10)**	**Short fallow (n=13)**
Total richness	32	15	19	20	20	12
Taxonomic richness	4.23^{a} (±1.57)	1.79 ^{c} (±1.44)	0.88 ^{d} (±1.29)	2.39 ^{b,c} (±1.56)	4.00 ^{a,b} (±2.71)	1.85 ^{c} (±1.46)
Taxonomic diversity (H’)	1.25 ^{a} (±0.39)	0.43 ^{c,d} (±0.54)	0.20 ^{d} (±0.43)	0.65 ^{b,c} (±0.54)	1.07 ^{a,b} (±0.65)	0.50 ^{b,c,d} (±0.51)
Functional diversity (Rao’s Q)	0.01954 ^{a} (±0.008)	0.006 ^{c} (±0.01)	0.007 ^{c} (±0.01)	0.016 ^{a,b} (±0.017)	0.019 ^{a,b} (±0.009)	0.011 ^{b,c} (±0.009)

A Kruskal-Wallis test showed that significant differences in taxonomic richness (chi-squared = 72.443, df = 5, p-value = 3.176e-14), taxonomic diversity (chi-squared = 66.699, df = 5, p-value = 4.975e-13) and functional diversity (61.365, df = 5, p-value = 6.345e-12) were found among the six categories of land-use. Post hoc comparisons using the Dunn’s test indicated which land-use types differs significantly from each other. Buffer zone was statistically similar to protected area in all the diversity metrics and to long fallow as well for functional diversity. Protected area was similar to long and short fallow in all diversity metrics but only to long fallow for richness. Long fallow was similar to short fallow for all diversity metrics, as well to bush field except for functional diversity. In term of richness, short fallow was statistically similar to bush fields, as well as home field in terms of taxonomic and functional diversity. Finally, bush field was satistically similar to home field only when it comes to taxonomic and functional diversity. Results are summarized in Table 4 and Figure 4 with group letters were land-use types sharing the same letter were not found to be significantly different.

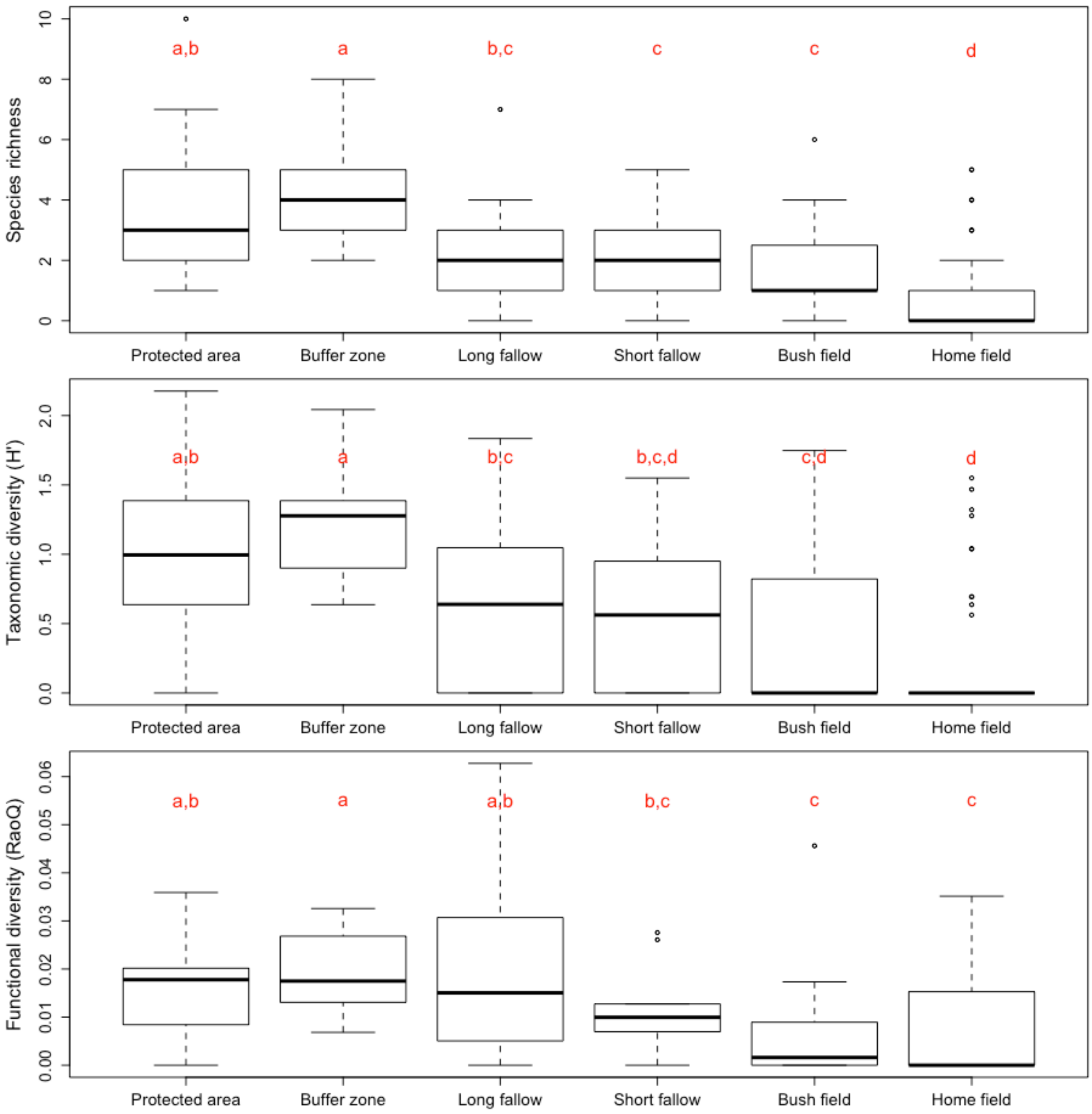


Figure 7 *Boxplot of diversity parameters of the regenerating community in each land use type. Land uses are ordered by increasing management intensity. Boxes accompanied by a different letter are significantly different at P < 0.05.*

Functional composition - The null-hypothesis of equal median for the six land-use types was rejected for 5 out of the 9 functional traits evaluated (one-way ANOVA or Kruskall-Wallis test, *P* <0.05). Post hoc test that revealed that maximum height was significantly lower in home field than in other land-use except bush field (Fig. 5a). Twig dry matter content was significantly higher in buffer zone and long fallow compared to bush field (Fig. 5b). Buffer zone and long fallow had higher SLA than bush and home fields (Fig. 5c). Leaf thickness was higher in bush and home field than buffer zone, long fallow and protected area. Short fallow did not differ significantly from the other land-use types for this trait (Fig. 5d). There were less deciduous individuals (i.e. deciduousness was lower) in the protected area than in the other land-use types. Additionally, there were significantly more deciduous trees in bush field than in buffer zone and long fallow (Fig. 5f).

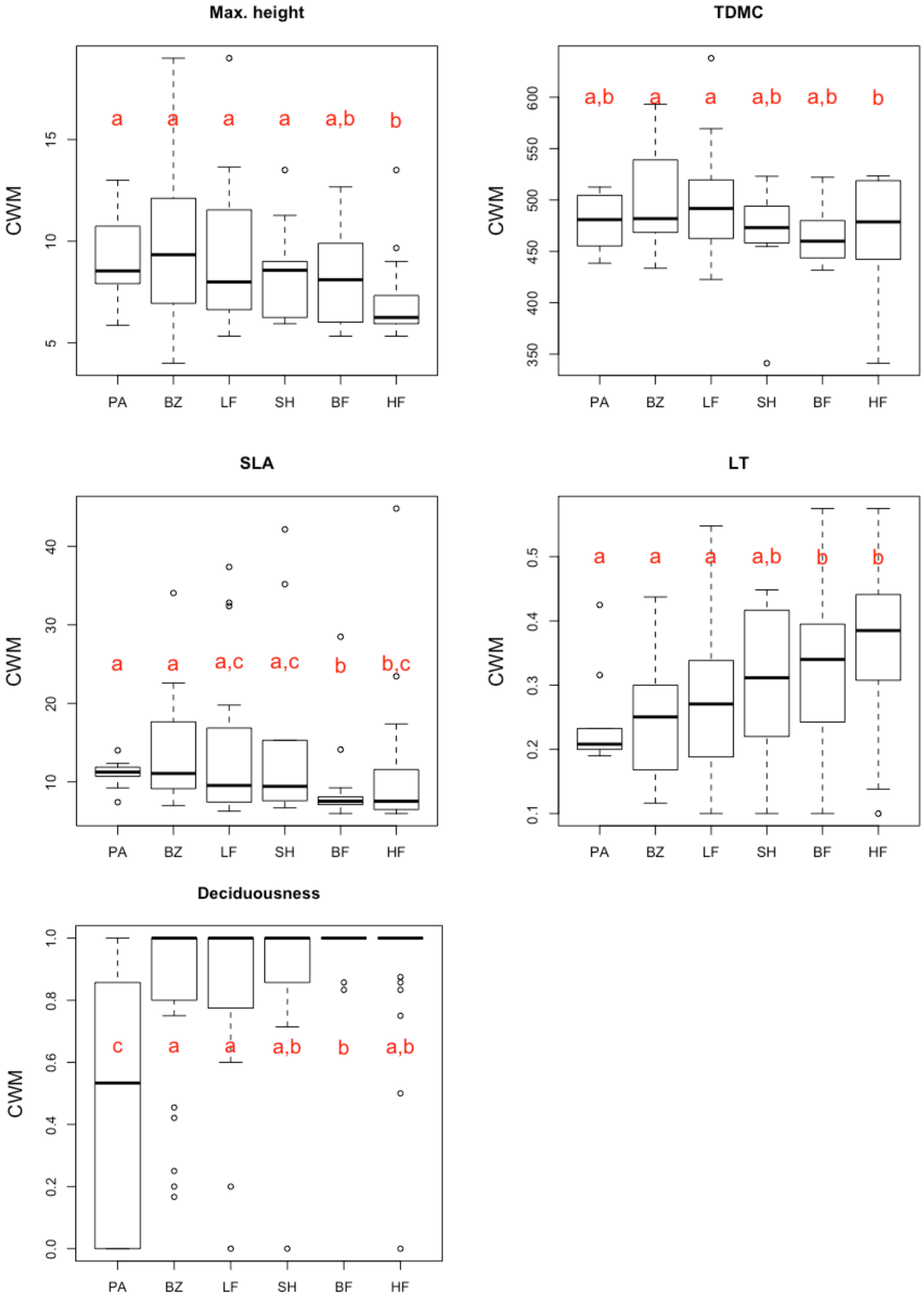


Figure 8 *Differences in functional composition of the regenerating community across different land-use types. Land uses are ordered by increasing management intensity. Boxes accompanied by a different letter have significantly different CWM values at P < 0.05.*

3.3. Response of regenerating tree community to farming practices

Taxonomic and functional diversity - Regression results showed that selective tree cutting and fire had a positive effect on the richness (coefficient: 0.33, *P*<0.001 and 0.22, *P*<0.001 respectively) and the taxonomic diversity (coefficient: 0.05, *P*<0.001 and 0.04, *P*<0.001 respectively) of the regenerating community. Tree cutting also had a positive effect on functional diversity (coefficient: 0.001, *P*<0.001). On the other hand, livestock grazing was found to have a negative effect on richness (coefficient: -0.58, *P*<0.001) and taxonomic diversity (coefficient: -0.09, *P*<0.01) only (Figure 6).

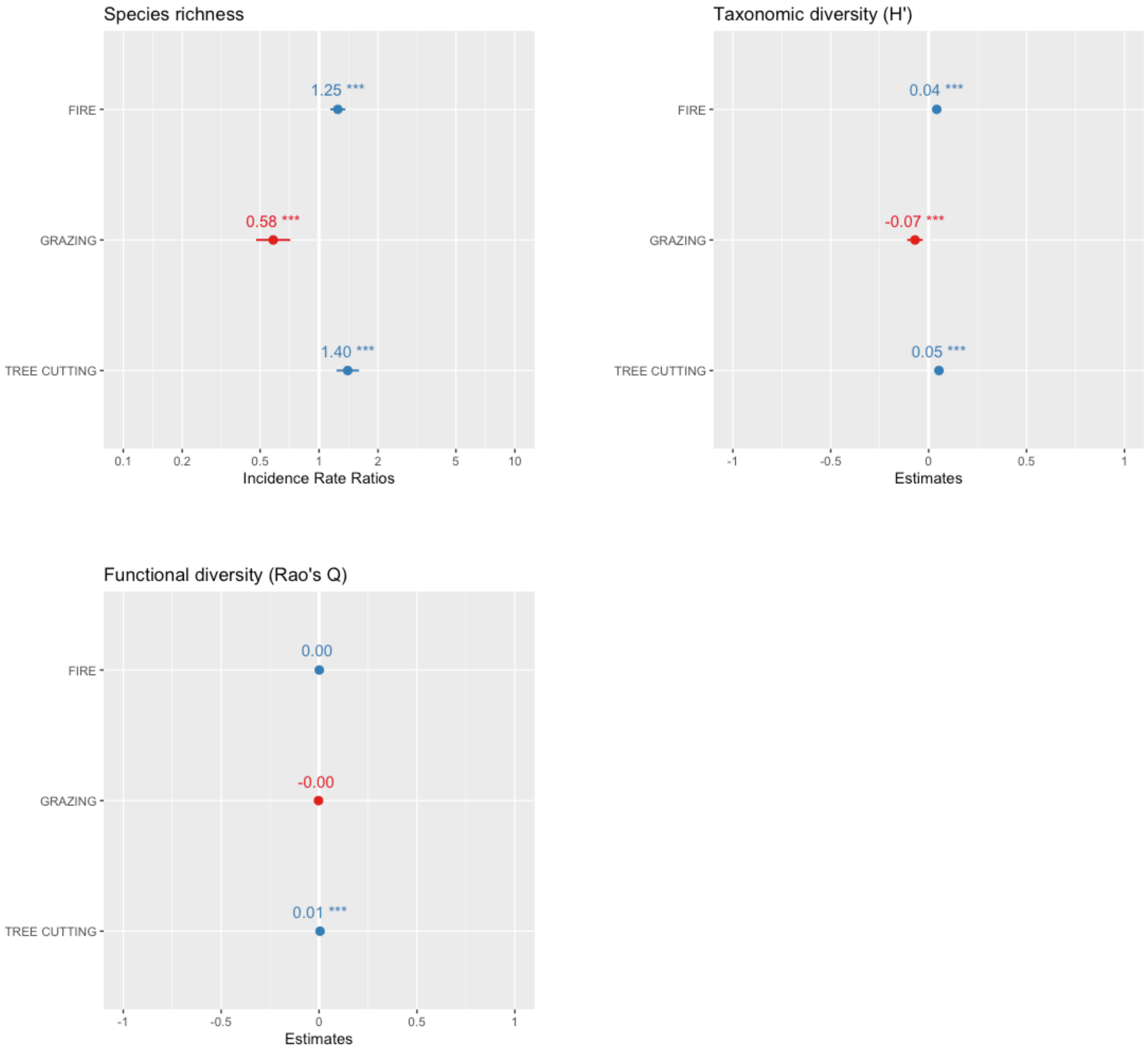


Figure 9** Effect sizes of farming practices regime on richness, taxonomic diversity and functional diversity of the seedling community across all plots. Graphs show the coefficient (incidence rate ratio or estimate and confidence interval) of the regression models. Asterisks indicate the significance level of the explanatory variables (<0.001, ** <0.01, * <0.05).*

Functional composition - Results from the regression analysis showed that selective tree cutting had a significant effect on the community-weighted mean of 5 functional traits. LA, LT, and deciduousness decreased while LDMC and TDMC both increased with tree cutting intensity. Fire only had a significant effect on WD and the effect of grazing was never significant (Figure 7). Leaf compoundness, deciduousness and SLA were not affected by any of the farming practices.

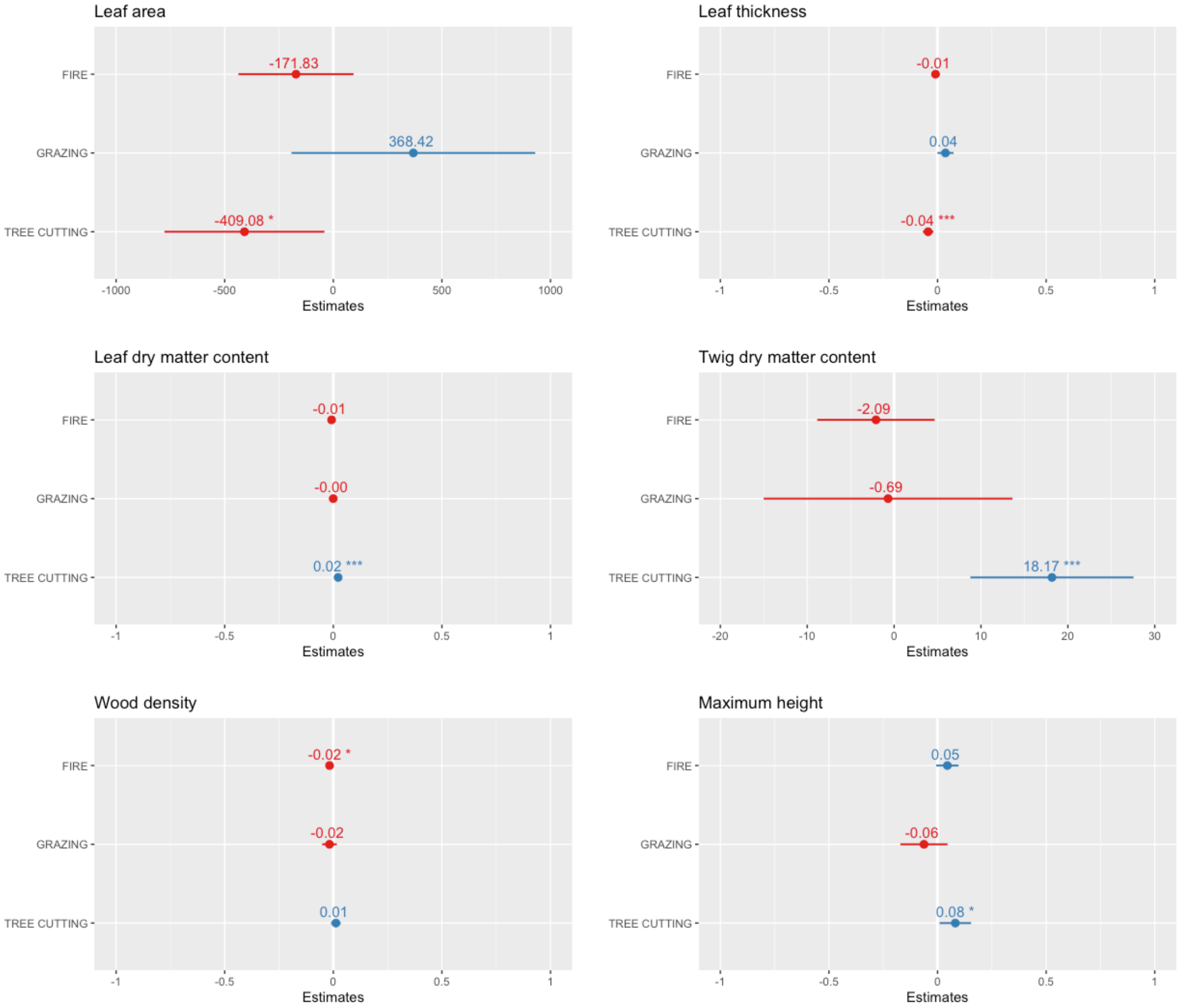


Figure 10 *Effect sizes of fire, grazing and tree cutting on the functional traits of the regenerating tree community across all plots. Plots show the coefficient (estimate and confidence interval) of the regression models. Asterisks indicate the significance level of the explanatory variables (***<0.001, ** <0.01, * <0.05).*

Species richness was best explained when all the variables (land-use type, selective tree cutting, fire and livestock grazing) were included in the model. Taxonomic diversity was best explained when only land-use type and tree cutting were used, although the difference was not significant when excluding or including all the farming practices in the optimal model (ΔAIC > 2). Functional diversity was best explained when only land-use was included as explanatory variable, although when tree cutting was added the difference was not significant. The best model for richness, taxonomic and functional diversity explained 57%, 41% and 16% of the variance respectively (Table 5). The statistics for the optimal models are provided in Table 1 of Appendix 4.

Table 5 Alternative models tested to explain the diversity parameter of regenerating vegetation. The optimal model was obtained using a backward selection where non-significant independent variables were dropped and the model with the best fit was selected based on Akaike information criterion (AIC).

#	Model	AICc	Δ	R_{adj}^2
1	Richness ~ Land use + Tree cutting + Fire + Grazing	542.6	0	0.57
2	Richness ~ Land use + Tree cutting	545.7	3.03	0.54
3	Richness ~ Land use + Tree cutting + Fire	546.0	3.40	0.55
4	Richness ~ Land use	547.2	4.57	0.53

#	Model	AICc	Δ	R_{adj}^2
1	Taxonomic diversity ~ Land use + Tree cutting	-196.6	0	0.41
2	Taxonomic diversity ~ Land use	-194.8	1.86	0.40
3	Taxonomic diversity ~ Land use + Tree cutting + Fire + Grazing	-194.7	1.92	0.42
4	Taxonomic diversity ~ Land use + Tree cutting + Fire	-194.4	2.24	0.41

#	Model	AICc	Δ	R_{adj}^2
1	Functional diversity ~ Land use	-710.9	0	0.16
2	Functional diversity ~ Land use + Tree cutting	-710.4	0.48	0.17
3	Functional diversity ~ Land use + Tree cutting + Fire	-708.3	2.62	0.16
4	Functional diversity ~ Land use + Tree cutting + Fire + Grazing	-706.7	4.22	0.16

4. Discussion

I analyzed the similarity between the regenerating and the established tree community in term of species and functional composition as well as taxonomic and functional diversity. I then evaluated changes in the taxonomic and functional diversity of the regenerating tree across six types of land-use commonly found in West African parklands. Finally, I investigated the response of these diversity indicators to different regime of farming practices (tree cutting, grazing and fire).

Here, I discuss my main results and the implications for the future of the parklands as well as the success of FMNR as a restoration practice in African drylands.

To what extend the regenerating vegetation contribute to future tree diversity?

As predicted, the established tree community had a higher taxonomic diversity (richness and Shannon diversity index) than the regenerating community, but the difference was not significant (Table 3). The opposite was observed in the buffer zone, where the regenerating population was significantly more diverse than the established one (Table 3). Lykke (2000) mentioned that large tree species are often overexploited in reserves for their wood, edible fruit and fodder. This could explain the difference in taxonomic diversity observed in the buffer zone, where selective logging is allowed.

Functional diversity was also higher in the established than regenerating community except in buffer zone and long fallow, but the difference was not significant in any of the land use type (Table 3).

It should be noted that a higher species and functional diversity in the established community could be due to a larger coverage range of DBH compared to the regenerating community. By chance, more species could have been accumulated in the established tree class since it covers a longer period of time. However, my results are similar to those found by Kindt et al. (2008) who used six diameter classes, demonstrating that several species in west African parklands have no individuals in the smaller diameter classes.

I found that species composition of the established and regenerating tree community was significantly different in every land-use type. As predicted, more than half of the species making up the regenerating community in the whole study site belonged to the Combretaceae and the Caesalpiniaceae family (Figure 5a). An increase of these shrub species has already been reported by previous studies focusing on regeneration dynamics in the Sahel region (Gijsbers et al., 1994; Hänke et al., 2016; Paré, 2008). The authors attribute this to the remarkable ability of these species to resprout after the destruction of above-ground biomass (due to browsing or cutting for example), giving them an competitive advantage in disturbed environments. This would explained why in this study they were the most abundant species in cultivated fields and fallows (Table 2 Appendix 2). Additionally, these species belong to the northern Sahelian zone and therefore are adapted to drier environment (Arbonnier, 2004). While they are often exploited as construction and fuel wood as well as for their medicinal value, the predominance of these species is generally regarded as a characteristic of degraded land in semi-arid West Africa (Houerou, 1989).

Shea tree (*Vitellaria paradoxa*) was also abundant in the regenerating population and made up one third of the trees in the established population (Figure 5a and 5b). The high economic value of *V. paradoxa* provides an incentive for farmers to retain this species when preparing their fields and protecting the seedlings against livestock, thus explaining the abundance of Shea tree in the study site (Jurisch et al., 2012). *Anogeissus leiocarpus* was the second most abundant adult tree species. It is a pioneer species that regenerates mainly through seedling sprout, i.e an individual of seed origin that was affected by shoot dieback, but resprouted from the root collar of the seedling (Ky-Dembele, et al., 2007). This could explain why its presence was mainly restricted to the protected area, buffer zone and long fallows, where disturbance is moderate (Table 2 Appendix 2). *Lannea microcarpa* and *Lannea acida* were also two dominant species in the established tree population. The maturity of their fruits occurs at the end of the dry season when food is scarce, making them highly appreciated by people. Arbonnier et al. (2004) reported that they are often protected and sometimes planted by farmers but their regeneration was extremely low in this study (Table 1 Appendix 2).

There is an increasing amount of evidence suggesting that several tree species in west African parklands may be in danger of extinction (Kindt et al., 2008; Wezel & Lykke, 2006). These studies, based on size class distributions, showed that many native species are not regenerating well as they are under-represented in the smaller diameter classes. These include socio-economically important species like *Parkia biglobosa*, *Bombax costatum*, *Adansonia digitata* and *Afzelia africana* (Bayala et al., 2011). In this study, I found no individuals in the regenerating portion of these species (Table 1 Appendix 2). Lykke (1998) reported that a shift in vegetation type has been taking place, from one represented by valuable large tree species to one dominated by shrubs and small trees, consistent with my results.

Based on the CWM of functional traits, I found that the regenerating portion of trees showed more signs of drought tolerance than the established trees, supporting my findings on the difference in species composition above (Figure 6). The regenerating community had a higher leaf thickness, TDMC and wood density but a lower leaf area than the established community. Increased leaf thickness and high stem density can enhance long-term survival for perennial flora in arid environment (Leigh et al., 2012; Bucci et al., 2004) while smaller leaves are less prone to drought-related death (Pérez-Harguindeguy et al., 2013). The maximum potential height was lower for the regenerating community in the whole study site, confirming the shift from large tree species to shrubby vegetation mentioned by Hänke et al. (2016). CWM of leaf dry matter content was always higher in the regenerating community but the difference was significant only in the long fallow. I expected that the regenerating community would have more individuals with a deciduous and compound leaf habit but the opposite was observed, although the difference was only significant in two and three land-use types respectively. The reason for this is not clear, but it is possible that deciduousness and leaf compoundness are not the primary mechanisms used by the regenerating vegetation to avoid or resist drought. Below-ground traits like rooting depth are considered to be particularly important when it comes to drought survival (Lopez-Iglesias et al., 2014). Unfortunately they are difficult traits to measure and could not be included here.

My results are consistent with recent studies that found a change in species composition in the West African Sahel featured by a replacement of mesic species by xeric species, despite an increase in annual precipitation since the mid-1980s (Gonzalez et al., 2012). Hänke et al. (2016) attributed this shift in species composition to an in intensification of land use systems, characterized by the abandonment of fallowing and intensive soil tilling, increased grazing pressure and promotion of fast growing species for construction and fuel wood. In the predicted climate scenario for this part of Africa (i.e. erratic rainfall, persistent dry spells), such drought-resistant vegetation is likely to be more adapted and would ensure the provisioning of important ecosystem services for farmers like nutrient recycling and soil protection. On the other hand, these drought tolerant species are not important in term of fruit production, which represent an important resource for income generation, nutrition and food security in the parklands (Gaisberger et al., 2017).

Effect of land use on regenerating trees

Richness and species diversity - I hypothesized that diversity indicators would increase when moving from intensively managed to less disturbed land-use type. Results indeed revealed that mean richness of the regenerating community was lower in home and bush fields, intermediate in both fallow categories and higher in the protected area and its buffer zone (Table 4; Figure 4). Lower richness in home fields compared to bush fields can be linked to their proximity to human settlements, causing home fields to be more intensively used and less fallowed (Kindt et al., 2008). The difference was however not significant for taxonomic diversity, indicating that a few species dominate these two land-use types. Species richness and diversity are known to increase with fallow age (Augusseau et al., 2006), consistent with my results were long fallows had a higher regeneration diversity than short fallows, although the difference was not significant (Table 4). The fallow phase allows the vegetation to invade abandoned land either through seed dispersal from neighboring trees, or through re-sprouting of stumps and germination from the seed bank. Hence, fallows are considered to be important important land units for the establishement and regeneration of many woody species in agroforestry parklands (Nikiema, 2005).

Interestingly, the regenerating community was more diverse in the buffer zone than the protected area, despite being under greater human pressure due to encroachment. It is possible that local disturbances in the buffer zone such as the creation of gaps by selective tree cutting, dead wood collection and the effect of livestock (grazing, trampling, seed dissemination) favors the establishment of additional species (pioneer species for example), thus enhancing the diversity of regenerating trees (Molino & Sabatier, 2009). This is in line with the intermediate disturbance hypothesis, which predicts a peak of local species diversity at an intermediate level of disturbance (Bongers et al., 2009), although an experimental set up would be needed here and quantifying disturbance and canopy gaps in order to validate this hypothesis. It is however important to note that, when looking at the established community, the buffer zone had a lower richness and diversity than the protected area (Table 3), highlighting the importance of the latter for biodiversity conservation.

Functional diversity and composition – Functional diversity of the regenerating community followed the same pattern as taxonomic diversity, being significantly higher in the protected area and buffer zone compared to highly disturbed environment like home and bush fields (Table 4; Figure 4). In a global meta-analysis, Laliberté et al. (2010) also found that land-use intensity reduced the diversity of plants functional trait associated with disturbance (i.e. response diversity). The importance of response diversity for ecosystem stability has been demonstrated experimentally in arid rangelands, where authors found that plant communities with the lowest diversity of trait related to grazing tolerance were the most affected during disturbance (Chillo et al., 2011). Similarly, Hallett et al., (2017) showed that land-use practices that maintained greater functional diversity were associated with stability in total biomass and vegetation cover across experimental non-drought and drought conditions. The decrease of functional diversity with land-use conversion observed in my study suggests that emerging tree communities in the parklands might not be able to cope with climate variability in the future. The loss of multiple ecosystem services provisioned by native tree species could dramatically affect the food and income security of rural populations living west African parklands. In this study, minor species have been omitted in the measure of functional traits (see methods), but it should be noted that rare species may contribute substantially to resilience, especially if they respond differently to climatic and disturbance factors (Díaz & Cabido, 2001).
I predicted that increasing land use intensity would result in a shift towards more drought-resistance trait values at the community level. Although not all the functional traits responded to land use type (5 out of 9), those which did indicated a dominance of drought resistant plants in home and bush fields as expected (Figure 5). Regenerating vegetation in these land use types had lower values for maximum height, TDMC and SLA but higher values for leaf thickness and deciduousness compared to buffer zone or protected area where there is less human pressure. Twigs with low dry matter content are expected to dry out slower during the dry season (Cornelissen et al., 2003) while a low SLA is correlated with lower relative growth rates (RGR), which has been associated with longer drought survival time (Lopez-Iglesias et al., 2014).

It is increasingly recognized that vegetation dynamics in the parklands are strongly driven by land use changes (Usman & Nichol, 2018). One explanation is that growing rural population in West Africa is leading to land shortage and farmers being forced to shorten fallow period in order to maintain sufficient crop production (Gijsbers et al., 1994; Boffa, 2000; Nikiema, 2005). Abandonment and shortening of the fallow cycle may prevents many species to regenerate and reach critical height before the clearing of a new field for cropping, thus explaining the decrease of tree diversity in the parklands (Kindt et al., 2008). The extension of land under continuous cultivation is also promoting the growth of xeric species, suggesting that current land management is producing new and different types of parklands (Hänke et al., 2016). Other consequences of demographic pressure such as increasing livestock density and overexploitation of tree products have also been found to threaten the regeneration of woody vegetation in Burkina Faso (Gaisberger et al., 2017).
With the current trend of agricultural intensification and expansion in Burkina Faso, there is high risk of further environmental degradation in the parklands and urgent action is necessary. In these shifting cultivation systems, management at the landscape level is key to maintain/restore biodiversity (Augusseau et al., 2006). Scholars have recommended that management schemes should promote landscape heterogeneity, as it has been found to favor species richness (Weibull et al., 2003). Building on traditional land use practices, the proportion of long fallows should be increased and remnants of old growth forest patch maintained or increased (Robiglio & Sinclair, 2011). Improving connectivity between landscape units, using ecological corridors for example, is also important for biodiversity persistence (Opdam et al., 2003). Interventions to promote sustainable land use planning and management as a way to enhance tree diversity should also be compatible with farmers' objectives and needs. This could be

achieved by creating value chains for parklands tree products, domestication and planting of valuable species for local use (e.g. fruit and fodder trees) and payment for ecosystem services for example.

Response of regenerating woody vegetation to farming practices

Richness and species diversity - I predicted that selective tree cutting, fire and livestock would have a negative effect on the richness and taxonomic diversity of regenerating tree community. Counterintuitively, studied parameters actually increased with fire and tree cutting intensity (Fig. 6). The severity of fire on vegetation dynamics depends largely on its frequency and intensity, as well as species-specific characteristics of plants (Nikiema, 2005). For instance, recurrent fires can prevent seedlings to reach the necessary height to grow above the fuel bed and escape the next fire, reducing the rate of recruitment in tree populations (Lykke, 1998). In the Sudanian zone, fire at the end of the dry season (late fire) reduces tree density and diversity more significantly than early fire (i.e. right after the end of the rainy season), when the vegetation moisture content is still high (Sawadogo et al., 2002). This is explained by the difference in fire temperature, where highest temperature recorded in early fires was 333°C, which has minor destructive effects on vegetation, against 677°C in late fires (Nikiema, 2005). Additionally, plant species differ in their response to fire; some may resist any fire conditions and actually depend on it for the germination of their seeds while others rely on rapid vegetative reproduction after stems have been burnt (Savadogo, 2007). The negative effect of fire on the diversity of woody species in savanna ecosystems has been previously reported by many authors (Breman & Kessler, 1995; Hoffmann & Solbrig, 2003; Zida et al,. 2007), contradicting my results. One explanation to this could be that the majority of the species in the study site depend on fire as explained above, and absence of the latter may perturb their regeneration. Another explanation is that fire intensity is actually correlated with land use type. Land use type that harbored a higher diversity (protected area and buffer zone) also had a higher frequency of plots with high fire regime (level 2 and 3) whereas in home and bush fields fire regime was mainly zero.

In arid and semi-arid areas, opening of the canopy due to stand removal can result in unfavorable thermal conditions in the understory promoting the growth of drought-tolerant species (Savadogo, 2007). Studies have also showed that removal of trees can also increase grass production, leading to more competition with woody regeneration (Gambiza et al., 2000). In a factorial experiment, Zidal et al. (2007) reported that species richness did not vary significantly between cut and uncut plots. This was not the case in my study, as selective tree cutting had a positive effect on the regenerating tree richness and diversity. This can be attributed to increasing light conditions in the understory and reduction of competition for water and nutrient with taller trees (Hutchinson et al., 2005). Similarly to fire, the response of regenerating vegetation to selective tree cutting could actually be the effect of land use type rather than the regime of disturbance. Indeed, high regimes of selective tree cutting were more frequent in the buffer zone, where tree diversity was higher, than in farmers' fields, which could explain the positive effect of tree cutting on the regenerating community.

In this study, livestock grazing was found to have a negative effect on the richness and diversity of regenerating trees (Fig. 6). Livestock grazing is often regarded as one of the main causes of soil degradation and previous studies have found that species richness decrease at high grazing pressure (Warren et al., 2001; Mwendera et al., 1997). Additionally, the trampling caused by livestock can damage vegetation, especially seedlings, as well as influence the infiltration of water into the soil due to compaction (Savadogo et al., 2007). However, livestock can also have beneficial effect on the vegetation, for example by enhancing soil nutrients through dung deposition (Rufino et al., 2006) or favoring the dispersal and germination of seeds, thus enhancing plant richness at a given site (Razanamandranto et al., 2004). The degree of livestock impact depends on several factors like the type of animal, season of use, animal density, soil characteristics and plant communities present (Savadogo, 2007).

Crop damages by roaming livestock is a common source of conflicts between farmers and pastoralists in West Africa (Petit, 2003). For that reason, livestock is often herded in fallow lands during the growing season. As mentioned earlier, fallows are necessary for the regeneration of the vegetation, and increasing browsing pressure in these landscape units could seriously hampered the replenishment of trees in the parklands. In the context of FMNR, the protection of seedlings against livestock can include individual protection of desired trees or fencing around fields (Reij & Garrity, 2016). Financially supporting the cost of establishing such protective measures could contribute to accelerating the scaling-up of FMNR in African drylands.

Functional diversity and composition – I found that FD of the regenerating community did not respond to fire and livestock severity, but increased with tree cutting (Fig. 6).

FC of the regenerating woody community were also mainly affected by selective tree cutting. The CWM of leaf area and leaf thickness decreased with tree cutting severity while CWM of leaf and twig dry matter content increased (Fig. 7). As stated before, removal of stands and branches can result in increased temperature and loss of moisture in the understory, promoting the growth of drought-tolerant species (Savadogo, 2007). This could explain why the regenerating trees indeed exhibit traits values related to drought avoidance and tolerance strategies, such as low leaf area and high leaf and twig dry matter content (Cornelissen et al., 2003; Pérez-Harguindeguy et al., 2013; Leigh et al., 2012).

Possibly, there is a time lag between the onset of disturbance and the response of the vegetation but since no information on the duration and frequency of disturbances is available, it is difficult to draw conclusions on their effect on the vegetation. This is in part due to the method used in the LDSF to score habitat impact (Vågen et al, 2013). The regime of farming practices was only evaluated visually and not with quantifiable parameters (see methods). To accurately test the effects of tree cutting, fire and browsing on the regenerating tree community, an experimental setup over time could yield more insights.

Restoring resilient and multifunctional ecosystems with FMNR

There is evidence that FMNR is responsible for the rapid recovery of tree cover in some parts of the Sahel, with the environmental and economic benefits that came along (Francis & Weston, 2015). In landscapes with a history of anthropogenic disturbance, the extent to which FMNR can be successful in maintaining tree diversity depends on the natural replenishment of seed stock and dispersal mechanisms and on the environmental conditions that will allow the survival of young trees (Ordonez et al. 2013).

Considering the relatively low number of regenerating species found in the farmers' fields, it is reasonable to assume that FMNR may favor tree density, but not necessarily tree biodiversity. As explained earlier, tree diversity in these shifting cultivation systems depends, among other things, on a diverse agricultural matrix where establishment and regeneration sites (long fallows) as well as seed source sites (protected woodlands) are maintained. Apart from promoting natural regeneration on farmers' field, FMNR will also need to operate at a landscape scale, reflecting the spatial pattern of species regeneration and dispersal strategies (Usman & Nichol, 2018). There are examples where local communities took initiative to rehabilitate land and preserve tree biodiversity with different activities like soil conservation, tree planting and agroforestry improvement (Paré, 2008). In the study site, some villages were restoring and managing community-owned forests (forêts villageoises) as a way to conserve useful plant species found mainly in the protected area. The promotion of such initiatives by national and regional policies could greatly improve rural livelihoods while maintaining ecosystem-regulating services in the future parklands.

Authors have already stated the necessity of enrichment planting in order to assist tree regeneration, notably of valuable fruit tree species that fail to regenerate (Smith Dumont et al., 2014). Pro-active planting can be done directly in vegetation features like hedges and fallows, which provides better growing conditions than a bare field and low-cost protection against livestock and wind (Love et al., 2009).

The advantage of enrichment planting is that farmers can chose the species composition and density of the trees. It has also been suggested that tree species could be grafted on plant regenerating from rootstock, making use of the already developed root system, but to my knowledge this technique has not been tested yet. Where suitable pollinators are absent, beekeeping could be introduced to increase fruit yield and seed production (Ræbild, 2012). Successful technological innovations could then be introduced trough training courses and hands-on training for extension staff and village communities.

Conclusion

In this study, I characterized tree dynamics across the different land use types that constitute a typical parkland landscape of West Africa. Similar to previous studies, I found a change in species composition where the established vegetation, characterized by large single-trunked trees, is replaced by shrub species. Further analysis of functional composition revealed that the regenerating tree community is dominated by species with traits that confer resistance to dry conditions. This was particularly marked in land units like home and bush fields, suggesting that current land use management in the farmed parklands changed local conditions and may have favored the regeneration of drought-resistant species at the expense of valuable tree species (Hänke et al., 2016). I found that land use intensification also decreases not only the taxonomic diversity but also the functional diversity of the regenerating vegetation, putting at risk the long-term stability of this agroecosystem. This has considerable implications for FMNR and the extent to which it can alone restore tree diversity and associated ecosystem services. In order to move from simply increasing tree cover to restoring tree diversity, FMNR should be accompanied by a change in land planning and management where land units are maintained to allow replenishment of multiple tree species. This will require effective policies and actions that include farmers' knowledge and perception on use-preference, ecological conditions and vegetation dynamics of the parklands (Lykke et al., 2004). Technical innovation like enrichment planting and off-farm protection of individual trees may also be necessary where natural regeneration is failing.

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Appendices

Appendix 1: Detailed methods on functional trait assessment

Sampling strategy

Ten functional traits were measured on at least five individuals of each of the focal species. For focal species that occur both in Burkina Faso. The functional traits will be measured on random individuals in the area across different habitat range to account for intraspecific variation (Perez-Harguindeguy et al, 2013). Functional traits do not necessarily need to be measured individuals occurring in the LDSF plots, but for focal species that are relatively rare in the area, it is possible to find individuals using the GPS location of plots in which they have been encountered.

Whole-plant traits

Plant maximum height

Plant height (m) corresponds to the distance between the superior limit of the main photosynthetic tissues of a plant and the soil level (Cornelissen et al. 2003). Maximum height will be retrieved from the LDSF tree inventory by averaging the height of the three tallest individual of each species.

Plant maximum height is linked to competitive vigor, whole-plant fecundity and regeneration time between disturbance events (Pérez-Harguindeguy et al., 2013). There are also important trade-offs between maximum height and tolerance or avoidance of environmental stress (Cornelissen et al., 2003). For example, studies have showed that larger trees are more prone to drought-induced death than smaller ones in a variety of savanna ecosystems (O'Brien et al., 2017). Additionally, plant height tends to be correlated allometrically with the root depth (Cornelissen et al., 2003), which can be a good indicator of a tree strategy for water acquisition in situation of drought.

Deciduousness

Deciduousness refers to the ability (binary variable: 0=yes, 1= no) of a plant species to lose its leaves for a period of time during the year (Cornelissen et al., 2003). It will be determined using the literature farmers knowledge and field observations. Deciduous species shed their leaves before the dry season as a mechanism to prevents water loss and is therefore a strategy to avoid drought (Poorter & Markesteijn, 2007).

Leaf traits

Because leaf traits varies within plants, four leaves will be collected on each individual in order to obtain an accurate indication (Perez-Harguindeguy, 2013). I will select sun-exposed leaves from the outer canopy that are relatively young, fully expanded and that show no visible signs of damage. The leaves will be taken from the outer canopy where they have had proper sun exposure. In the field, the whole twig will be cut from the tree, and put into a coded plastic bag to protect the twig and its leaves. Once back at field base, we will cut the bottom part of the twig and they will be rehydrated in a bucket with water. The leaves will only be removed when starting with the measurement. For woody species with simple leaves, the individual leaf lamina will be measured. When it comes to compound-leaved species, the leaflet area as well as the whole leaf area will be measured. Furthermore, the petiole and rachis will be included in the measurements.

Leaf area (LA)

Leaf area is defined as the one-sided area (cm^2) an individual leaf (Cornelissen et al., 2003).
For measurement, leaves will be photographed on a white surface along with a precision ruler in the frame for calibration, and leaf area will be calculated using pixel counting software ImageJ (Schneider, Rasband & Eliceiri, 2012).
Leaf area is linked to allometric factors such as plant size, twig size, anatomy and architecture, and also to ecological strategy with respect to environmental nutrient stress and disturbances (Cornelissen et al., 2003). Plant species with relatively smaller leaves are usually more resistant to environmental stressors such as heat, cold, drought and high radiation (Pérez-Harguindeguy et al., 2013).

Specific leaf area (SLA)

SLA refers to the one-sided area of a fresh leaf divided by its oven-dry mass expressed in m^2 kg^{-1} (Cornelissen et al., 2003). Area is as determined for LA (Perez-Harguindeguy, 2013). To measure the leaf dry mass, each individual leaf will at first be stored in paper bags and hang on a line to dry inside an well-aerated room. The leaf dry mass will be weighted every 24h until it ceases to decrease (Pérez-Harguindeguy et al., 2013).
SLA is a soft trait often positively correlated with potential relative growth rate (RGR). A slow RGR has been associated with longer drought survival time (Lopez-Iglesias et al., 2014).

Leaf thickness

Leaf thickness (mm) will be determined with a digital caliper in the middle of the leaf after rehydration (Pérez-Harguindeguy et al., 2013). Leaf thickness is related to the physical strength in leaves, which plays an important role in protection against physical damage (e.g. herbivory) and contribute to longer lifespan (Pérez-Harguindeguy et al., 2013). It was also suggested that increased leaf thickness lower the risk of extreme heat stress and can enhance long-term survival for perennial desert flora (Leigh et al., 2012).

Leaf dry matter content (LDMC)

Leaf dry matter content is the oven-dry mass of a leaf divided by its water-saturated fresh mass, expressed in g g^{-1} (Cornelissen et al., 2003). Leaves will be weighted once after rehydration and again after having dried for a couple of days (when weight have ceased to decrease).
Leaves with high content of dry matter tend to be relatively strong, and it is assumed that they are more resistant to physical hazards such as herbivory and have a higher tolerance against water limitation and heat (Cornelissen et al., 2003; Poorter & Markesteijn, 2007).

Leaf compoundness

Compoundness of leaves (binary variable: 0=simple, 1= compound) will be assessed using field observation. Compounded-species have the ability to fold their leaflets in situation of high temperatures (e.g. at noon or during the dry season) to avoid high insolation and excessive evaporation (Lohbeck et al., 2015). Compoundness also affect positively thermoregulation and control of water loss (Yate et al., 2010).

Stem traits

Wood density

Wood density (g cm^3) is the oven-dry mass of a section of the main stem of a plant divided by the volume of the same section, when still fresh (Cornelissen et al., 2003). In order to avoid damaging the farmers trees, the WD information is based on the global wood density database (Chave et al., 2009). When a species was not part of the database, an average of the WD from the species within the same genus located in Africa was taken or, if the latter last one was not available either, from the whole tropic. Eventually if no species of the same genus was part of the database, an average was taken from the family located in Africa.

High wood density is usually associated with high survival in dry environment, because it is associated with the prevention of cavitation (Bucci et al., 2004) and xylem implosion by negative pressure (Hacke et al., 2001), and the ability to endure more negative water potential in hot conditions (Mitchel et al,. 2008). Wood density is also correlated with RGR, where a low density leads to a fast growth, because of cheap volumetric construction costs but lower survival (Pérez-Harguindeguy et al., 2013).

Twig dry matter content (TDMC)

Twig dry-matter content is the oven-dry mass of a terminal twig, divided by its water-saturated fresh mass, expressed in g g^{-1} (Pérez-Harguindeguy et al., 2013). On the field, one terminal (highest ramification-order; smallest diameter-class), sun-exposed twig of 20-30 cm long will be collected and stored in plastic bags. If the farmers allow it, 2 or 3 twigs will be collected to increase sample size. Rehydration and dry mass weighting procedures will be the same as the one used for LA. (Pérez-Harguindeguy et al., 2013)

TDMC is supposed to be positively correlated with wood density and thus represent a good alternative if wood density cannot be measured on the field for practical reasons. Additionally, the twigs with high dry matter content are expected to dry out relatively fast during the dry season (Cornelissen et al., 2003).

Appendix 2. List of species found in the study site and in each land-use type

Table 1 List of all the species found in the 160 plots of study site with the abundance of regenerating and established tree. Focal species for which functional traits were measured are highlighted in yellow. Functional traits were not measured for the focal species indicated in red because they turn out to be too difficult to find.

ID	**Species**	**Family**	**Number of regenerating tree**	**Number of established tree**
1	*Acacia dudgeonii*	Fabaceae	4	4
2	*Acacia gourmaensis*	Fabaceae	8	11
3	*Acacia macrostachya*	Fabaceae	4	2
4	*Acacia nilotica*	Fabaceae	0	2
5	*Acacia pennata*	Fabaceae	1	0
6	*Acacia seyal*	Fabaceae	2	13
7	*Acacia senegal*	Fabaceae	1	0
8	*Acacia sieberiana*	Fabaceae	0	5
9	*Adansonia digitata*	Malvaceae	0	4
10	*Afrormosia laxiflora*	Fabaceae	3	0
11	*Afzelia africana*	Fabaceae	0	6
12	*Annona senegalensis*	Annonaceae	35	0
13	*Anogeissus leiocarpus*	Combretaceae	34	83
14	*Balanites aegyptiaca*	Zygophillaceae	7	23
15	*Bombax costatum*	Malvaceae	0	8
16	*Bridelia ferriginea*	Phyllanthaceae	2	0
17	*Burkea africana*	Fabaceae	0	1
18	*Combretum fragrans*	Combretaceae	8	6
19	*Combretum glutinosum*	Combretaceae	80	2
20	*Combretum molle*	Combretaceae	8	8
21	*Combretum nigricans*	Combretaceae	55	20
22	*Crossopteryx febrifuga*	Rubiaceae	7	2
23	*Daniellia oliveri*	Fabaceae	1	0
24	*Detarium microcarpum*	Fabaceae	19	32
25	*Dichrostachys cinerea*	Fabaceae	21	0
26	*Diospyros mespiliformis*	Ebenaceae	13	21
27	*Entada abyssinica*	Fabaceae	0	1
28	*Entada africana*	Fabaceae	0	2
29	*Erythrina senegalensis*	Fabaceae	0	1
30	*Feretia apodanthera*	Rubiaceae	8	1
31	*Flueggea virosa*	Phyllanthaceae	16	0
32	*Ficus cycomorus*	Moraceae	0	3

33	*Ficus ingens*	Moraceae	0	1
34	*Ficus platyphylla*	Moraceae	0	1
35	*Gardenia aqualla*	Rubiaceae	1	0
36	*Gardenia erubecens*	Rubiaceae	11	0
37	*Gardenia ternifolia*	Rubiaceae	7	0
38	*Grewia bicolor*	Malvaceae	1	1
39	*Guierra senegalensis*	Combretaceae	50	0
40	*Lannea acida*	Anacardiaceae	2	45
41	*Lannea microcarpa*	Anacardiaceae	2	40
42	*Lannea velutina*	Anacardiaceae	0	1
43	*Maytenus senegalensis*	Celastraceae	8	1
44	*Mitragyna inermis*	Rubiaceae	0	6
45	*Parkia biglobosa*	Fabaceae	0	9
46	*Piliostigma reticulatum*	Fabaceae	2	1
47	*Piliostigma thonningii*	Fabaceae	76	12
48	*Prosopis africana*	Fabaceae	0	2
49	*Pteleopsis suberosa*	Combretaceae	1	0
50	*Pterocarpus erinaceus*	Fabaceae	2	6
51	*Saba senegalensis*	Apocynaceae	1	1
52	*Sclerocarya birrea*	Anacardiaceae	7	20
53	*Sterculia setigera*	Malvaceae	0	3
54	*Stereospermum kunthianum*	Bigoniaceae	0	3
55	*Strychnos spinosa*	Loganiaceae	9	1
56	*Tamarindus indica*	Fabaceae	0	7
57	*Terminalia avicennioides*	Combretaceae	16	14
58	*Terminalia laxiflora*	Combretaceae	6	5
59	*Terminalia macroptera*	Combretaceae	1	0
60	*Vitelaria paradoxa*	Sapotaceae	73	201
61	*Ximenia americana*	Olacaceae	11	1
62	*Ziziphus mauritiana*	Rhamnaceae	1	0

Table 2 Species abundance of the established (E) and the regenerating (R) tree community in each land-use type (BZ= buffer zone; PA= protected area; LF= long fallow; SF= short fallow; BF = bush fields; HF= home fields).

	BZ		**PA**		**LF**		**SF**		**BF**		**HF**	
	E	R	E	R	E	R	E	R	E	R	E	R
Acacia seyal	0	0	1	0	1	0	10	2	0	0	0	0
Acacia dudgeonii	0	3	1	0	3	0	0	0	0	0	0	1
Acacia gourmaensis	2	4	8	4	1	0	0	0	0	0	0	0
Acacia macrostachya	2	4	0	0	0	0	0	0	0	0	0	0
Acacia nilotica	0	0	0	0	0	0	2	0	0	0	0	0
Acacia pennata	0	1	0	0	0	0	0	0	0	0	0	0
Acacia senegal	0	1	0	0	0	0	0	0	0	0	0	0
Acacia sieberiana	3	0	0	0	0	0	2	0	0	0	0	0
Afromosia laxiflora	0	3	0	0	0	0	0	0	0	0	0	0
Adansonia digitata	0	0	1	0	0	0	1	0	0	0	2	0
Afzelia africana	1	0	0	0	3	0	0	0	2	0	0	0
Annona senegalensis	0	20	0	2	0	4	0	1	0	2	0	6
Anogeissus leiocarpus	27	24	46	5	7	5	2	0	0	0	1	0
Balanites aegyptiaca	2	1	12	3	5	0	0	2	2	0	2	1
Bridelia ferriginea	0	1	0	0	0	1	0	0	0	0	0	0
Bombax costatum	2	0	1	0	2	0	1	0	2	0	0	0
Burkea africana	0	0	1	0	0	0	0	0	0	0	0	0
Combretum fragrans	0	0	6	5	0	1	0	2	0	0	0	0
Combretum glutinosum	0	11	1	0	1	20	0	8	0	17	0	24
Combretum molle	0	4	8	2	0	0	0	0	0	2	0	0
Combretum nigricans	1	31	11	19	5	5	0	0	3	0	0	0
Crossopteryx febrifuga	2	4	0	2	0	0	0	0	0	1	0	0
Daniellia oliveri	0	0	0	0	0	1	0	0	0	0	0	0
Detarium microcarpum	3	11	20	2	8	2	0	0	1	2	0	2
Dichrostachys cinerea	0	4	0	0	0	3	0	0	0	13	0	1
Diospyros mespiliformis	1	2	2	0	5	9	4	0	5	0	4	2
Entada abyssinica	0	0	0	0	0	0	0	0	1	0	0	0
Entada africana	1	0	0	0	1	0	0	0	0	0	0	0
Erythrina senegalensis	0	0	0	0	0	0	0	0	0	0	1	0
Feretia apodanthera	0	2	0	0	0	2	0	0	0	1	0	3
Ficus cycomorus	0	0	0	0	1	0	0	0	0	0	2	0
Ficus ingens	0	0	0	0	1	0	0	0	0	0	0	0
Ficus platyphylla	0	0	0	0	0	0	0	0	0	0	1	0
Flueggea virosa	0	10	0	3	0	0	0	1	0	1	0	1
Gardenia aqualla	0	1	0	0	0	0	0	0	0	0	0	0

Gardenia erubecens	0	4	0	1	0	0	0	1	0	4	0	1
Gardenia ternifolia	0	4	0	1	0	1	0	0	0	0	0	1
Grewia bicolor	1	0	0	0	0	0	0	0	0	1	0	0
Guierra senegalensis	0	12	0	0	0	15	0	17	0	0	0	6
Lannea acida	17	1	9	0	15	1	1	0	3	0	0	0
Lannea microcarpa	5	0	2	0	19	2	2	0	4	0	9	0
Lannea velutina	1	0	0	0	0	0	0	0	0	0	0	0
Maytenus senegalensis	0	2	0	1	0	1	0	0	1	1	0	3
Mitragyna inermis	0	0	0	0	4	0	0	0	0	0	2	0
Parkia biglobosa	0	0	0	0	2	0	0	0	7	0	9	0
Piliostigma reticulatum	1	0	0	0	0	2	0	0	0	0	0	0
Piliostigma thonningii	3	20	0	1	5	13	1	11	2	12	1	19
Pteleopsis suberosa	0	1	0	0	0	0	0	0	0	0	0	0
Prosopis africana	1	0	1	0	0	0	0	0	0	0	0	0
Pterocarpus erinaceus	3	0	1	0	2	0	0	0	0	0	0	2
Saba senegalensis	0	0	0	1	0	0	0	0	0	0	1	0
Sclerocarya birrea	12	0	1	0	4	0	0	1	0	1	3	1
Sterculia setigera	0	0	1	0	2	0	0	0	0	0	0	0
Stereospermum kunthianum	0	0	1	0	0	0	1	0	1	0	0	0
Strychnos spinosa	0	5	1	4	0	0	0	0	0	0	0	0
Tamarindus indica	0	0	4	0	0	0	0	0	0	0	3	0
Terminalia avicennioides	4	7	9	3	0	1	0	1	0	0	1	4
Terminalia laxiflora	2	4	3	2	0	0	0	0	0	0	0	0
Terminalia macroptera	0	0	0	0	0	0	0	0	0	1	0	0
Vitelaria paradoxa	41	17	43	5	37	15	5	7	28	22	47	7
Ximenia americana	0	10	1	1	0	0	0	0	0	0	0	0
Ziziphus mauritiana	0	0	0	0	0	0	0	0	0	0	0	1

Appendix 3. Results of difference in functional composition between regenerating and established trees

Table 1 Statistics of the Wilcoxon signed-rank test for the differences in functional composition. V correspond to the value of the signed rank statistic. Significant differences are indicated in bold.

Land-use	Wilcoxon signed-rank test		Land-use	Wilcoxon signed-rank test	
	V	*P*		*V*	*P*
Buffer zone			**Short fallow**		
Max. height	176	**<0.001**	Max. height	20	0.0625
LA	429	**<0.001**	LA	13	0.6875
SLA	104	**<0.01**	SLA	6	0.4375
LT	61	**<0.001**	LT	4	0.2188
LDMC	157	0.1241	LDMC	11	1
Compound	308.5	**<0.001**	Compound	3	1
Deciduous	101	**<0.05**	Deciduous	1	1
TDMC	75	**<0.001**	TDMC	11	1
WD	55.5	**<0.001**	WD	5	0.3125
Protected area			**Bush field**		
Max. height	51	**<0.05**	Max. height	176	**<0.001**
LA	42	0.1602	LA	165	**<0.05**
SLA	12	0.1309	SLA	145	0.1429
LT	9	0.06637	LT	0	**<0.001**
LDMC	28	1	LDMC	58	0.08255
Compound	45	0.08398	Compound	80	**<0.05**
Deciduous	49	**<0.05**	Deciduous	1	0.1056
TDMC	39	0.2754	TDMC	22	**<0.001**
WD	17	0.3223	WD	6	**<0.001**
Long fallow			**Home field**		
Max. height	114	**<0.05**	Max. height	91	**<0.001**
LA	153	**<0.001**	LA	101	**<0.05**
SLA	48	0.185	SLA	52	0.6788
LT	27	**<0.05**	LT	5	**<0.01**
LDMC	15	**<0.01**	LDMC	26	0.05536
Compound	104	**<0.01**	Compound	55	0.05569
Deciduous	28	1	Deciduous	12	0.8339
TDMC	6	**<0.001**	TDMC	14	**<0.01**
WD	5	**<0.001**	WD	7	**<0.01**

Appendix 4. Statistics for the optimal models

Table 1 Statistics for the optimal models explaining a) richness; b) taxonomic diversity and c) functional diversity. Given are the estimates and associated confidence intervals. P-values reflect the z-associated p-value and significance is indicated in bold.

a) Richness ~ land use + fire + livestock + tree cutting

Predictor	**Estimate**	**CI**	***P***
Intercept	0.23	–0.32 to 0.77	0.41
Protected area	1.017	0.50 to 1.52	**<.001**
Buffer zone	1.054	0.59 1.52	**<.001**
Long fallow	0.599	0.13 to 1.07	**<.05**
Short fallow	0.314	–0.26 to 0.86	0.27
Bush field	0.590	0.17 to 1.00	**<.01**
Impact fire	0.085	–0.04 to 0.21	0.18383
Impact livestock	–0.271	–0.49 to –0.04	**<.05**
Impact tree cutting	0.214	0.04 to 0.38	**<.05**

b) Taxonomic diversity ~ land use + tree cutting

Predictor	**Estimate**	**CI**	***P***
Intercept	0.02	–0.01 to 0.07	0.17
Protected area	0.217	0.12 to 0.30	**<.001**
Buffer zone	0.251	0.18 to 0.31	**<.001**
Long fallow	0.082	0.002 to 0.16	**<.05**
Short fallow	0.055	–0.03 to 0.14	0.21
Bush field	0.065	0.005 to 0.12	**<.05**
Impact tree cutting	0.030	0.0001 to 0.06	**<.05**

c) Functional diversity ~ Land use

Predictor	**Estimate**	**CI**	***P***
Intercept	0.007	0.003 to 0.01	**<.001**
Protected area	0.008	–0.0003 to 0.01	0.059
Buffer zone	0.011	0.005 to 0.01	**<.001**
Long fallow	0.011	0.004 to 0.01	**<.01**
Short fallow	0.003	–0.004 to 0.01	0.41
Bush field	–0.001	–0.008 to 0.005	0.65